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Depth investigation of rapid sand filters for drinking water production reveals strong stratification in nitrification biokinetic behavior

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Abstract

The biokinetic behavior of NH_4^+ removal was investigated at different depths of a rapid sand filter treating groundwater for drinking water preparation. Filter materials from the top, middle and bottom layers of a full-scale filter were exposed to various controlled NH_4^+ loadings in a continuous-flow lab-scale assay. NH_4^+ removal capacity, estimated from short term loading up-shifts, was at least 10 times higher in the top than in the middle and bottom filter layers, consistent with the stratification of Ammonium Oxidizing Bacteria (AOB). AOB density increased consistently with the NH_4^+ removal rate, indicating their primarily role in nitrification under the imposed experimental conditions. The maximum AOB cell specific NH_4^+ removal rate observed in the bottom was at least 3 times lower compared to the top and middle layers. Additionally, a significant up-shift capacity (4.6 and 3.5 times) was displayed from the top and middle layers, but not from the bottom layer at increased loading conditions. Hence, AOB with different physiological responses were active at the different depths. The biokinetic analysis predicted that despite the low NH_4^+ removal capacity at the bottom layer, the entire filter is able to cope with a 4-fold instantaneous loading increase without compromising the effluent NH_4^+ . Ultimately, this filter up-shift capacity was limited by the density of AOB and their biokinetic behavior, both of which were strongly stratified.

Keywords

Groundwater treatment, Ammonium Oxidizing Bacteria, Nitrification capacity, Nitrification kinetics, Loading

1. Introduction

Rapid sand filtration is widely used in drinking water production to remove compounds such as NH_4^+ , Fe^{2+} and Mn^{2+} , which are typical for anoxic groundwaters. Among these compounds, Fe^{2+} and Mn^{2+} are chemically and biologically oxidized to low-solubility oxyhydroxides and are removed by precipitation in the filter, while NH_4^+ is biologically oxidized to NO_3^- . EU member states must comply with the guideline limit of 0.5 mg/L NH_4^+ in the water leaving the waterworks (Council Directive 98/93/EC 1998), while stricter limits can be applied by the individual member states e.g. 0.05 mg/L NH_4^+ in Denmark (Danish Ministry of Environment 2014). Guideline limits for NH_4^+ are set to ensure biological stability in the down-stream distribution network in non-chlorinated systems (Chu et al. 2005), and to control the disinfection residual in chlorinated systems (Zhang et al. 2009).

Single media filters are typically assumed to be homogeneous due to their frequent backwashing (Uhl and Gimbel 2000). However, recent studies have shown that some filters have heterogeneous flow patterns (Lopato et al. 2013) and stratified biomass distributions even with regular backwashing (Albers et al. 2015, Bai et al. 2013, Feng et al. 2013). Stratification can be the result of changes in the filter material density if mineral precipitates form a persistent coating on the sand grains (Gülay et al. 2014). In this case, the lighter mineral-coated grains accumulate preferentially at the top of the filter after each backwashing. There, they are exposed to higher NH_4^+ loadings compared to the heavier less coated grains, which reside deeper in the filter. Stratification of both nitrifying bacterial density and nitrification activity has, indeed, been observed in samples collected from full-scale filters (de Vet et al. 2009, Kihn et al. 2000, Madoni et al. 2001). The assays used in these works involved strong physical mixing that is likely to detach the microbial activity from the filter material, poorly reflecting the *in situ* biokinetics. Proper estimation of the *in situ* biokinetics would maintain the spatial attachment of the biomass, and the associated supposed mass transfer limitations.

Stratification of nitrification activity was also observed in pilot-scale filters (Lee et al. 2014, van den Akker et al. 2008). Specifically, Lee and collaborators investigated filter response to short-term NH_4^+ loading increases, created by increasing either the influent NH_4^+ concentration or the flowrate. No difference was observed between the two increased loading scenarios, and the overall filter capacity was estimated to be 5 times higher than the normal operating loading (Lee et al. 2014). Most of this additional capacity was concentrated in the top 0.15 m, while deeper filter layers exhibited very small additional capacities (Lee et al. 2014). This stratification in activity was solely attributed to the change of Ammonium Oxidizing Bacteria (AOB) density with depth, assuming that AOB have the same biokinetic behavior everywhere in the filter (Lee et al. 2014). The biokinetic behavior itself, however, was not directly examined for stratification with depth.

In the present study, we investigate in detail the previously observed limited nitrification capacity at the bottom layer of a rapid sand filter. We aim to examine whether this stratification in activity is exclusively the result of a decrease in AOB density. Hence, the NH_4^+ removal capacity at different depths of a full-scale filter was investigated by exposing the filter material to a series of controlled loading conditions in an offline continuous-flow assay. Molecular analysis before and after the experiments was used to identify the predominant nitrifiers, and to calculate cell specific NH_4^+ removal rates at each depth. The activity observations were augmented by a 1-D biofilm model analysis to estimate the kinetic parameters at each depth. Ultimately this analysis aimed to reveal if the NH_4^+ removal biokinetics is also stratified in the filter.

2. Materials & Methods

2.1. Filter core sampling and depth profiling

Islevbro (Copenhagen, Denmark) waterworks was selected due to the strong activity stratification observed in previous pilot-scale investigations (Lee et al. 2014). The raw water is abstracted from a deep chalk aquifer and the treatment train consists of an aeration step, followed by a retention tank providing about 20 min contact time for oxidation of Fe^{2+} , and a double filtration step consisting of pre- and after-filters. Pre-filters have a bed of coarse stones (3-6 cm diameter) to retain the formed Fe-hydroxides, while after-filters are designed to biologically remove NH_4^+ . After-filters have an average influent flowrate of 1.73×10^6 L/d, an influent concentration of $0.13 (\pm 0.04)$ mg/L NH_4^+ -N, a cross sectional area of 18 m^2 and are 0.70 m deep. The hydraulic loading is 4 m/h and the filters have been in operation for approx. 30 years without filter material replacement. Nominal grain

diameter is 1 mm and after-filters are backwashed every 14 days by air scouring (3 min at 90 m/h) and high water flowrates (10 min at 25 m/h).

Filter material from one after-filter was core-sampled by inserting and gently removing a plexiglass cylinder (1 m height and 5 cm inner diameter), closed at the top with a rubber stopper. From the collected 0.50 m filter material core three depth layers were separated: 0-0.10 m (top), 0.20-0.30 m (middle) and 0.35-0.50 m (bottom) that were used for the lab-scale NH_4^+ removal investigations. The full-scale filter was core-sampled at 3 randomly selected locations and composite samples were created by mixing filter material from the same depth. Sampling took place approximately midway between two backwash events (day 9 after last backwashing). Additionally, 6 filter material sub-samples were segregated every 0.05-0.10 m from the collected core for molecular quantification of AOB and Ammonium Oxidizing Archaea (AOA). Filter material samples for the NH_4^+ removal investigations were stored wet at 10°C for 4 days, while sub-samples for molecular quantification were drained and stored at -20°C.

The *in situ* NH_4^+ profile in the filter was obtained by sampling water at 0.05, 0.1, 0.15, 0.2, 0.3 and 0.4 m depth during 6 sampling campaigns conducted over 2 filter-run cycles. Stainless steel pipes were fixed in the middle of the filter and were connected to a multi-channel peristaltic pump (110 ACR, Ole Dich) sampling water at a 0.6 L/h. Water samples were immediately filtered (Sartorius Minisart 0.20 μm), frozen and analyzed for NH_4^+ (Merck Spectroquant test kit 1.14752) within 7 days. The *in situ* volumetric NH_4^+ loading at each filter depth was calculated as the product of the measured NH_4^+ concentration and the flowrate, normalized for the packed filter material volume of the respective depth section.

2.2. Quantification of depth specific NH_4^+ removal rates

Filter material from the three investigated depth layers was exposed to a series of NH_4^+ loading levels in a lab-scale column assay to observe the removal rate at each imposed condition. The assay is described in details elsewhere (Tatari et al. 2013). In brief, it consisted of small (5 cm bed height, 2.6 cm inner diameter) columns packed with the collected filter material that were operated under continuous flow conditions. Filter material from each depth layer was packed in separate columns and the three columns were operated in parallel. The effluent from each column was recirculated at a high recirculation ratio (50) in order to approximate completely-mixed bulk hydrodynamic conditions (Tatari et al. 2013). Each column was supplied with effluent water from Islevbro waterworks, supplemented with 1 mg/L NH_4^+ -N (as NH_4Cl , Merck chemicals) at an influent flowrate of 0.94 L/d. Recirculation for each column was at 46.8 L/d and the hydraulic loading (influent plus recirculation) was 3.8 m/h, matching full-scale conditions (Tatari et al. 2013). The resulting volumetric (expressed per volume of packed filter material) loading was 35 g NH_4^+ -N/m³ filter material/d, termed the reference loading. The reference loading was chosen as an equivalent to the full-scale filter loading, based on the initial assumption that complete NH_4^+ removal occurs in the upper 0.40 m of the filter. The *in situ* loading at each depth was calculated based on the depth profiles and was compared to the reference loading. The experimental set-up is schematically illustrated in the SI (Figure S1).

The columns were operated at reference loading for 9 days, with intermittent short-term (3 to 5 hours) loading up-shifts (Figure S2 in SI). The loading up-shifts were performed within the first 4 days of operation, and the columns were operated for an additional 5 days to identify growth of nitrifiers. The loading was up-shifted to 88, 175 and 350 g NH_4^+ -N/m³ filter material/d by increasing the influent flowrate to 2.34, 4.68 and 9.36 L/d at day 3, 4 and 2 of operation, respectively, while maintaining the influent concentration at 1 mg NH_4^+ -N/L. Each loading up-shift lasted 3-5 h to prevent substantial growth of nitrifiers during these short-term perturbations.

External mass transfer conditions were maintained by keeping a nearly steady total (influent and recirculation stream) hydraulic loading to the columns (Tatari et al. 2013). Experiments were carried out at 10°C to mimic the *in situ* water temperatures. Column influents and effluents were manually sampled and analyzed for NH_4^+ and NO_2^- (Merck Spectroquant test kits 1.14752 and 1.14776 with detection limits of 0.01 and 0.002 mg N/L respectively). The volumetric removal rate at each loading was calculated from concentration differences multiplied by the influent flowrate and normalized for the volume of packed filter material in the column. The maximum observed NH_4^+ removal rate from each column was termed the NH_4^+ removal capacity and was used to predict the full-scale filter performance as described in section 2.5. At the end of the experiments, filter material from each column was collected and stored at -20°C for molecular quantification.

2.3. Molecular quantification of total bacteria and nitrifiers

Total *Eubacteria*, *Nitrospira*, AOB and AOA were quantified by real-time quantitative PCR (qPCR). Genomic DNA was extracted in triplicate from 0.5 g drained-wet filter material before and after the column assays (FastDNA SPIN, MP Biomedicals LLC). The concentration and purity of the extracted DNA were estimated by measuring absorbance at 260 and 280 nm (NanoDrop Products). A group-specific region on the 16S ribosomal-RNA gene (16S-rRNA) was targeted by the primer sets 1055f and 1392R for Total *Eubacteria* (Ferris et al. 1996, Lane 1991), and CTO189A/B/C and RT1 for AOB (*Nitrosomonas* plus *Nitrospira* clades) (Hermansson and Lindgren 2001) and Nspra675f and Nspra746r for the *Nitrospira* genus (Graham et al. 2007). AOB and AOA were additionally quantified at the 6 sub-sampled depths by targeting the ammonia monooxygenase (*amoA*) functional gene, using the primer sets amoA1f and amoA2r for AOB (Rotthauwe et al. 1997) and amoAF and amoAR for AOA (Francis et al. 2005). The qPCR analyses were conducted in triplicate for each DNA-extract in a Chromo 4 thermocycler operated by the Opticon Monitor 3 software (Bio-Rad). Each qPCR reaction contained 12.5 μl of 2 \times iQ SYBR Green Supermix (Bio-Rad), 500 nM of each primer, DNA template (10 ng) and DNA/RNA-free water (Mol. Bio.) to 25 μl . The thermal cycling conditions consisted of an initial 5 min denaturation at 95°C, followed by 40 cycles of 30 s at 94°C, primer annealing for 30 s at 55°C for Total *Eubacteria*, 30 s at 56°C for AOB (16S-rRNA), 40 s at 55°C for AOB (*amoA*), 30 s at 64°C for *Nitrospira*, 60 s at 60°C for AOA, and 1 min extension at 72°C. After the 40th cycle, a final DNA extension at 72°C for 10 min was performed. A melting curve analysis was run (gradient 0.2°C/s, range 70-95°C) to confirm specific amplification (Ririe et al. 1997). The gene copy number was obtained by comparing the cycle threshold values of the sample against a standard curve for each qPCR target. The gene copy numbers were converted to cell density/g drained-wet filter material under the assumption that one bacterial cell contains one 16S-rRNA gene copy (Lee et al. 2006). Volumetric (expressed per volume of packed filter material) densities were calculated by multiplying the cell density/g drained-wet filter material with the drained-wet packed bulk density at each depth. To measure the drained-wet packed bulk density, filter material from each depth was drained by gravity for 10 min and weighed. The drained-wet packed bulk density was calculated from the drained-wet mass and the volume of the packed filter material in a graduated cylinder (Table S1 in SI).

2.4. Biofilm model outline

The experimental observations from the depth specific investigations were interpreted by a mathematical model to estimate the kinetic parameters at each depth layer. A model for each lab-scale column was implemented in Aquasim (Reichert 1998) and conceptually considered reaction and mass transfer at the biofilm scale, and calculated the effluent NH_4^+ and NO_2^- concentrations by mass balances at the macro-scale. The biofilm on the surface of spherical filter material grains was considered continuous and homogeneous.

Nitrification was modeled as a two-step reaction: NH_4^+ oxidation to NO_2^- by AOB, and NO_2^- oxidation to NO_3^- by nitrite oxidizing bacteria (NOB). Growth was described by a single substrate limited Monod function for AOB and by a 1st order function for NOB, since no significant NO_2^- accumulation ($< 0.15 \text{ mg NO}_2^- \text{-N/L}$) was detected during any of the experiments. Biomass loss was described by a 1st order function with an assumed coefficient b equal to 0.05 d^{-1} for both AOB and NOB (Queinnec et al. 2006) to include both cell decay and detachment. External mass transfer of dissolved components was subject to a boundary layer with thickness calculated at $4.2 \times 10^{-5} \text{ m}$ (Tatari et al. 2013), whereas diffusion in the biofilm was modeled considering that biofilm diffusivity equals 0.8 times the water diffusivity (Eberl et al. 2006). Diffusivity in water, calculated at 10°C by the Wilke-Chang equation (Wilke and Chang 1955), was $1.34 \text{ m}^2/\text{d}$ for NH_4^+ , NO_2^- and NO_3^- , and $1.27 \text{ m}^2/\text{d}$ for O_2 . Influent dissolved O_2 and $\text{NH}_4^+\text{-N}$ concentrations were 8 and 1 mg/L respectively. Growth yields were assumed at 0.14 and 0.08 g biomass-COD/g N removed for AOB and NOB, respectively (Queinnec et al. 2006).

The model for each column was calibrated by fitting the observed NH_4^+ and NO_2^- effluent concentrations during the 3 loading up-shifts simultaneously. Initial AOB density at each up-shift was the experimentally determined density before the experiments, corrected for growth until the time of the up-shift. Density increase due to growth was calculated by multiplying the AOB yield with the mass of NH_4^+ removed until the time of each up-shift. Due to the unexpectedly high density of *Nitrospira* compared to AOB (Section 3.2) and the uncertainty on their actual role in NH_4^+ removal under the tested conditions, the initial NOB biofilm density was estimated by a 3 to 1 ratio of AOB to NOB densities based on electron stoichiometry (Hagopian and Riley 1998). Details on the calculation of biofilm densities and the parameter estimation algorithm are in the SI (Sections II and III, respectively). For each column the estimated parameters were the half-saturation coefficient for NH_4^+ removal K_S (mg $\text{NH}_4^+\text{-N/L}$), and the maximum specific NH_4^+ utilization rate q_{max} (g $\text{NH}_4^+\text{-N/g COD/d}$). For NOB, a 1st order growth coefficient was estimated, although the values are not reported because the NOB density was not experimentally determined. The estimated kinetic parameters were used to predict performance from the three columns at reference loading. Uncertainty intervals of effluent performance predictions were based on parameters estimate variation within their standard deviation.

2.5. Prediction of filter performance based on depth-specific NH_4^+ removal capacities

The observed depth-specific NH_4^+ removal capacities were used to predict full-scale filter performance at instantaneous loading up-shifts, generated by increasing either the influent NH_4^+ concentration or the influent flowrate. Essentially, the filter was considered to consist of 3 depth sections, top (0-0.10 m), middle (0.10-0.30 m) and bottom (0.30-0.50 m), wherein volumetric removal occurred at the experimentally observed NH_4^+ removal capacity of the respective depth layer. Using appropriate mass-balances across these sequential sections, the NH_4^+ effluent concentration could then be computed under increasing loading applied to the top section. Details of the procedure are in section VIII of the SI.

3. Results & Discussion

3.1. Observed NH_4^+ removal rates and removal capacity

Filter materials from top, middle and bottom layers were exposed to a reference loading of $35 \text{ g NH}_4^+\text{-N/m}^3/\text{d}$ for 9 days in a lab-scale column assay, with influent concentration constant at $1 \text{ mg NH}_4^+\text{-N/L}$. The top layer column displayed complete removal from day 1, and throughout the entire experiment (Figure 1A). Effluent $\text{NH}_4^+\text{-N}$ from the middle layer column was 0.70 mg/L 1 day after

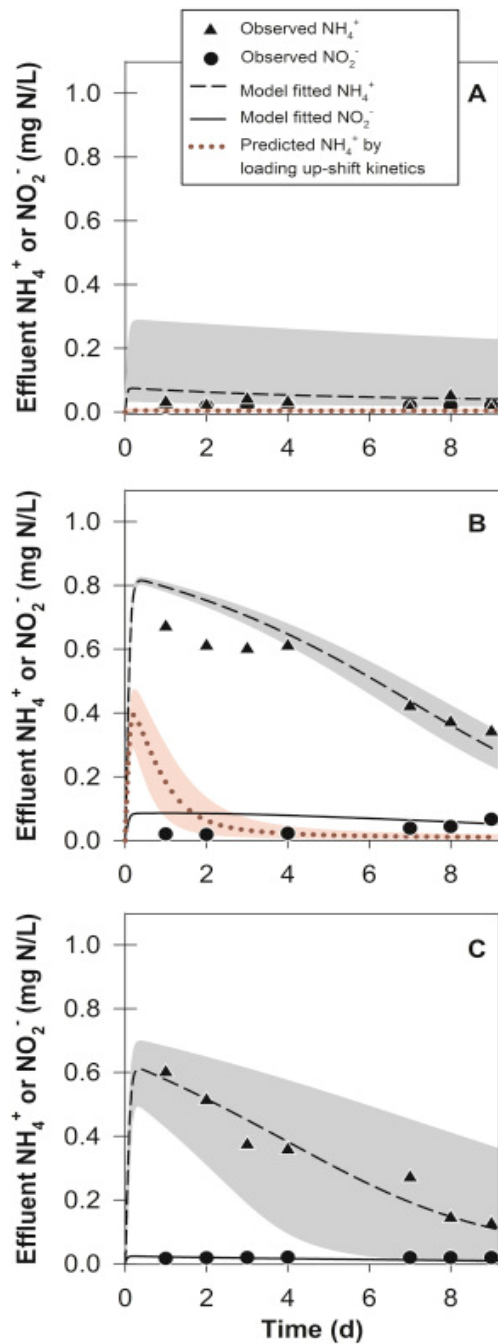


Figure 1. Observed and model fitted effluent NH_4^+ and NO_2^- concentrations from the top (A), middle (B) and bottom (C) layer columns at reference loading ($35 \text{ g/NH}_4^+\text{-N/m}^3$ filter material/d). The green line and green shaded areas in panels A and B illustrate the NH_4^+ effluent concentration and its uncertainty interval predicted by applying the kinetic parameters estimated at loading up-shifts. The black lines and grey shaded areas illustrate the concentrations fitted at reference loading and their uncertainty interval, after estimation of new maximum specific NH_4^+ utilization rates. Uncertainty intervals were calculated from the parameters estimate variation within their standard deviation.

start-up, and decreased gradually to 0.34 mg/L on day 9 (Figure 1B). The bottom layer column showed a similar trend with the effluent $\text{NH}_4^+\text{-N}$, decreasing from 0.60 to 0.12 mg/L over the 9 days (Figure 1C). $\text{NO}_2^-\text{-N}$ effluent concentrations were below $0.15 \text{ mg NO}_2^-\text{-N/L}$ for all columns. The low effluent $\text{NO}_2^-\text{-N}$ concentrations indicate an excess of NO_2^- vs NH_4^+ removal activity. The highest NO_2^- concentration was observed on days 7-9 in the middle layer column (Figure 1B), possibly caused by the rapid increase in the NH_4^+ removal activity. Different performances of the three layers on day 1 indicate stratification of NH_4^+ removal in the filter, with the highest removal rate at the top, consistent with the pilot-scale observations by Lee et al. (2014).

During the experiment, the loading was transiently up-shifted for 3-5 h to observe the effluent NH_4^+ and NO_2^- concentration responses from the three columns. Influent concentration was maintained at $1 \text{ mg NH}_4^+\text{-N/L}$. Loading up-shift to $88 \text{ g NH}_4^+\text{-N/m}^3$ filter material/d did not affect the $\text{NH}_4^+\text{-N}$ effluent concentration in the top layer column (Figure 2A). At up-shifts to 175 and $350 \text{ g NH}_4^+\text{-N/m}^3$ filter material/d, the effluent $\text{NH}_4^+\text{-N}$ concentration increased and stabilized to new pseudo steady-state values of $0.09 (\pm 0.02)$ and $0.34 (\pm 0.02) \text{ mg/L}$, respectively. For the middle layer column, the effluent $\text{NH}_4^+\text{-N}$ concentration increased from $0.60 (\pm 0.01) \text{ mg/L}$ at reference loading to $0.73 \text{ mg/L} (\pm 0.03)$ at the lowest loading up-shift and to $0.86 (\pm 0.05) \text{ mg/L}$ at the highest loading up-shift (Figure 2B). Similarly, for the bottom layer column, the effluent increased from $0.37 (\pm 0.09)$ at reference loading to $0.68 (\pm 0.02) \text{ mg/L}$ at the lowest loading up-shift and to $0.93 (\pm 0.10) \text{ mg/L}$ at the highest loading up-shift (Figure 2C). Overall, the top layer column maintained

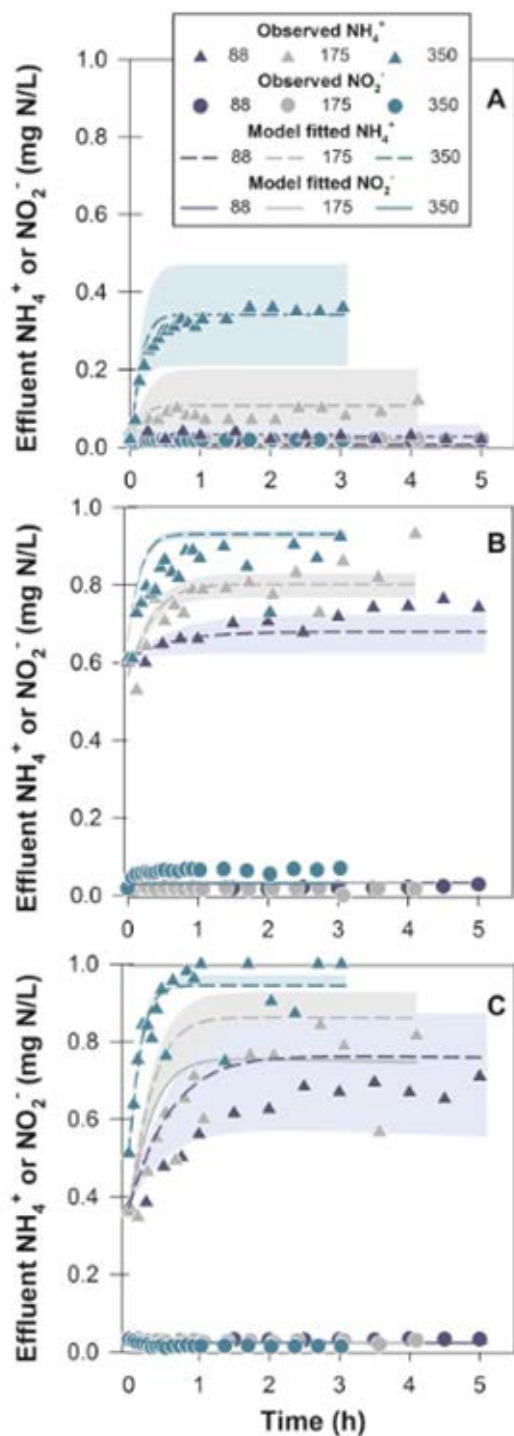


Figure 2. Observed and model fitted effluent NH_4^+ and NO_2^- concentrations from the top (A), middle (B) and bottom (C) layer columns during the short-term loading up-shifts to 88, 175 and 350 $\text{g}/\text{NH}_4^+\text{-N}/\text{m}^3$ filter material/d. Shaded areas illustrate the uncertainty interval of the fitted NH_4^+ effluent concentrations at each up-shift, calculated from the parameters estimate variation within their standard deviation.

the lowest NH_4^+ effluent concentrations at all investigated loadings.

Based on these pseudo steady-state NH_4^+ concentrations, the removal rate for each depth layer was calculated at all loading conditions (Table 1). The maximum observed NH_4^+ removal rate from the bottom layer column was 28 $\text{g NH}_4^+\text{-N}/\text{m}^3$ filter material/d, from the middle layer column 42 $\text{g NH}_4^+\text{-N}/\text{m}^3$ filter material/d, and from the top layer column 223 $\text{g NH}_4^+\text{-N}/\text{m}^3$ filter material/d (Table 1). These maximum observed removal rates are essentially the NH_4^+ removal capacities at the respective depths, except for the top layer where a true maximum was not yet reached within the investigated loading range.

Overall, these observations demonstrate strong stratification of NH_4^+ removal and capacity. These NH_4^+ removal capacities are consistent with the ones estimated by Lee et al. (2014) in their pilot-scale study at the same waterworks that ranged from roughly 250 $\text{g NH}_4^+\text{-N}/\text{m}^3$ filter material/d at the top to 35 $\text{g NH}_4^+\text{-N}/\text{m}^3$ filter material/d at the bottom of the filter. Capacity estimation using the lab-scale column assay is much quicker, needing only 3 days instead of the long start-up and stabilization times required for pilot-scale investigations. Moreover, the main advantage of this method is that the loading applied at the deeper layers can be controlled, while in pilot-scale filters loadings are dictated by the removal from the top layer.

3.2. Active NH_4^+ oxidizers

During reference loading conditions, the effluent NH_4^+ concentration from the middle and bottom filter layers decreased with time, suggesting growth of active NH_4^+ oxidizers. *Nitrosomonas* and *Nitrospira* (hereafter referred to together as AOB) are typically considered as the active NH_4^+ oxidizers, although AOA have also been identified in drinking water filters (Bai et al. 2013, de Vet et al. 2011, Kasuga et al. 2010, Niu et al. 2013) and are present at the investigated waterworks (Gülay et al. 2014, Lee et al. 2014). Moreover, some *Nitrospira* have recently been described to have the potential of oxidizing both

NH_4^+ and NO_2^- , carrying out the complete NH_4^+ oxidation pathway (Daims et al. 2015, Palomo et al. 2016, van Kessel et al. 2015). To observe if growth of any of the above microbial types was related with the activity increase in the middle and bottom layer columns, we examined their densities before and after the column experiments.

Only for AOB there was a substantial increase in density observed for the middle and bottom layer columns, at ca. 700 and 200% respectively (Figure 3). In the top layer column, AOB density increased by 30% during the 9 days of experimental time (Figure 3). Densities of other microbial types did not consistently increase in the middle and bottom layer columns. AOB growth is therefore associated with the observed activity increase, and AOB were likely the main NH_4^+ oxidizers during the experiments. Additionally, initial AOB densities varied consistently with the observed activity variation, with the highest density measured at the top ($31 \pm 12 \times 10^{12}$ cells/ m^3 filter material) and lower densities measured at the middle ($2.3 \pm 0.4 \times 10^{12}$ cells/ m^3 filter material) and the bottom ($6.2 \pm 3.7 \times 10^{12}$ cells/ m^3 filter material) layers (Figure 3).

The presence of AOA at densities as high as $8.5\text{-}16 \times 10^{10}$ cells/ m^3 is puzzling, since these densities are comparable with densities quantified by *amoA* targeted qPCR, which ranged from $1\text{-}70 \times 10^{10}$ cells/ m^3 filter material. To examine the possible involvement of AOA to nitrification in the experiments, we calculated their hypothetical cell specific NH_4^+ removal rates at reference loading, based on the observed removal rates (Figure S5 in SI). The removal rates are approximately 4 orders of magnitude higher than previous reports (Kim et al. 2012, Prosser and Nicol 2012, Santoro and Casciotti 2011), suggesting that AOA are not responsible for nitrification under the imposed experimental conditions. Nevertheless, AOA presence at densities comparable with the AOB suggests *in situ* physiological activity in the filter. The loading conditions applied during the column experiments were compared with the *in situ* filter loading conditions, based on the NH_4^+ depth profile in the filter. As also observed by Lee et al. (2014) and Tatari et al. (2013), the NH_4^+ concentration decreased steeply with depth, dropping below 0.03 mg/L $\text{NH}_4^+\text{-N}$ at 0.15 m depth during all 6 sampling campaigns (Figure S6 in SI). The *in situ* loadings at the top, middle and bottom filter layers, calculated from the observed NH_4^+ concentration (average from all sampling campaigns at that depth) and the flowrate, were 138, 3.5, and 0.3 g $\text{NH}_4^+\text{-N}/\text{m}^3/\text{d}$ at the top, middle, and bottom layers, respectively. The reference loading (35 $\text{NH}_4^+\text{-N}/\text{m}^3$ filter material/d) in the column assays was therefore lower than the average full-scale loading at the top layer, but much higher than the average *in situ* loadings in the middle and bottom layers.

Densities of AOB and AOA with depth were quantified at higher resolution and were related to the *in situ* NH_4^+ loadings at each depth (Figure 4). Densities illustrated in Figure 4 are based on *amoA* qPCR quantification, and thus AOB densities are lower than the 16S-rRNA based densities reported in Figure 3. AOA densities marginally increased with depth and decreasing loading, whereas AOB densities decreased up to 2 orders of magnitude with depth (Figure 4). AOA exceeded AOB at low loadings, and AOB exceeded AOA at high loadings (Figure 4). These observations agree with previous findings that AOA activity is relevant at low NH_4^+ concentrations due to their high affinity (Martens-Habbena et al. 2009, Martens-Habbena and Stahl 2011, Niu et al. 2013).

Ultimately, our results suggest that under the imposed experimental conditions, AOB govern NH_4^+ oxidation due to high NH_4^+ loadings. AOA are present and outnumber AOB in the middle and bottom filter layers, suggesting a key role of AOA for *in situ* NH_4^+ removal due to the low NH_4^+ loadings at these filter depths.

3.3. AOB cell specific NH_4^+ removal rates

Cell specific NH_4^+ removal rates were calculated at each loading by normalizing the observed NH_4^+ removal rates for the volumetric AOB densities, which were experimentally determined (day 1) or

calculated (for the up-shifts). The cell specific NH_4^+ removal rates are within the high end of previously reported estimates that range from 0.30 to 11×10^3 fg $\text{NH}_4^+\text{-N}/\text{cell}/\text{d}$ (de Vet et al. 2011, Lee et al. 2014, Prosser 1989) (Figure 5). At reference loading, the cell specific NH_4^+ removal rate in the top layer column was lower than the other columns due to the higher AOB density in this layer. With loading increase, the cell specific NH_4^+ removal rates increased 6.5 and 2.8 times in the top and middle layer column, but no increase was observed in the bottom layer column (Table 1), indicating that cells in the bottom layer had already reached their maximum NH_4^+ removal rate of 3.4×10^3 (± 0.95) fg $\text{NH}_4^+\text{-N}/\text{cell}/\text{d}$ (average from all loadings).

Table 1. Overview of NH_4^+ removal rates (volumetric and cell specific) at reference loading and loading up-shifts for the three investigated depth layers.

	Top	Middle	Bottom
Observed removal rates (g $\text{NH}_4^+\text{-N}/\text{m}^3$ filter material/d)			
Reference loading (35 g $\text{NH}_4^+\text{-N}/\text{m}^3$ filter material/d)	34 ($\pm 0.87^\dagger$)	12 (± 2.2)	21 (± 3.8)
Up-shift to 88 g $\text{NH}_4^+\text{-N}/\text{m}^3$ filter material/d	88 (± 0.54)	11 (± 2.6)	24 (± 1.8)
Up-shift to 175 g $\text{NH}_4^+\text{-N}/\text{m}^3$ filter material/d	167 (± 3.2)	29 (± 11)	46 (± 22)
Up-shift to 350 g $\text{NH}_4^+\text{-N}/\text{m}^3$ filter material/d	223 (± 6.6)	42 (± 19)	20 (± 35)
Ratio of removal rate at 350 to removal rate at 35 g $\text{NH}_4^+\text{-N}/\text{m}^3$ filter material/d loadings	6.6	3.6	1
NH_4^+ removal capacity (g $\text{NH}_4^+\text{-N}/\text{m}^3$ filter material/d)	>223	42.0	27.5 [‡]
Highest observed cell specific NH_4^+ removal rate for AOB ($\times 10^3$ fg $\text{NH}_4^+\text{-N}/\text{cell}/\text{d}$)	7.1	14.2	2.6
Ratio of AOB cell specific removal rate at 350 to removal at 35 g $\text{NH}_4^+\text{-N}/\text{m}^3$ filter material/d loadings	6.5	2.8	1

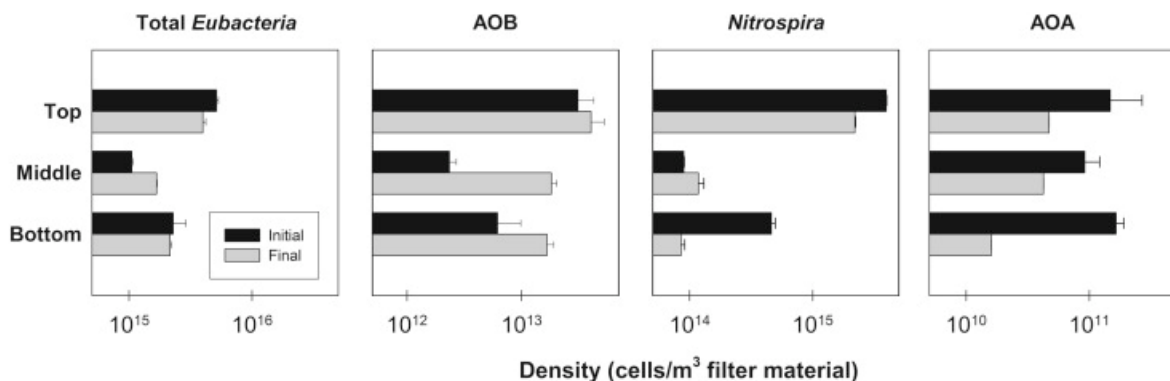


Figure 3. Microbial densities measured by qPCR (16S-rRNA based for Total *Eubacteria*, AOB and *Nitrospira*, *amoA* based for AOA) before and after the 9 days of column experiments. Error bars represent the standard deviation of triplicate DNA extractions and qPCR measurements.

This maximum cell specific NH_4^+ removal rate was lower than the rates observed in the top and middle layer columns. Additionally, the maximum cell specific NH_4^+ removal rate in the top layer column was not reached within the investigated loading range, and is higher than the values reported in Figure 5. Ultimately, the three layers exhibited different maximum cell specific NH_4^+ removal rates, suggesting that AOB with different physiological behaviors were active at the different layers.

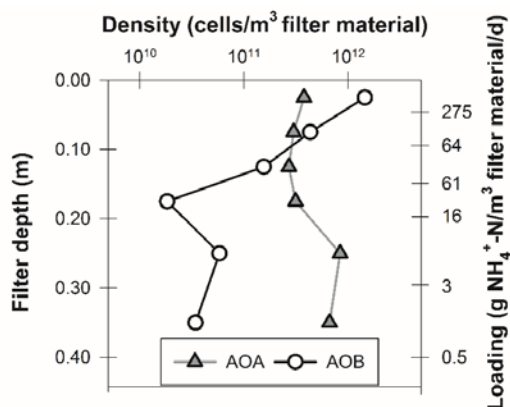


Figure 4. Distribution of AOB and AOA densities (*amoA* based qPCR) and *in situ* filter loading with depth in the full-scale filter.

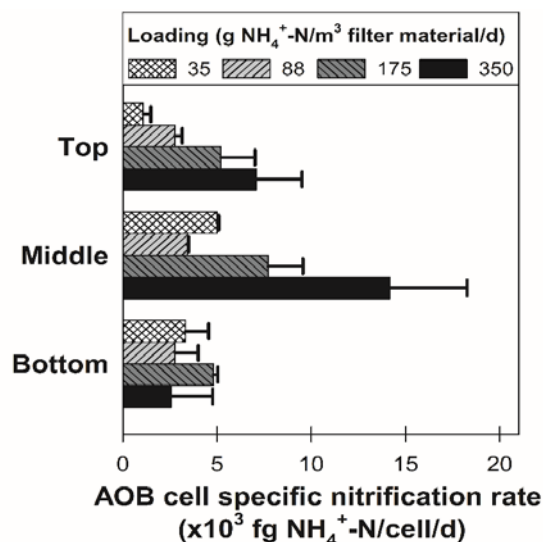


Figure 5. Cell specific NH_4^+ removal rates, calculated by normalizing the observed NH_4^+ removal rates for the AOB density at the time that each loading was applied in the columns. Error bars illustrate the error calculated by propagation of the standard deviations of the removal rate and the AOB density.

3.4. Model-based kinetic interpretation

NH_4^+ removal during the loading up-shifts was described by a 1-D biofilm model to estimate the best fit kinetic parameters K_S and q_{\max} for each column. Estimated K_S values for NH_4^+ ranged from 0.22 for the top to 0.13 mg NH_4^+ -N/L for the bottom layer column (Table 2). The estimated K_S values compared with the measured effluent NH_4^+ concentrations indicate a shift of removal kinetics from first order in the top to zero order in the bottom layer column. Identifiability of K_S was poor for the middle layer column; K_S was therefore set to 0.22 mg NH_4^+ -N/L, as for the top layer, to allow q_{\max} estimation. The chosen K_S value was justified by the zero order behavior observed from the middle layer, and only slight differences in the residual errors and the estimated values of the other parameters were observed by setting the K_S value to 0.13 mg NH_4^+ -N/L, as estimated in the bottom layer (data not shown). Estimated q_{\max} values for the loading upshifts ranged from 0.32 g NH_4^+ -N/g COD/d for the middle to 0.07 g NH_4^+ -N/g COD/d for the bottom layer column. Residual errors for the targeted variables (effluent NH_4^+ and NO_2^- concentrations) are reported in the SI (Table S2). Estimation of different q_{\max} for the three layers is consistent with the activity observations and confirms the variation of the intrinsic NH_4^+ removal capacity with depth.

Reference loading performances for the top and middle layer columns were overestimated by applying the parameters estimated at loading up-shifts: predicted NH_4^+ effluent concentrations from the top layer column were roughly 0.004 (± 0.0004) mg NH_4^+ -N/L, whereas the observed concentration ranged from 0.01 to 0.03 mg NH_4^+ -N/L (Figure 1A). Similarly, the observed NH_4^+

effluent concentrations exceeded the model predicted concentrations from the middle layer column, falling outside the model uncertainty interval (Figure 1B). Hence, new q_{\max} values were estimated by fitting the NH_4^+ and NO_2^- profiles at reference loading and the final AOB density at the end of the experiments, with K_S values and all other parameters kept constant during model re-calibration. Newly estimated q_{\max} values at reference loading were lower for both columns, decreasing by factor 4.8 for the top and 3.6 for the middle layer columns (Table 2). This performance overestimation was not observed in the bottom layer column, where kinetic parameters estimated at the loading up-shifts adequately described the effluent NH_4^+ concentration at reference loading (Figure 1C). Increase of the q_{\max} value with loading suggests that active nitrifiers in the top and middle layers have the capacity to immediately up-shift their removal rates according to the substrate loadings. To the authors' knowledge, such up-shift capacity has not previously been reported; neither was it observed in the bottom layer column where active nitrifiers had already reached their capacity at the lowest loading up-shift.

Table 2. Model estimated kinetic parameters for the three columns at reference loading. Values are reported with the standard error of the parameter estimates.

	NH_4^+ half saturation concentration, K_S (mg NH_4^+ -N/L)	Maximum specific NH_4^+ utilization rate at loading up-shifts, q_{\max} (g COD/g NH_4^+ -N/d)	Maximum specific NH_4^+ utilization rate at reference loading, q_{\max} (g COD/g NH_4^+ -N/d)	q_{\max} up-shift factor[‡]
Top	0.22 ±0.09	0.29 ± 0.05	0.06 ±0.03	4.8
Middle	0.22 [†]	0.32 ±0.02	0.09 ±0.01	3.6
Bottom	0.13 ±0.09	0.07 ±0.01	0.07 ±0.01	1

[†] Numerical estimation did not converge to a unique K_S value; the top layer column K_S value was used to estimate the remaining parameters.

[‡] Calculated as q_{\max} , at loading up-shifts/ q_{\max} at reference loading

3.5. Predicting full-scale filter capacity based on the depth-specific NH_4^+ removal capacity measurements

Using the depth specific NH_4^+ removal capacity, we predicted the overall filter response to transient loading changes. All filter layers displayed an NH_4^+ removal capacity that exceeded the *in situ* loading, although this additional capacity was clearly stratified and concentrated at the top. This stratification was a result of both AOB density and biokinetic behavior differences, with the top and middle layers exhibiting an up-shift capacity at high loadings not exhibited from the bottom layer.

Based on the estimated NH_4^+ removal capacities, we predicted the overall filter response to instantaneous loading increases. Ultimately, our goal was to assess to what degree the filter loading can be increased without compromising the filter effluent quality. The loading was increased either by increasing the influent NH_4^+ concentration or the influent flowrate, and the required filter depth to maintain strict effluent guideline limits (0.05 mg NH_4^+ /L) was computed for both scenarios.

The model predicted filter response to loading increases was different for the two scenarios (Figure 6). At increasing influent concentration, we predict that the filter (assumed 0.5 m depth) can maintain effluent limits up to a 91 g NH_4^+ /m³ filter material/d loading (Figure 6), corresponding to an increase from 0.13 to 0.38 mg NH_4^+ -N/L in the influent concentration. Higher permissible

loading increases are predicted for the increased flowrate scenario, up to $116 \text{ g NH}_4^+/\text{m}^3$ filter material/d (Figure 6) corresponding to a flowrate increase from 1.73 to $6.4 \times 10^6 \text{ L/d}$. Predicted effluent concentrations for both scenarios are illustrated in SI (Figure S7). Filter responses are clearly more robust to flowrate increase than to influent concentration increase at equivalent volumetric loadings. Yet, removal from each filter section is only dependent on loading; differences in the depth needed to remove equivalent loadings are the result of higher degrees of dilution at higher flowrates (eq.S9-S11 in SI). Overall the predicted NH_4^+ removal capacity is significantly higher (ca. 28%) than the safe operating loading window predicted by Lee et al. 2014, who estimated a maximum allowable loading of $75 \text{ g NH}_4^+/\text{m}^3$ filter material/d for the scenario of increased influent concentration. This difference may reflect a slightly different evolution of the NH_4^+ removal during the pilot-scale filter start-up and stabilization, possibly due to different flow distributions and backwashing hydraulics in the two systems.

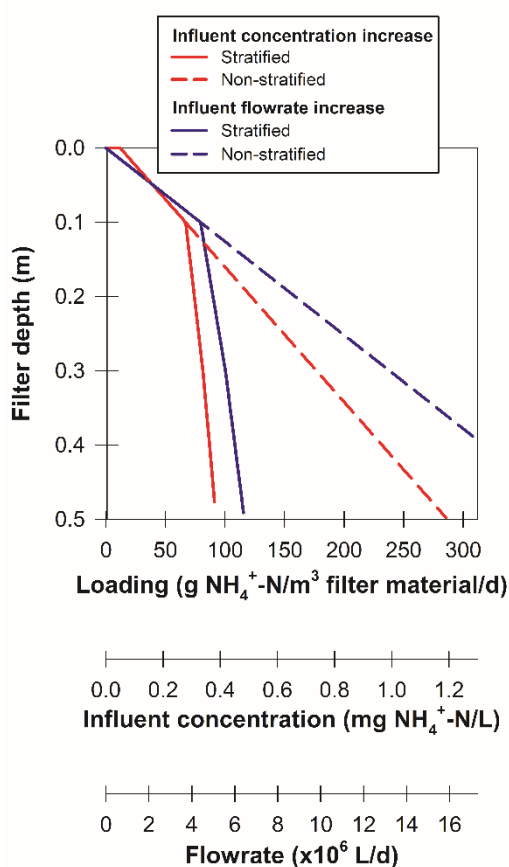


Figure 6. Model predicted depth needed in the full-scale filter to remove instantaneous loading up-shifts and retaining compliance with strict effluent guideline limits (0.05 mg/L NH_4^+). Loading was increased by increasing the influent concentration (from 0 to $1.3 \text{ mg NH}_4^+/\text{N/L}$ keeping the flowrate constant at $1.73 \times 10^6 \text{ L/d}$), or increasing the flowrate (from 0 to $17.3 \times 10^6 \text{ L/d}$ keeping the influent concentration constant at $0.13 \text{ mg NH}_4^+/\text{N/L}$).

Filter performance in both scenarios is overestimated if the filter is assumed non-stratified and biokinetics observed for the top layer are considered (Figure 6). The different filter layers behave differently because they are exposed to different *in situ* loadings and have different NH_4^+ removal capacities. The deeper filter layers have higher NH_4^+ removal capacities than the ones exhibited at the *in situ* loadings, and these additional capacities can be accessed when NH_4^+ penetrates deeper in the filter. NH_4^+ removal capacity of these deeper layers remains however lower than for the top layers. Therefore, it is essential to examine and account for possible stratification in a filter, to avoid low filter efficiency or NH_4^+ breakthrough in the effluent, when loading up-shifts are considered.

3.6. Conclusions

This study investigated the NH_4^+ removal capacity at different depths of a full-scale rapid sand filter. Filter material collected from selected depths was investigated in a lab-scale continuous-flow assay under conditions that maintained the antecedent hydrodynamic properties in order to reflect behavior of the full-scale filter.

The main findings were:

- Strong stratification of NH_4^+ removal rates and NH_4^+ removal capacity were consistent with the decrease of AOB density with depth.
- AOB density increased consistently with the increase of NH_4^+ removal rate over time, suggesting the primary involvement of AOB to nitrification under the experimental conditions. No increase in the density of total *Eubacteria*,

Nitrospira or AOA was observed.

- The maximum AOB cell specific NH_4^+ removal rate in the bottom layer was lower than in the top and middle layers, suggesting that nitrifiers with different metabolic physiologies or different microbial types were active at the different depths.
- The NH_4^+ removal biokinetic behavior was stratified in the filter and different up-shift capacities were observed with depth. In the top and middle layers, an instantaneous activity up-shift was observed with loading increase, while this was absent for the bottom layer.
- The predicted maximum filter capacity that would retain compliance with the effluent NH_4^+ guideline limits was approximately 4 times higher than the reference operating NH_4^+ loading.

Acknowledgements

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Supplementary Information

Depth investigation of rapid sand filters for drinking water production reveals strong stratification in nitrification biokinetic behavior

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I. Experimental set-up and design of the offline NH_4^+ removal investigations

The experimental set-up is illustrated in Figure S1.

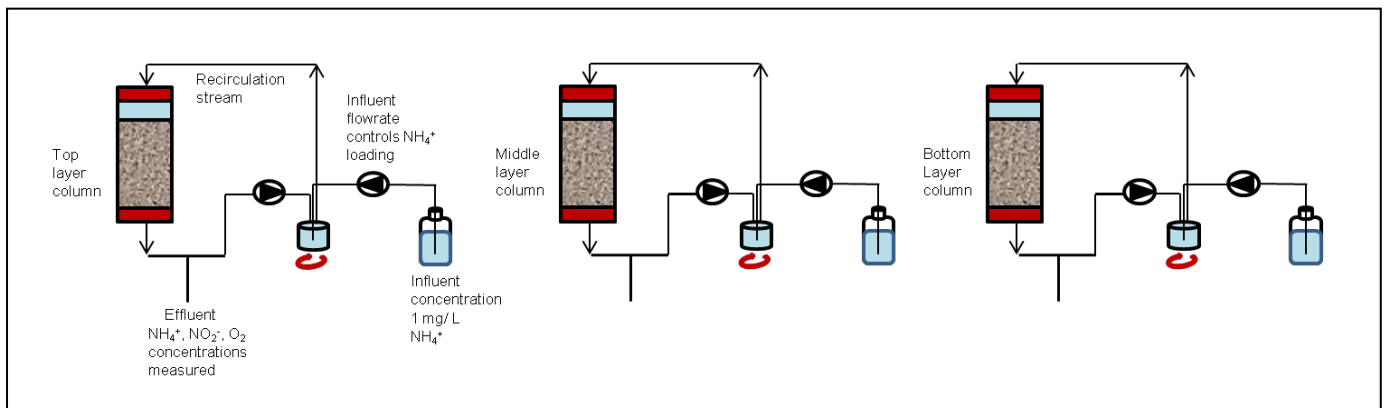


Figure S1. Experimental set-up used for the offline activity experiments. Three columns were operated in parallel, each packed with filter material from the top, middle or bottom layer.

The columns during the offline activity investigations were started-up at reference loading, that equaled $35 \text{ g NH}_4^+\text{-N/m}^3 \text{ filter material/d}$. At day 2 of operation, the loading in all three columns

was up-shifted to $350 \text{ g NH}_4^+\text{-N/m}^3$, by increasing the influent flowrate 10 times, while keeping constant the $\text{NH}_4^+\text{-N}$ influent concentration at 1 mg/L . The up-shifted loading was fed for 3 hours, after which the loading was down-shifted to the reference until the next day. On day 3 of operation, the loading in all columns was up-shifted to $88 \text{ g NH}_4^+\text{-N/m}^3$, similarly by regulating the influent flowrate and keeping the NH_4^+ influent concentration constant. This up-shift lasted 5 hours, after which the loading was downshifted to the reference. On day 4 of operation, the loading was up-shifted to $175 \text{ g NH}_4^+\text{-N/m}^3$ for 4 hours, before decreasing the loading back to the reference for the next 5 days. The loading applied to the columns is schematically illustrated in Figure S2. Duration of each loading shift was determined by salt tracer tests, so that both hydraulic and effluent pseudo steady states at the transient conditions could be reached (Tatari et al. 2013).

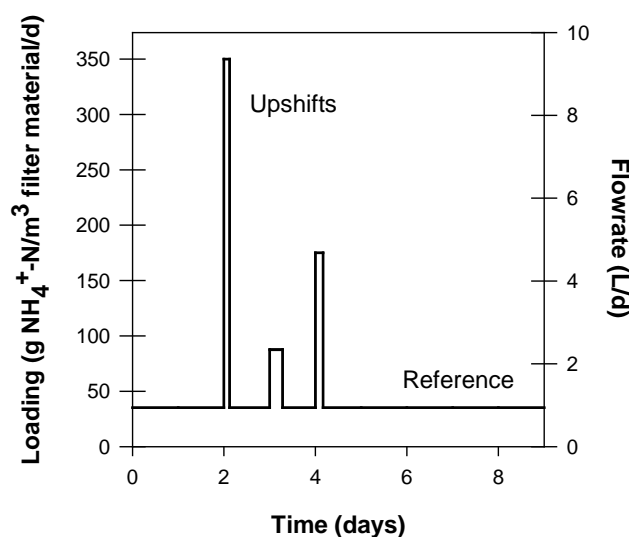


Figure S2. Schematic diagram of the loading applied to each column. Influent NH_4^+ concentration was constant at $1 \text{ mg NH}_4^+\text{-N/L}$ and the loading was controlled by adjusting the flowrate.

II. Calculation of biofilm densities for the model

To interpret the behavior of the three columns kinetically, a 1-D biofilm model was implemented in Aquasim (Reichert 1998). Each column was modeled independently as a 2 equal volume compartments in series with a total volume of $8.8 \times 10^{-2} \text{ L}$ to match the hydrodynamic behavior characterized by tracer tests (Tatari et al. 2013). The biofilm on the surface of the sand grains was considered homogeneous and continuous.

The initial biofilm thickness (L_F) was estimated by confocal laser scanning microscopy (Leica TCS SP5) on sectioned and stained (Syto-9, Invitrogen) grains (Gülay et al. 2014) (Table S1). A known mass of drained-wet filter material was dried at 60°C overnight and weighed again to calculate the dry to drained-wet mass conversion factor. The number of grains per mass was manually counted in triplicate samples of 7-16 g dry mass and was converted in grains/draind-wet mass by multiplying with the conversion factor at each depth (Table S1). The particle size distribution was determined by sieving (Vibratory Sieve Shaker AS 200 control, Retsch) of the dry filter material through 2.00-0.063 mm mesh sized sieves. The median diameter at each depth was determined from the cumulative distribution of the mass fractions retained on each sieve (Table S1).

Table S1. Experimentally determined physical properties of the filter material at the three investigated depths

	Top	Middle	Bottom
Median grain radius, r_M (m) [†]	7×10^{-4}	4×10^{-4}	5×10^{-4}
Drained-wet packed bulk density, ρ (g/m³ filter material) [‡]	1.1×10^6	1.6×10^6	1.7×10^6
Number of grains per filter material dry mass, N (grains/g) [*]	198	385	382
Initial biofilm thickness, L_F ($\times 10^{-5}$)	5.5	1	1

[†] Particle size distribution determined by sieving of the dry filter material.

[‡] Calculated from the filter material drained-wet mass and volume in a graduated cylinder.

^{*} Manually counted in triplicate.

The total biofilm surface area in each column A_F (m²) was expressed as function of the distance from the biofilm base z (m) by (Reichert 1998):

$$A_F = 4 \pi (r_G + z)^2 N \rho V \quad (\text{eq. S1})$$

where: r_G is the radius of the filter material grain without the biofilm (m), N is the number of grains per drained-wet filter material mass (grains/g), ρ is the drained-wet packed bulk density (g/m³ filter material) and V is the packed filter material volume in the column (m³ filter material). r_G was calculated by:

$$r_G = r_M - L_F \quad (\text{eq. S2})$$

where: r_M is the median grain radius estimated from the particle size distribution (m).

The initial volumetric AOB density $X_{AOB,V}$ (cells/m³ filter material) determined by molecular tools was converted into COD units for consistency with most modeling studies as follows (IWA task group on biofilm modeling 2006):

$$X_{AOB,V}^{\text{COD}} = 1.42 \times 10^{-12} X_{AOB,V} \quad (\text{eq. S3})$$

where: $X_{AOB,V}^{\text{COD}}$ is the volumetric AOB density in COD units (g COD/m³ filter material).

The biofilm volume on a filter material grain was calculated by the volume of a spherical shell of thickness L_F on the outer surface of a sphere of radius r_G . The total biofilm volume V_F (m³ biofilm) in a column was:

$$V_F = \frac{4}{3} \pi ((r_M)^3 - (r_G)^3) N \rho V \quad (\text{eq. S4})$$

The initial AOB biofilm density $X_{AOB,F}^{\text{COD}}$ in each column (g COD/m³ biofilm) was calculated as (Wanner and Reichert 1996):

$$X_{AOB,F}^{\text{COD}} = \frac{X_{AOB,V}^{\text{COD}} V}{V_F} \quad (\text{eq. S5})$$

III. Parameter estimation algorithm

The parameter estimation algorithm minimized the sum of squares of the weighted deviations between the observed and model predicted values $\chi^2(p)$ for each targeted variable p (effluent NH_4^+ and NO_2^- concentration) calculated by ¹:

$$\chi^2(p) = \sum_{i=1}^n \left(\frac{y_{\text{meas},i} - y_i(p)}{\sigma_{\text{meas},i}} \right)^2 \quad (\text{eq. S6})$$

where: $y_{\text{meas},i}$ is the i -th experimentally observed value, $y_i(p)$ is the model predicted value corresponding to the i -th measurement of the variable p at the time and the location of this measurement, $\sigma_{\text{meas},i}$ is the standard deviation of the experimental values and n is the number of data points. $\sigma_{\text{meas},i}$ was determined from triplicate measurements of the same sample and was 0.023 mg N/L for NH_4^+ and 0.002 mg N/L for NO_2^- . Three simulations, one for each loading up-shift were created for each column and were run simultaneously, so that parameter estimation fitted the effluent NH_4^+ and NO_2^- concentrations at these three conditions. Re-calibration of the top and middle layers at reference loading targeted both the NH_4^+ and NO_2^- effluent concentrations and the final AOB density. The final AOB biofilm density in COD units was calculated from the experimentally determined volumetric densities as above. $\sigma_{\text{meas},i}$ for the final AOB density was calculated from the triplicate qPCR extractions and quantifications of the same sample at each depth. Error contribution of each targeted variable in all parameter estimations is reported in Table S2.

Table S2. Error contribution (χ^2) of each targeted variable during model calibration at loading up-shifts and during model re-calibration of the top and middle layers at reference loading.

	Top			
	35 (Reference)	88	175	350
Loading (g NH_4^+-N/m³ filter material/d)				
Effluent NH_4^+-N concentration	11.8	1.50	39.0	93.0
Effluent NO_2^--N concentration	487	526	46.2	33.2
Final AOB density	0.26	-	-	-
Total χ^2	499 [†]	-	-	738 [‡]
	Middle			
	35 (Reference)	88	175	350
Loading (g NH_4^+-N/m³ filter material/d)				
Effluent NH_4^+-N	95.8	45.1	90.5	310
Effluent NO_2^--N	204	30.3	70.3	291
Final AOB density	14.3	-	-	-
Total χ^2	314 [†]	-	-	837 [‡]

Loading (g NH ₄ ⁺ -N/m ³ filter material/d)	Bottom			
	35 (Reference)	88	175	350
Effluent NH ₄ ⁺ -N	-	275	900	421
Effluent NO ₂ ⁻ -N	-	205	100	258
Final AOB density	-	-	-	-
Total χ^2	-	-	-	2159 [‡]

[†] Total

χ^2 error contribution from all three targeted variables (effluent NH₄⁺ and NO₂⁻ effluent concentrations and final AOB density) at reference loading.

[‡] Total χ^2 error contribution of the two targeted variables (effluent NH₄⁺ and NO₂⁻ effluent concentrations) at all three loading up-shifts.

IV. Sensitivity analysis

Local sensitivity analysis was conducted at reference loading to evaluate model sensitivity to eventual changes or uncertainty in the biofilm properties and kinetic parameters. The local sensitivity analysis extrapolated linearly the effect on model output due to change in one parameter each time within a defined range of uncertainty². An absolute-relative sensitivity measure δ was calculated by¹:

$$\delta = p \frac{\partial y}{\partial p} \quad (\text{eq. S7})$$

where: y is the output variable examined for sensitivity (in this case the effluent NH₄⁺ or NO₂⁻ concentrations) and p is the input variable that is allowed to vary each time. The δ sensitivity measure indicates the absolute change in the y output variable value caused by a 100% change in the p input variable value. Large δ means that the parameter p has a substantial effect on the considered model output. Positive δ values indicate a positive effect on the y output variable value, whereas negative δ values indicate a negative effect on the y output variable value.

All model parameters were included in the sensitivity analysis, and were ranked according to their effect on the predicted effluent NH₄⁺ or NO₂⁻ concentration. Figures S3 and S4 illustrate the first 18 parameters in the ranking i.e. the ones whose change had the greatest impact on the model output.

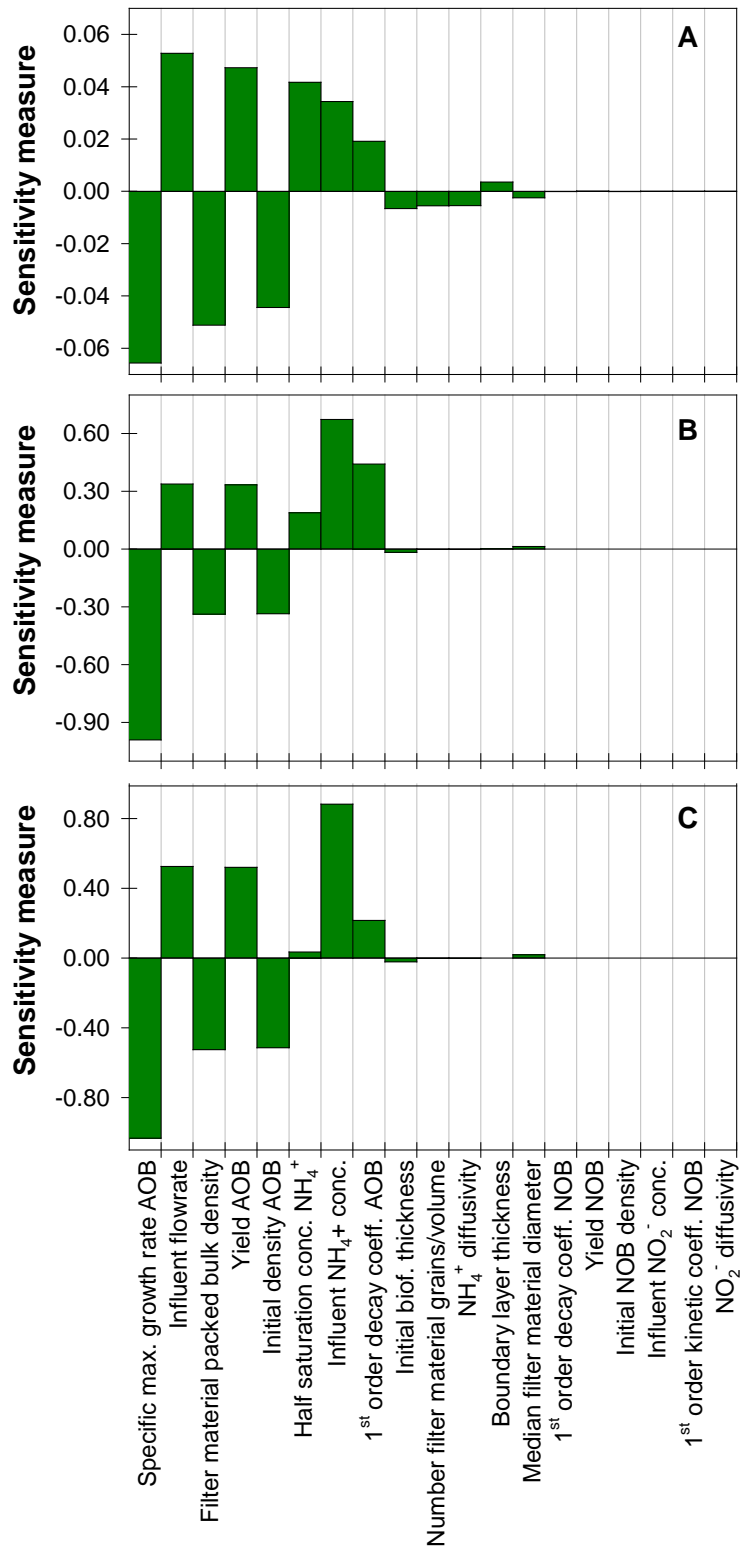


Figure S3. Sensitivity measure of input model parameters on the predicted effluent NH_4^+ concentration at reference loading ($35 \text{ g NH}_4^+\text{-N/m}^3$ filter material/d) in top (A), middle (B) and bottom (C) layer columns.

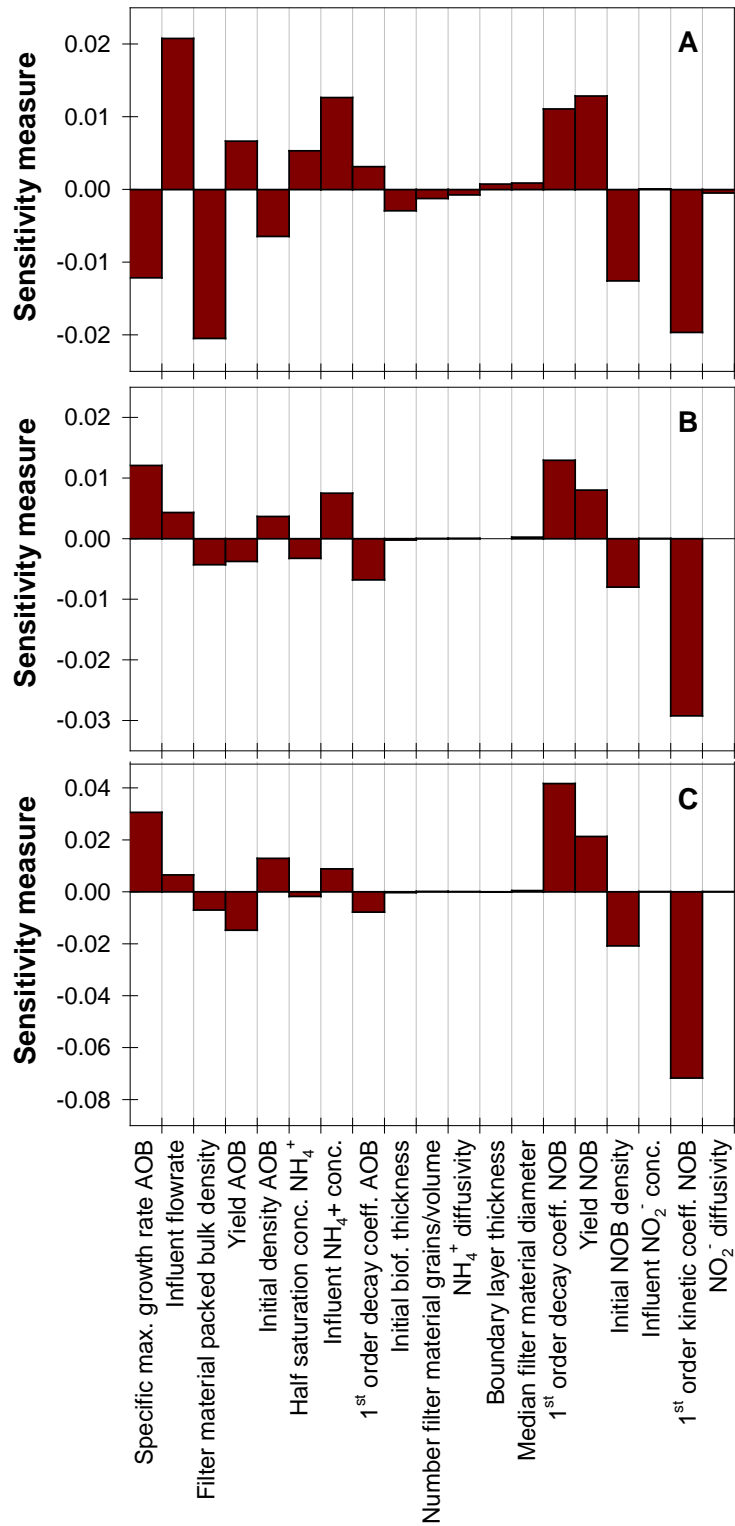


Figure S4. Sensitivity measure of input model parameters on the predicted effluent NO_2^- concentration at reference loading ($35 \text{ g NH}_4^+ \text{-N/m}^3 \text{ filter material/d}$) in top (A), middle (B) and bottom (C) layer columns.

V. Evaluation of mass transfer limitations

To exclude that NH_4^+ removal in the columns was limited by external mass transfer, we calculated the relative ratio K^* of the NH_4^+ external transport and the maximum NH_4^+ utilization and diffusion rate in the biofilm (Rittmann and McCarty 2001):

$$\text{VI. } K^* = \frac{D_W}{L_L} \left[\frac{K_S}{q_{\max} X_{\text{AOB,F}}^{\text{COD}} D_F} \right]^{1/2} \quad (\text{eq. 8})$$

where: D_W is the NH_4^+ diffusivity in water at 10°C (m^2/d), L_L is the calculated external boundary layer thickness (m), $X_{\text{AOB,F}}^{\text{COD}}$ is the AOB density expressed in COD units per unit biofilm volume as calculated in eq. S1-S5 in SI, and D_F is the NH_4^+ diffusivity in the biofilm (m^2/d). K^* values below 1 indicate slow external transport and therefore potential limitation by mass transfer. Calculated K^* values for the three columns ranged at reference loading from 11 to 28 and at loading up-shifts from 10 to 15, hence external mass transfer was not rate limiting. This conclusion was consistent with the model predicted NH_4^+ and NO_2^- spatial concentration profiles at selected time points, which showed a maximum of 14% decrease between the NH_4^+ concentration in the bulk and at the biofilm base (data not shown). This indicates that internal biofilm diffusion was also not limiting NH_4^+ removal. This conclusion based on kinetic analysis is consistent with the experimental observations from Lee et al. (2014), who observed that the pilot filter removal was only determined by the loading and not the flowrate. Other studies however support that mass transfer can limit removal in these filters (Lopato et al. 2013, Stembal et al. 2005, van den Akker et al. 2008)

VI. Calculated cell specific NH_4^+ removal rates for the examined microbial types

To examine whether other microbial types than AOB, and particularly AOA could be the dominant NH_4^+ oxidizers, we calculated what their cell specific NH_4^+ removal rates would be at reference loading. Since the densities of these microbial types did not change consistently during the column experiments, the observed NH_4^+ removal rates were normalized with the average of the initial and final microbial densities. Resulting cell specific NH_4^+ removal rates are illustrated in Figure S5.

The calculated cell specific NH_4^+ removal rates for AOA ranged within $1.3\text{-}17.4 \times 10^5$ fg $\text{NH}_4^+\text{-N}/\text{cell}/\text{d}$ (Figure S4), which is roughly 4 orders of magnitude higher than most reported literature values ranging within $28\text{-}192$ fg $\text{NH}_4^+\text{-N}/\text{cell}/\text{d}$ (Kim et al. 2012, Prosser and Nicol 2012, Santoro and Casciotti 2011), ultimately suggesting that AOA should not be the predominant active nitrifiers at the given conditions. However, Kasuga et al. 2010 reports extremely high AOA cell specific NH_4^+ removal rates ranging within $1.4\text{-}24 \times 10^4$ fg $\text{NH}_4^+\text{-N}/\text{cell}/\text{d}$ from an activated carbon filter where AOA appeared to be the predominant nitrifiers.

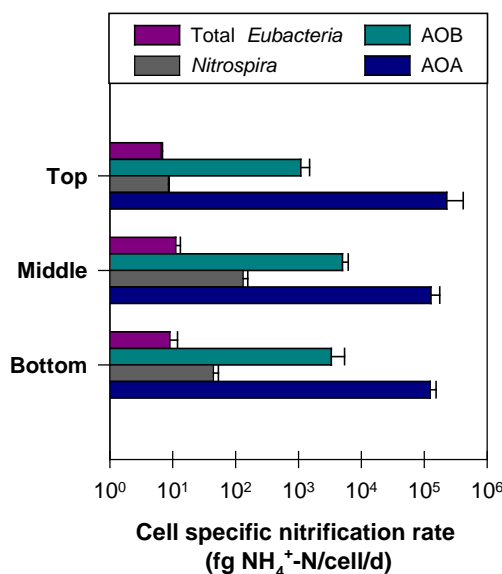


Figure S5. NH_4^+ removal rates at reference loading normalized by the number of cells for each investigated microbial type in the three columns. The average of the initial and final microbial densities was used for the normalization.

VII. Full scale filter depth profiles

The full scale filter NH_4^+ depth profile was obtained by sampling water in the 0.05-0.40 m interval during 6 sampling campaigns conducted over 2 filter-run cycles is illustrated in Figure S6.

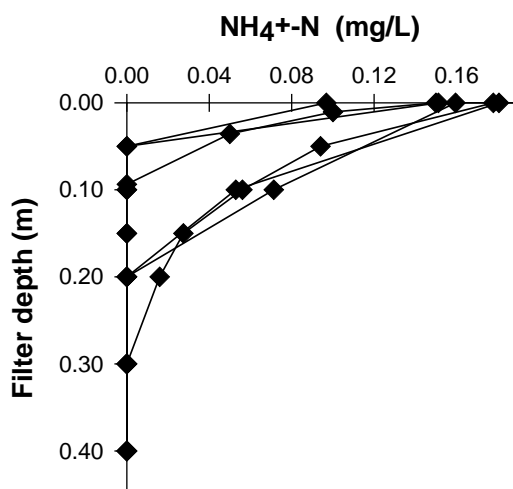


Figure S6. NH_4^+ depth profiles obtained over two filtering cycles in the investigated filter at Islevbro waterworks.

VIII. Predicting filter performance based on depth-specific nitrification capacities

The measured depth-specific nitrification capacities were used to predict full-scale filter performance at increased loading rates. The depth needed to remove the influent loading to below guideline limits was calculated for a 10-fold loading increase, examined for two scenarios: concentration and flowrate increase. In both scenarios the filter was assumed to be 0.50 m deep, as the deepest 0.2 m of the filter were not investigated due to sampling limitations. The 0.5 m deep

filter bed was divided into three sections: top (0-0.10 m), middle (0.10-0.30 m) and bottom (0.30-0.50 m), to include all intermediate depths that were not investigated in the lab-scale assays. Maximum removal from these filter sections was considered equal to the experimentally estimated nitrification capacity of the respective layer. Based on mass balance the concentration in the top section was calculated by:

$$S_{\text{top},x} = \frac{S_{\text{in}}Q - 10^3 r_{\text{max,top}} x A_{\text{F}}}{Q} \quad (\text{eq. S8})$$

where: x is the vertical coordinate reflecting the depth within the top section ($0 \leq x \leq 0.1$ m), $S_{\text{top},x}$ is the NH_4^+ concentration at a depth x (mg N/L), S_{in} is the influent NH_4^+ concentration (mg N/L), Q is the flowrate (L/d), A_{F} is the cross sectional area of the filter (m^2), and $r_{\text{max,top}}$ is the nitrification capacity of the top filter layer ($\text{g NH}_4^+\text{-N}/\text{m}^3$ filter material/d). Similarly, the concentration in the middle section was calculated by:

$$S_{\text{mid},y} = \frac{S_{\text{in}}Q - 10^3 (r_{\text{max,top}} V_{\text{top},0.1} + r_{\text{max,mid}} y A_{\text{F}})}{Q} \quad (\text{eq. S9})$$

where: y is the vertical coordinate reflecting the depth within the middle section ($0 \leq y \leq 0.2$ m), $S_{\text{mid},y}$ is the NH_4^+ concentration at a coordinate y (mg N/L), equivalent to a $y+0.1$ m depth in the filter, $r_{\text{max,mid}}$ is the nitrification capacity of the middle layer ($\text{g NH}_4^+\text{-N}/\text{m}^3$ filter material/d), and $V_{\text{top},0.1}$ is the volume of the entire top section ($x=0.1$ m). Lastly, the concentration in the bottom section was calculated by:

$$S_{\text{bot},z} = \frac{S_{\text{in}}Q - 10^3 (r_{\text{max,top}} V_{\text{top},0.1} + r_{\text{max,mid}} V_{\text{mid},0.2} + r_{\text{max,bot}} z A_{\text{F}})}{Q} \quad (\text{eq. S10})$$

where: z is the vertical coordinate reflecting the depth within the bottom section ($0 \leq z \leq 0.2$ m), $S_{\text{bot},z}$ is the concentration at a coordinate z (mg N/L), equivalent to a $0.3+z$ m depth in the filter, $r_{\text{max,bot}}$ is the nitrification capacity of the bottom layer ($\text{g NH}_4^+\text{-N}/\text{m}^3$ filter material/d), and $V_{\text{mid},0.2}$ is the volume of the entire middle section ($y=0.2$ m).

For the increasing concentration scenario, S_{in} was given values from 0.13 to 1.3 mg $\text{NH}_4^+\text{-N}/\text{L}$ at a 0.001 mg/L step size, while Q was constant at 1.73×10^6 L/d. For the increasing flowrate scenario, Q was given values from 1.73×10^6 L/d to 17.3×10^6 L/d at a 24 L/d step size, while S_{in} remained constant at 0.13 mg $\text{NH}_4^+\text{-N}/\text{L}$. The calculation algorithms for both scenarios were implemented in MATLAB, and the minimum depth so that the NH_4^+ concentration is below strict guideline limits (0.05 mg NH_4/L) was identified.

The calculated effluent concentrations at both scenarios as well as the concentration at 0.1 and 0.3 m depth are illustrated in Figure S7.

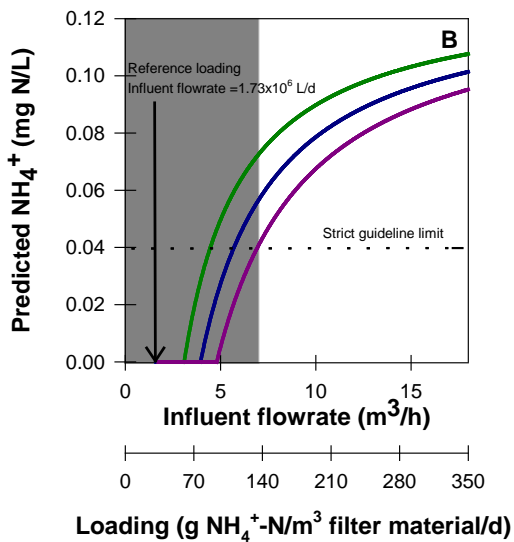
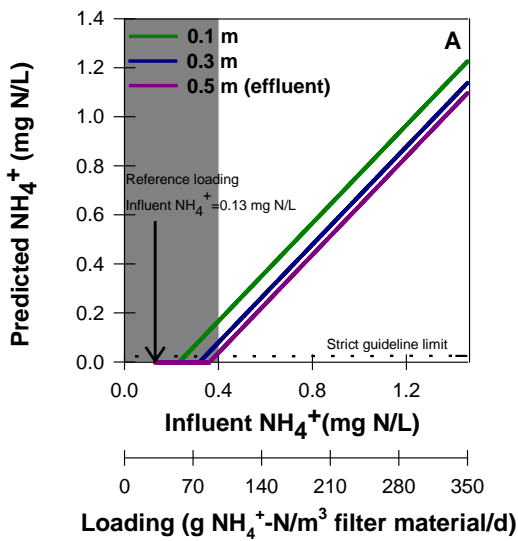


Figure S7. Predicted NH_4^+ effluent and concentration at selected filter depths for loadings ranging from the reference ($35 \text{ g NH}_4^+\text{-N/m}^3$ filter material/d) to $350 \text{ g NH}_4^+\text{-N/m}^3$ filter material/d. Loading was increased by: (A) increasing the influent concentration from 0.13 to 1.3 mg $\text{NH}_4^+\text{-N/L}$ keeping the flowrate constant at $1.73 \times 10^6 \text{ L/d}$, (B) increasing the flowrate from $1.73 \times 10^6 \text{ L/d}$ to $17.3 \times 10^6 \text{ L/d}$ keeping the influent concentration constant at 0.13 mg $\text{NH}_4^+\text{-N/L}$. The shaded areas correspond to the safe operational loading windows if a strict guideline limit of 0.05 mg NH_4^+/L

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