Vegetated greywater treatment walls: Design modifications for intermittent media filters

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VEGETATED GREYWATER TREATMENT WALLS:

Design Modifications for Intermittent Media Filters

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ABSTRACT

The incorporation of on-site wastewater treatment systems is one means to meet the infrastructure needs of the rapidly expanding urban population. With space at a premium, the existing technologies for on-site systems should be reexamined and adapted to fit the needs of the urban setting. This study examined the potential to combine the existing wastewater treatment technology of the intermittent media filter with the new architectural trend of green walls, creating a treatment system with minimal spatial footprint and with a built-in urban greening component.

A novel vegetated intermittent media filter wall was constructed in Ås, Norway and dosed with domestic greywater for a period of three months. Overall treatment performance and removal trends over the 175cm filter depth were monitored. Three separate wall sections were constructed to monitor the treatments effects of containing wall material choice and presence of vegetation. Despite a daily dosing rate of nearly 1000 l/m^2 the system achieved average reduction rates of over 95%, 80%, 90%, 30%, and 69% for BOD_5, COD, TSS, total nitrogen, and total phosphorus, respectively, as well as approximately two log unit reduction of bacteria indicator *E. coli*. Examination over the depth of the system showed that most organic (COD) and solids removal takes place in the upper 15cm, but with a sudden increase in loading a greater removal was seen at lower depths. With regard to nitrification, increased nitrate levels did not appear before 100cm filter depth, likely suppressed by high organic loading at the surface. The findings suggest that the great filter height associated with the wall design was useful for buffering sudden increases in hydraulic loading, as well as for facilitating nitrification under extreme loading conditions. Wall material with a more permeable construction (geotextile grid) preformed slightly better in the treatment of organics than non-permeable wall material (plastic liner), but confounding variables reduce the confidence in this finding. The vegetated wall section showed the greatest removal rates in almost every parameter measured, especially removal of *E. coli*. A difference in hydraulic retention times as shown by (NaCl) tracer tests is the likely cause of this phenomenon, rather than the vegetation itself.

The significant reduction of constituents of concern using only a small spatial footprint make this system a worthy candidate for further research and development regarding urban wastewater treatment applications.

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LIST OF ACRONYMS AND ABBREVIATIONS

USEPA	United States Environmental Protection Agency
BOD_5	5-day Biochemical Oxygen Demand
BOD_7	7-day Biochemical Oxygen Demand
COD	Chemical Oxygen Demand
WHO	World Health Organization
E. coli	*Escherichia coli*
DALY	Disability Adjusted Life Year
TSS	Total Suspended Solids
N	Nitrogen
P	Phosphorus
LECA	Lightweight Expanded Clay Aggregates
PE	Polyethylene
PP	Polypropylene
NaCl	Sodium Chloride
ANOVA	Analysis of Variance

1. INTRODUCTION

Half of the global population currently lives in cities and this is estimated to grow to sixty percent within two decades (UNW-DPAC 2010). The demographic shift towards a greater number and density of cities poses concerns regarding the needs of human inhabitants and the integrity of the surrounding natural environment. Serving additional urban populations with basic infrastructure requirements, particularly water supply and wastewater management, is one of the greatest challenges faced by society in the coming years. Yet as the built environment expands to accommodate urban needs, the replacement of vegetated land by impermeable surfaces leads to additional concerns regarding polluted stormwater, increased city temperatures, reduced air quality, and loss of species diversity (Hopkins and Goodwin 2011).

Among the proposed solutions for infrastructure and environmental concerns is the *decentralization* of urban services. A decentralized approach for collecting and treating wastewater has been suggested by some experts as a method for incorporating additional human populations into growing cities (Gikas and Tchobanoglous 2009). A distributed approach is also gaining support regarding provision of green spaces in the urban landscape. Breaking up large areas of impermeable surfaces with the incorporation of plant life is shown to improve both air and stormwater quality, reduce the urban heat island effect, and at the same time improve the aesthetics and human comfort levels in a city environment (Hopkins and Goodwin 2011; Dunnett and Kingsbury 2008; Nowak et al. 1998; Currie and Bass 2008). This concept of distributing green spaces has evolved to the extent that covering the actual building envelope with plants, in the form of green roofs and green walls, is becoming a common architectural practice (Hopkins and Goodwin 2011; Dunnett and Kingsbury 2008). Green roofs and walls provide all of the benefits of traditional green spaces, in addition to noise buffering (Hopkis and Goodwin 2001) and better insulation for the buildings themselves (Castleton et al. 2010). There is even the possibility to cultivate edible plant species, expanding the available space for urban agriculture.

Despite the potential benefits, green wall installations in particular have been criticized for their questionable environmental sustainability (Gandy 2010). Most systems constructed today are commissioned by commercial retailers as an aesthetic or architectural element to draw the interest of the public. Maintaining an assemblage of plants against a large vertical surface typically requires an automated irrigation system, which demands the input of water, fertilizer, and energy. To overcome these drawbacks and introduce an additional benefit outside of the aesthetic realm, this study proposes to combine the green wall concept with decentralized wastewater treatment technology, creating a multifunctional space that fulfills multiple infrastructure requirements of the growing urban population.

1.1 Decentralization of wastewater treatment in the urban environment

In many parts of the world the standard model for urban wastewater management is *centralized* treatment. Wastewater is collected and transported from homes, businesses, and industry via large underground sewers to a shared facility for treatment and disposal. A major benefit to this scheme is central control, which allows technical expertise and expensive technology to be focused into one treatment facility and also enables quality monitoring of the effluent released into the environment.

In recent years a faction of wastewater treatment experts has questioned the expansion of the centralized wastewater management model due to inherent weaknesses. Much criticism is directed at the sewer transport system, which is difficult and expensive to maintain and pollutes large amounts of relatively clean water in the transport process (Reijnders 2001; Heip et al. 2001). In Oslo, Norway it is estimated that sixty percent of the water treated at wastewater treatment facilities is actually derived from leaking drinking water pipes, rainwater, and groundwater entering the sewer network (Oslo Kommune 2000). Additional criticism is aimed at the centralized urban wastewater treatment scheme due to its focus on waste elimination versus resource recovery. The system collects water from a wide range of sources and mixes these chemically dissimilar wastewater streams, which complicates purification and potential reuse applications (Wilderer 2001). As a remote and out of sight process, there is little or no incentive for the user to conserve the quality or quantity of wastewater sent to the treatment facility.

The alternative solution is to collect and treat wastewater from single residences or small clusters of buildings using *decentralized* or *on-site* wastewater treatment systems. Treating wastewater close to its point of origin evades the problems associated with sewer transport systems and the mixing of waste streams, offers opportunities for local reuse of resources, and fits in line with a growing trend for environmental accountability regarding the discharge of waste (WHO 2006). Decentralized wastewater treatment has historically been seen as an option for rural developments where connection to the centralized sewer network is impossible or impractical, but more recently experts are suggesting the incorporation of decentralized systems into the urban environment as a method to ease the capacity problems faced by existing centralized wastewater infrastructure (Gikas and Tchobanoglous 2009).

Before widespread incorporation of decentralized wastewater management into the urban setting, research and development is needed to optimize treatment alternatives. Unlike the centralized model which benefits from the economies of scale in both cost and management, the decentralized treatment scheme is accused of being difficult to supervise and control, leading to improperly functioning treatment systems producing low effluent quality (Wilderer 2001). In terms of overall sustainability, the sum of the materials and energy used in the various decentralized systems should not surpass the resources necessary to treat the equivalent wastewater at a single centralized facility (Reijnders 2001). In order for an on-site or decentralized wastewater treatment scheme to offer a realistic alternative to the traditional centralized scheme in a more urbanized environment, the system must be: cost effective, require minimal expertise or maintenance, reliably produce an effluent quality which meets regulation standards, and must also require minimal use of energy and materials. Additional advantage would come from a targeted reuse option built into the treatment system.

Based on these criteria, a promising category of treatment options is the biological filter system. Sometimes called nature based or land based systems, this loose categorization may cover a wide range of options, from simple soil infiltration to sand/media filters or constructed wetlands. What these arrangements have in common is the use of a bed of porous media which support biological growth— the key to treatment of the wastewater as it passes through the system. In general the treatment units follow a septic tank and are operated at ambient temperatures, without addition of chemicals, and with a minimum of moving parts and specialized equipment. The major drawback to these systems is the large area requirement. For example, the suggested dimensioning in Norway for a constructed wetland system treating domestic greywater is 3 – 5 m^2 per person, and up to 8 – 10 m^2 per person for full strength domestic wastewater (Norsk Rørsenter 2001). Until recently this has excluded the use of these systems in the urban setting, but given their key advantages adapting the biological filter systems for the urban community is a worthwhile goal.

Researchers in Norway have designed systems aimed at reducing the space requirement for on-site wastewater treatment using biological filter methods. In Oslo, a treatment system was installed in 2000 in the courtyard of an apartment building which uses a combination of septic tank, single-pass vertical flow aerobic biofilter, and horizontal flow constructed wetland filter to treat the greywater produced in the building (Jenssen 2005). The system uses a spatial footprint of only 1m^2 per person, and initial reports showed superior effluent quality. Additional compact on-site treatment systems have been designed in Norway using the septic tank, aerobic biofilter, saturated filter treatment combination, treating even full-strength domestic wastewater (Heistad et al. 2006). While initial treatment shows excellent effluent quality, additional research is necessary to determine the lifetime of these filter systems with respect to hygienic barriers and phosphorus removal (Heistad et al. 2009).

In other parts of the world the on-site treatment systems using biological filtration methods are being pushed even further. Instead of designing compact systems which fit into the surrounding landscape, researchers are designing compact systems which fit onto the building itself. Separate projects in both Spain and the United Kingdom have developed plans for compact constructed wetland/reed-bed systems which can be installed on flat rooftops to treat and recycle the greywater from a building (Gomez-Gonzalez et al. 2011; Memon et al. 2007). These systems represent an interesting hybrid which provides the benefits of wastewater treatment in combination with the benefits of urban greening.

With the more recent emergence of green wall installations onto the mainstream architectural scene, the impulse to combine this technology with wastewater treatment seems logical. Promoters of the green wall mention in publications that greywater or recycled greywater is a possible irrigation source for the vegetation system (Weinmaster 2009). There are some examples of green wall installations which use recycled greywater, such as the The Gauge in Melbourne, Australia built by The Greenwall Company (Hopkins and Goodwin 2011). However, the idea that the vegetated wall itself can function as the wastewater treatment step is much less developed. One publication suggests that, “Greywater is another possible source for irrigation. A green wall also filters the water before releasing or recycling it” (Weinmaster 2009). Another takes the idea a step further by mentioning treatment mechanisms: “Living wall systems can be developed to recycle greywater from the building by cleaning it through a linear wetland or biofiltration system incorporated into a green wall system” (Hopkins and Goodwin 2011). Possibly the furthest development of the green wall greywater treatment system is given by Folke Günther on his personal website (Günther 2006). Here he gives design and construction details for a wall structure which incorporates plants and also uses greywater as the irrigation source. However the limited information available on the water treatment aspect is the short description that, “Bacterials in the porous material break down organic pollutants. The water trickling down through the wall will nourish the plants at the same time as it will be purified".

Although the idea that a vegetated wall structure can potentially treat wastewater has been suggested, there is a lack of scientific study regarding this claim. Before architects and home gardeners begin to take on this notion and incorporate greywater as the irrigation source for their green walls or even attempt to reuse the irrigation water in homes and buildings, the actual wastewater treatment potential for such a structure should be examined and eventually optimized through scientific investigations. This is the underlying motivation behind this research project.

1.2 Vegetated greywater treatment walls- Implications for intermittent media filter design

The approach for this study was to begin with a well-established wastewater treatment method and modify the design to become a vegetated wall structure. The chosen treatment method was the intermittent biological filter. This method involves intermittent dosing of wastewater onto a bed of porous media—a technique which has been used in various wastewater treatment systems in Europe and the United states since the 19th century (Widrig et al. 1996). Today, applications of this treatment method are commonly employed as part of on-site wastewater management systems, often as a secondary treatment step for septic tank effluent. This can include soil infiltration systems, intermittent sand/media filters, or aerobic biofilters used in the pre-treatment of constructed wetland systems (Crites and Tchobanoglous 1998; Heistad et al. 2006).

A large amount of the scientific research and practical experience available fall under the heading of intermittent *sand* filters. Switching the media in the porous bed from sand to other granular media has differing treatment effects due to mineral composition and particle size distribution, but the treatment mechanisms remain similar. Therefore the term intermittent media filters will be used to encompass all systems utilizing intermittent dosing of wastewater over a porous bed of sorted homogenous media (i.e. excluding soil infiltration systems).

A description of the typical intermittent media filter for on-site secondary treatment of wastewater is given by the United States Environmental Protection Agency (USEPA) (USEPA 2002). The typical design, shown in Figure 1, is a below ground excavation 91 – 122cm deep, with a filter media depth of 46 – 92cm. A distribution network doses the surface of the media between 12-24 times per day, and an under drain system collects the filter effluent for further treatment or disposal. Usually the filter is covered or buried.

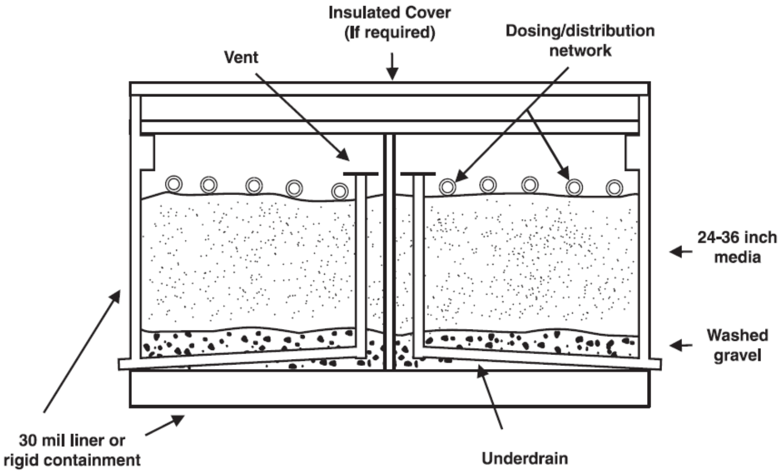


Figure 1 Typical intermittent sand filter design (Source: USEPA 2002)

Using this description as the typical intermittent media filter design and modifying the form into a vegetated wall structure introduces distinct design differences.

The first major design difference involves the drastic modification of surface area to height ratio. While the traditional intermittent media filter utilizes a filter height of around 60cm and a large surface area upon which wastewater is applied, a wall structure presents a form with much reduced dosing area but much increased height. With fixed wastewater volume a smaller dosing area increases the hydraulic loading, which is calculated as the unit volume of wastewater applied daily to the filter per unit surface area ($m^3/m^2 \cdot day$). The hydraulic load and filter height are both factors that directly impact the effluent quality achieved from intermittent media filters (USEPA 2002; Crites and Tchobanoglous 1998; Anderson et al. 1985; Stevik et al. 1999b; Widrig et al. 1996; Schwager and Boller 1997; Torrens et al. 2009a).

The second major design difference for the intermittent media filter modified into a wall structure is the above-ground construction. Most systems in use today are underground in lined excavations or prefabricated tanks. Above-ground construction introduces possibilities regarding the material choice for the containing walls holding the filter material. The use of a perforated material for the containing walls, as opposed to a watertight container typical of underground constructions, implies greater exposure to the atmosphere. This may have implications for treatment, as well as practical implications such as increased smell.

The final design difference for the modified intermittent media filter is the incorporation of vegetation. The green walls popping up in cities today use plants primarily as an aesthetic feature. The function of plants in a green wall which also doubles as an intermittent media filter for wastewater treatment may extend beyond the aesthetic. Experience with constructed wetlands has shown that vegetation has several direct and indirect impacts on wastewater treatment in biological filter systems (Stottmeister et al. 2003). Similar treatment impacts may be observed when plants are incorporated into an intermittent media filter with the vegetated wall design.

1.3 Objectives

The overall goal of this research was to investigate the treatment performance of an intermittent media greywater filter constructed in the form of a wall. Specifically, the objectives were to:

1) Examine the removal characteristics over the depth of the filter.
2) Examine the effect of permeable containing walls
3) Examine the combined effect of the vegetation and vegetation irrigation system.

The additional goal was to determine practical implications of the design modifications.

2. LITERATURE REVIEW

2.1 Source separation

The design for this project relies on a source separating system which divides domestic wastewater into blackwater, the toilet fraction, and greywater, the remaining fraction. These two wastewater streams have distinct chemical characteristics and separation presents advantages not only for reaching treatment goals but also for harnessing the resource potential in both streams.

Blackwater generally includes urine and feces, together with toilet paper and flush water or cleansing water. Urine and feces contribute the majority of the nutrients nitrogen, phosphorus, and potassium found in domestic wastewater (Larsen et al. 2009). These elements are vital plant macronutrients required for productive agriculture, which is the driver behind many efforts to harness blackwater as a fertilizer source (*Ibid*.). The use of excreta in agriculture also benefits soil structure by increasing water-holding and ion-buffering capacities due to the content of organic matter in feces (WHO 2006).

Separating out the blackwater and recycling the nutrient load back to the land is favorable also from the wastewater treatment aspect because organic matter and nutrients present a major risk for eutrophication of natural surface waters. This is the major reason that treatment requirements for wastewater are set in place. In Norway the regulations for wastewater treatment systems releasing treated effluent to the most sensitive environments requires a 90% reduction of phosphorus and 90% reduction of organic material in the form of BOD_5 (5-day Biochemical Oxygen Demand) (Miljøverndepartementet 2012). However, the lower nutrient content in greywater allows reduced requirements in some cases.

Apart from high concentrations of nutrients and organic matter, the blackwater fraction of domestic wastewater includes the majority of human pharmaceuticals and hormones (Lienert et al. 2007), as well as the major pathogenic microorganism load (WHO 2006). Therefore, with the removal of the toilet fraction the greywater stream has a decreased load of all of these constituents, allowing for more simplified and/or more compact treatment systems. Studies have shown that this leads to lower operational and yearly costs for source separating systems versus systems treating full strength domestic wastewater (Müllegger et al. 2004). This also makes greywater an attractive candidate for targeted reuse schemes such as irrigation, toilet flushing or car washing, and groundwater recharge.

2.2 Greywater composition

Greywater represents a highly variable liquid stream, in terms of both volume and chemistry. The composition of greywater depends on the quality of water supply, the materials used in the water distribution network, and on the activities in the household (Eriksson et al. 2002). The household pollutant contribution varies widely according to occupant lifestyle, age distribution, and consumer product use (Donner et al. 2010). Chemical characteristics also vary with the specific household fixture that water is collected from, (e.g. faucets, showers, kitchen, laundry) but due to the absence of contributions from the toilet, basic generalizations can be made. Without the addition of urine, feces, flushwater, and toilet paper, greywater constitutes a reduced volume and contains lower amounts of microorganisms, nutrients, and organic matter compared to combined municipal wastewater (Eriksson et al. 2002; Müllegger et al. 2004; Ledin et al. 2001).

2.2.1 Nutrients

On average, greywater represents 60-80% of the total household water consumption (Jenssen and Vråle 2004; Eriksson et al. 2002). In countries such as Norway which mandate phosphate-free detergents, greywater contributes only 10% of the nitrogen, 26% of the phosphorus, and 21% of the potassium to the combined stream (Jenssen and Vråle 2004). The BOD_5 : nitrogen : phosphorus ratio is around 100 : 20 : 5 for combined municipal wastewater stream, but only about 100 : 4 : 1 for greywater (Müllegger et al. 2004). Luckily, the optimal ratio for heterotrophic growth is very close to the greywater ratio (100 : 5 : 1), suggesting that biological treatment of greywater is possible without a nutrient limiting problem (*Ibid.*).

2.2.2 Organic matter

Greywater studies have reported wide variations in the concentrations of organic matter and suspended solids. For mixed greywater, biochemical oxygen demand (BOD) has been reported in a range of 90 – 360 mg l^{-1} and chemical oxygen demand (COD) in the range of 13 – 8000 mg l^{-1} (Eriksson et al. 2002). Since much of the COD load originates from chemical addition due to household product use, the COD : BOD ratio in greywater is usually high, up to 4 : 1 (Jefferson et al. 2001). In addition, the greywater fraction has a comparatively low level of solids, suggesting that a larger fraction of the organic load is dissolved (Jefferson et al. 2001). Despite a lower amount of solids, there is some concern over the combination of the particles and surfactants from detergents, which could cause a stabilization of the colloidal phase and reduce efficiency of pre-treatment such as settling (Ledin et al. 2001).

2.2.3 Pathogenic microorganisms

Pathogenic microorganisms have the potential to enter the greywater stream through the rinsing of uncooked food and raw meat, but the main risk of introducing pathogens into greywater comes from fecal contamination through laundry, diapers, childcare, and showering (Ottosson 2004). In general the fecal pathogen hazard is considered to be lower in greywater compared to mixed municipal wastewater. In addition, the high load of easily degradable organic compounds in greywater favors the growth of fecal indicators in greywater systems; thus, bacterial indicator numbers have the potential to largely overestimate the fecal loads and associated risk from greywater (WHO 2006).

Although lower than mixed municipal wastewater, the microbial contamination of greywater is significant and must be considered when developing treatment systems, especially if the goal is to re-use the treated effluent. For unrestricted irrigation with greywater, the World Health Organization (WHO) has set a guideline value of less than 10^3 *Escherichia coli* (*E. coli*) per 100 ml, and less than 1 Helminth egg per liter (WHO 2006). This value is meant to represent a tolerable burden of disease at ≤ 10^{-6} DALY (disability adjusted life year) per person per year, or in other words, the same level of health protection as existing WHO drinking water quality standards. The guideline *E. coli* value is believed to ensure a comparative level of safety against bacterial and viral pathogens, but a clear value for parasitic protozoa has not been established (*Ibid.*).

2.3 Intermittent biological filtration systems

2.3.1 General Concepts

In an intermittent biological filter system the bed of porous media is designed to operate as an aerobic fixed biomass reactor (Bancolé et al. 2003). Long-term intermittent dosing of wastewater establishes a diverse microbial community which attaches to the media surfaces in a zoogleal film, now known as biofilm (Calaway 1957). As water passes through the biological filter physical filtration and chemical interactions with the filter material contribute to the removal of pollutants, but the biological transformations within the biofilm are believed to play the most vital purification roles (Anderson et al. 1985).

A significant number of scientific investigations have aimed to relate variables of filter design to treatment performance. Generally accepted design variables impacting effluent water quality include: level of pre-treatment (i.e. chemical composition of wastewater applied to the filter), mineral composition and particle size distribution of the porous media, filter media depth, hydraulic and organic loading rates, and dosing technique and schedule (Widrig et al. 1996; Crites and Tchobanoglous 1998; USEPA 2002). Integrating these variables into an overall model for optimizing treatment performance has proven difficult. The design variables are interrelated and impact the nature of biofilm development and microbial processing efficiency. This is further complicated by the intermittent hydraulic application, which exposes the system to a constantly changing degree of saturation and pollutant concentration (Boller et al. 1994b).

Investigations into the hydraulic nature of the intermittent biologic filter have uncovered general patterns. After the application of an effluent dose, water travels through the filter with the main flow direction of gravitational force (Auset et al. 2005). Several studies have noted that immediately after the dosage event a fast-moving wetting front travels through the upper layer of the filter until at some depth a steady-state unsaturated flow is achieved (Stevik et al. 1999b; Boller et al. 1994b). In the drainage period following dose application the water saturation in the upper portion gradually falls to a minimum and pore spaces fill with increasing amounts of air. At the minimum saturation level the water in pore spaces is immobile, connected only by thin films along pore walls (Auset et al. 2005). With the next flush of water the pore spaces fill again with water, reconnecting the immobile water, and advancing the hydraulic pulse.

The immobile water and unsaturated flow stages are the most important hydraulic phases for purification. This is when water and pollutants come into intimate contact with the filter media and biofilm surfaces. Physical removal mechanisms such as adsorption as well as biological degradation processes include a contact time dependency between the pollutant and the media/biofilm surface (Bancolé et al. 2003; Stevik et al. 2004). When the hydraulic dose is too large, the level of saturation increases and preferential flow in the large pore spaces leads to faster moving wetting fronts and breakthrough at the outlet of the filter of unoxidized materials as well as pathogens (Boller et al. 1994b; Schwager and Boller 1997; Bancolé et al. 2003; Lance and Gerba 1984). It follows that elimination of organic matter, oxidation of nitrogen, and removal of pathogen indicator microorganisms are all strongly related to hydraulic retention time (Bancolé et al. 2003; Stevik et al. 1999b; Torrens et al. 2009b). Filter parameters which directly influence the hydraulic retention time— and thus treatment performance— include clogging layer development, filter media grain size, dosing method coupled with size and frequency of dose, and filter depth.

Additional filter characteristics which have been found to influence treatment performance are related to chemistry. This includes the mineral composition of the filter media and characteristics of the applied wastewater. This research will focus on the physical factors, but a note on chemistry is warranted.

2.3.2 Chemical Factors

The mineral composition of the filter media influences the removal of some of the constituents of concern in wastewater. While the removal of organic matter is not greatly dependent on the composition of inert filter material (Weaver et al. 1998) the removal of nutrients is affected by the choice of media. Boller et al. (1994b) state that in order for full nitrification to be possible, a filter media with at least some amount of calcium carbonate is necessary. The removal of phosphorus, which depends on sorption, is also reliant on the chemistry of the filter media. Whereas normal sand has a low phosphorus sorption capacity, with phosphorus breakthrough occurring even in the first weeks of filter operation (VanCuyk et al. 2001), newly engineered media has been designed specifically for improved phosphorus retention (Jenssen and Krogstad 2002).

Stevik et al. (1999a) examined the removal of *E. coli* in intermittent filters based on a range of physical factors, including media size and surface area and dosing rate, as well as chemical factors such as media cation exchange capacity, and wastewater ionic strength and pH. Theoretically certain chemical factors should have an influence on the removal of microorganisms, such as higher cation exchange capacity increasing adsorption of bacteria (Stevik et al. 2004). However, due to high flow velocity and irreversible fouling of the surfaces these chemical factors show less influence in infiltration systems (Stevik et al. 1999a). The investigation by Stevik et al. (1999a) concluded that physical factors of hydraulic loading rate, effective grain size, and specific surface area were much more significant than any chemical factor in the removal of *E. coli*.

Temperature has an effect on the rate of chemical and biological processes and reports claim that filter performance is better in locations with warmer climates (USEPA 2002). However, a number of studies failed to find a significant effect of temperature on the treatment capacities of intermittent filters within the temperature range of a given geographic location. Ausland (1998) found no significant difference between removal of fecal coliforms and organics (BOD_7) with variation of filter temperature between 2°-17°C. Chen (2003) found in fact that highest BOD_5 removal occurred at lowest temperature during a study of intermittent sand filters between 5°-20° C. Williamson (2012) found that effluent temperature variations in cold climates had almost no effect on BOD_5 values or on nitrogen removal. Torrens et al. (2009b) also did not find a significant difference of removal of bacterial or viral indicators based on temperature differences in vertical flow filters. Ausland (1998) explains that this phenomenon may be due to the fact that the minimum retention times were long enough to exhibit high removal rates regardless of temperature.

2.3.3 Hydraulic retention time – design factors

Clogging layer

Long-term operation of intermittent infiltration systems often results in a zone of reduced permeability at the infiltrative surface, known as a clogging zone. This is attributed to accumulation of organic materials and suspended solids at the infiltrative surface, in addition to a higher rate of biofilm development in uppermost section of filters (Siegrist 1987). The hydraulic effect of clogging zone development is a decrease in permeability, restricting infiltration rate into the filter. When wastewater loading rate exceeds the infiltration capacity ponding results above the filter surface, but unsaturated flow conditions are maintained below the clogging layer and throughout the entire filter volume (Siegrist 1987; Lance and Gerba 1984). Studies have also shown that clogging layer development at the infiltrative surface leads to a more uniform redistribution of effluent over the filter, especially when a gravity-dosed system is used (Van Cuyk et al. 2001; Ausland 1998). Uniform distribution and maintenance of unsaturated flow are both considered to increase the surface area and contact time between wastewater and the soil/media matrix, extending hydraulic retention time and leading to gains in purification.

Several investigations have reported that the highest rate of pathogen removal takes in the upper portion of filters (Stevik et al. 1999b; VanCuyk et al. 2001). The clogging layer is believed to play some role in this pattern. The accumulation of suspended sediments, organic matter, and biofilm at the infiltrative surface blocks large pore spaces, which is believed to enhance the effect of bacterial straining. Straining is one of the mechanisms for immobilization of bacteria travelling through porous media, and involves the physical blockage of movement through pore spaces smaller than the bacteria itself (Stevik et al. 2004); viruses, however, are too small to be immobilized by straining (Lance and Gerba 1984). Additional explanations given for the higher rate of removal in the upper part of the filter are higher densities of active protozoa and better oxygen conditions (Ausland 1998). VanCuyk et al. (2001) reports that a clogging layer in soil infiltration systems is in fact necessary, and when absent the purification performance will suffer.

Clogging layer at the infiltrative surface has documented hydraulic and purification benefits, but is also the major mode of system failure in intermittent biological filter systems. When clogging restricts infiltration excessively, complete hydraulic dysfunction will result in flooding of the system, anaerobic conditions, and reduced purification (Anderson et al. 1985; Crites and Tchobanoglous 1998; VanCuyk et al. 2001; Venhuizen 1998). As the goal of decentralized treatment involves a system with very low vulnerability, many intermittent media filter systems employ methods aimed to avoid severe clogging of the filter surface.

The techniques used to avoid clogging of intermittent filter systems include: the use of coarser filter media, even fine gravel in the range of 2.5-6mm; improving pre-treatment to lower the organic and solids load to the surface of the filter; uniform loading of wastewater using spray distribution; application of small but frequent wastewater doses, once every 30-60 minutes; and employing recirculation by mixing the treated effluent with the untreated wastewater before dosing the filter, thereby lowering the organic concentration of the dose (Venhuizen 1998).

Filter media grain size

The moisture retention capacity of the filter media is directly related to the grain size distribution (Boller et al. 1994b). Smaller grain sizes have higher capillary forces leading to more uniform flow and longer retention times, and thus higher treatment efficiency (Stevik et al. 1999a). However, this also restricts the maximum size of hydraulic load to avoid saturated flow regimes. Coarser filter media allows larger dosage volumes and has the advantage of better hydraulic performance due to less clogging of the surface, but with a grain size too large the wastewater retention is lowered to a point where biological decomposition is inadequate (Anderson et al. 1985). Additionally, the pore sizes in larger filter media (0-4mm or 2-4mm) are larger than bacterial cells, and thus straining is not expected to play a role in immobilization (Ausland 1998).

The typical filter material used in intermittent biological filter systems in Norway has a grain size of 0.5-4mm or 2-4mm (Norsk Rørsenter 2006). This is larger than the intermittent sand filter description provided by the USEPA of 0.25-1.00mm (USEPA 2002). The negative impacts of larger grain sizes can be counteracted with the use of smaller and more frequent dosing (Ausland 1998; Torrens et al. 2009b; Boller et al. 1994b). An important aspect of the filter media is that it is relatively uniform and sorted to exclude fine particles which have the potential to clog the system pores. Uniformity coefficients (d_{60}/d_{10}) of <4.0 and <5.0 have been suggested (USEPA 2002; Norsk Rørsenter 2006).

Dosing

Many intermittently dosed biological filters rely on a pressurized dosing system which allows more even distribution of the wastewater over the dosing surface when compared to gravity fed systems, resulting in much longer mean and minimum retention times in the filters (Ausland 1998). The use of spray nozzles is recommended with the pressurized system to achieve the most even dose distribution (Heistad et al. 2001; Norsk Rørsenter 2006).

Repeated studies have concluded that higher fractionation (i.e. smaller and more frequent application) of the total load of wastewater to the filter surface increases the removal efficiency of pathogen indicators (Torrens et al. 2009b; Ausland 1998; Stevik et al. 1999b) and gives greater reduction of COD and oxidation of nitrogen (Bancolé et al. 2003; Boller et al. 1994b). Larger and less frequent doses can transport unoxidized material quickly through the depth of the filter (“breakthrough”).

However, there exists some upper limit to the frequency of dosing events. Enough time must pass between dosing to allow for effluent infiltration and redistribution, otherwise an almost completely saturated flow regime will develop (Schwager and Boller 1997). Additionally, it has been noted that very high fractionation of the wastewater load encourages biofilm development to concentrate at the very surface of the filter– a higher risk for clogging— versus a lower fractioning of the wastewater load leading to more even biofilm development over the depth of the bed (Bancolé et al. 2003). The USEPA describes a dosing schedule of 12-24 times per day (USEPA 2002) while the Norwegian systems usually have a dosing schedule of 10-50 times per day (Norsk Rørsenter 2006). The differences in dosing are correlated to differing grain size, as smaller grain sizes have higher moisture retention and require more time for the water to infiltrate before the next dose application.

In terms of the total hydraulic load, the optimum value varies with filter media choice, strength of wastewater applied to the filter, method of dosing, etc., but recommended values are available. The USEPA reports the typical hydraulic loading for intermittent sand filters treating full strength domestic wastewater as 40-80 liters/$m^2 \cdot$day (USEPA 2002). In Norway the typical hydraulic loading for intermittent biofilters treating greywater from cabins and/or residences is reported as 100-250 liters/$m^2 \cdot$day for long term use (Norsk Rørsenter 2006). The corresponding minimum filter surface area for a single residence is reported as 4.5m^2 with a depth of 75cm (including dosing and underdrain layers).

Filter depth

It is widely accepted that purification performance of BOD/COD, total suspended solids (TSS), and ammonia-N is increased with increasing filter depth (Widrig et al. 1996; Bancolé et al. 2003; Torrens et al. 2009a). However, the general conclusion is that after certain filter or vadose zone depth, usually around 60 cm, additional depth is not warranted because removal of these constituents consistently reaches over 90 percent (VanCuyk et al. 2001; Crites and Tchobanoglous 1998). A study conducted over six years investigating intermittent sand filter depths between 30.5 – 76.2 cm sums up the conclusion thus: "Satisfactory treatment is achieved with the widely-accepted standard 61.0cm depth. Although gains in treatment may be achieved with greater depth, doing so is inefficient since any improvement is negligible" (Weaver et al. 1998). This sentiment of diminishing returns related to increased filter depth is encouraged especially because construction costs for traditional sand/media filters are related to excavation and difficulty of tank installation, cost of filter material and amount of excavated material which must be hauled off-site (Venhuizen 1998; Føllesdal 2005)

2.3.4 Hydraulic retention time- design implications for the vegetated wall structure

As discussed previously, modifying the traditional design characteristics of the intermittent media filter to resemble a vegetated wall structure, or "green wall" involves a major change in the traditional filter surface area to filter height ratio. A drastic decrease in filter surface area may be conducive to an urban environment, as this allows a decrease in spatial footprint, but a decrease in filter surface area also implies an increase in total hydraulic load. Assuming that a system has already reached the upper limit of dose fractionation (i.e. number of doses per day), this increase in hydraulic load implies an increase in the volume of each dose applied. An increase in dose volume is accompanied by a decrease in hydraulic residence time and thus lowered treatment performance (Boller et al. 1994b; Schwager and Boller 1997; Bancolé et al. 2003; Stevik et al. 1999b; Torrens et al. 2009b).

While the decrease in filter surface area presumably implies lowered treatment performance, the question arises as to whether an accompanied increase in filter depth can counteract that effect. A direct investigation of this hypothesis has not been found in scientific literature available; however, the investigations by Stevik et al. (1999b) give some insight. In this study, researchers found that a doubling of dose volume corresponded to a mean retention time reduced by about half. Additionally, the fast moving wetting front immediately after a dose event reached unsaturated steady state at approximately 20cm depth for the smaller dosage, and at about 40cm depth for the doubled dosage. The larger dose volume corresponded to reduced removal rate of *E. coli*.

It seems possible that the loss of retention time and reduced depth of unsaturated flow brought on by larger dose volumes can be compensated by additional height added to the filter. However, as Stevik et al. (1999b) found, a larger dose is accompanied by a much higher flow velocity through the upper portion of the filter. Stevik et al. (1999b) along with many other studies (Ausland 1998; Van Cuyk et al. 2001; Schwager and Boller 1997; Widrig et al. 1996) found that the bulk of pathogen and organic removal takes place in the very upper portion of intermittent filters. This is because the main biomass accumulation takes place in the upper section of filters, as one investigation found mainly in the upper 10cm of a filter and down to about 30cm depth (Schwager and Boller 1997). So as Stevik et al. (1999b) note, since the upper portion of the filter is the most important for treatment, adsorption processes will be less effective with larger doses due to higher velocities through the upper section, causing removal to decline. This implies that a greater depth cannot compensate for the loss of residence time through the first vital 10cm filter depth.

2.3.5 Aeration

One explanation for greater biofilm development and treatment efficiency at the surface of intermittent filters is the higher oxygen availability there (Ausland 1998; Petitjean et al. 2011). The transformation of the intermittent media filter into a vegetated wall structure necessitates a shrinking of this vital surface area and an expansion of filter height. Aeration of the entire filter, especially regions furthest from the surface, is therefore an important factor to consider in relation to treatment performance.

Oxygen transfer into intermittent media filters is supplied from three sources: dissolved oxygen present in wastewater, convection due to intermittent dosing, and diffusion processes (Torrens et al. 2009a). The dissolved oxygen in the wastewater itself is considered negligible. Convection and molecular diffusion of oxygen into the filter is considered to take place by exchanges with the atmosphere through the filter bed surface (Bancolé et al. 2003).

The convective transfer takes place immediately after a dosing event, as water percolates at higher velocities and induces airflow on its backside (Schwager and Boller 1997). Diffusive transfer is the dominant process for re-oxygenation in the time between dosing events, and is a function of air porosity, vertical distribution of oxidizable pollution, and the time available for diffusive transfers (Bancolé et al. 2003).

Studies which have attempted to model the air flow in vertical-flow intermittent media filters typically use a one-dimensional two-phase flow model, with boundaries placed at the top and bottom of filters (Petitjean et al. 2011; Schwager and Boller 1997; Forquet et al. 2009). These models all show that the bottom section of the filters are usually saturated (the “seepage face”), and thus do not contribute to aeration. This leaves the surface of the filter responsible for all oxygen flux into the system, which researchers are beginning to understand may not be highly efficient. The model constructed by Petitjean et al. (2011) found that even under optimal conditions, only fifty percent of the filter is properly re-oxygenated at any time. The researchers conclude that, “This clearly may lead to limitation in aerobic biodegradation and is one of the reasons, along with decrease in substrate availability, that aerobic bacterial activity happens mainly in the first few centimeters in aerobic filters”.

Little evidence is available regarding the relationship between the physical containment of the filter media and the oxygen exchange. As Schwager and Boller (1997) note, buried filters which are covered by soil can slow down air diffusion into the filter, especially during wet weather conditions. An open filter is described as enhancing the air access as well as allowing easier control of the filter surface, but of course this prohibits land use above the filter.

2.3.6 Aeration – design implications for the vegetated wall structure

It is logical that attempts to model the air flow in intermittent media filters consider the boundary between atmosphere and filter to be located at the filter surface, as nearly every example found is either buried under ground or enclosed in a tank construction. A major design modification when attempting to transform this typical intermittent filter design into a vegetated wall structure is the transformation from a subterranean to an above-ground construction. This allows the possibility to design the containing walls for the filter media to be in direct contact with the atmosphere. As Schwager and Boller (1997) found that diffusive processes play the most important role in re-oxygenation of intermittent filters, it is possible that a filter with vertical surfaces open to the atmosphere provides greater opportunity for diffusive transfer of oxygen.

Greater interaction with the atmosphere also introduces practical concerns. The unpleasant odor associated with wastewater could become prominent with an above-ground and open filter. An above-ground filter may also introduce greater risk for human contact with untreated wastewater, which contains some level of pathogens.

2.3.7 Vegetation

Experience with constructed wetland systems has shown that the incorporation of vegetation can influence wastewater purification in filter based treatment schemes. Some effects are due to the physical presence of vegetation, including temperature buffering and additional surface area for attached microbial growth in the root zone (Stottmeister et al. 2003). Other effects are derived from the metabolism of the plants themselves including nutrient uptake and gas transfer in the root zone.

The bulk of research on plant incorporation into filter systems has been conducted with species of marsh plants, especially reeds. These types of plants are extremely productive and their special adaptation to saturated conditions involves a transfer of oxygen into the root zone (Stottmeister et al. 2003). Comparisons of wastewater filters with and without reeds have shown that the incorporation of this vegetation yields significantly better organic matter, nitrogen, and phosphorus removal (Gikas and Tsihrintzis 2012; Torrens et al. 2009a). In a study comparing a vertical filter planted with reeds and an unplanted sand filter, the vegetated system generally removed pharmaceuticals and personal care products more efficiently than the sand filter, likely due to better oxygenation of the filter bed (Matamoros et al. 2007).

Some negative impacts of vegetation have been documented. An extensive root system inside the media filter can potentially clog the pore system (Stottmeister et al. 2003), or cause preferential pathways leading to hydraulic short circuiting (Torrens et al. 2009a). Very high transpiration rates in warm climates can lead to a more concentrated effluent, especially in terms of TSS and salinity (Stottmeister et al. 2003; Coleman et al. 2001). Vegetated systems require specific maintenance routines including harvesting dead plant material; if not performed the breakdown of plant material can increase organic and nutrient loads to the effluent.

Information regarding the effects of non-marsh plants on wastewater filter systems is limited. Henderson et al. (2007) investigated the effects of various shrub and groundcover species on the treatment of stormwater runoff in biofiltration mesocosms. While vegetation was reported to make little difference in the removal of organic matter, the nitrogen and phosphorus removal were significantly better in planted systems. The enhanced nutrient removal in the vegetated filters was attributed to higher microbial activity and population of microbes occurring in the rhizosphere. Garland et al. (2004) tested greywater use in hydroponic production systems and found that hydroponic systems containing lettuce and wheat rapidly degraded surfactant chemicals. This was also attributed to microbial activity in the root zone.

Documentation of the interactions between vegetation and wastewater also includes effects on the plants themselves. This is important especially when considering the cultivation of edible plant species in relation to wastewater reuse schemes.

Several studies have investigated the effects of using greywater to irrigate edible plants, with mixed results. According to Misra et al. (2010) greywater may present problems with inhospitable pH, excess salt, deficiency or toxicity of nutrients and pollutants including surfactants. Wiel-Shafran et al. (2006) irrigated lettuce plants with laundry greywater and found that the elevated boron and salt concentrations produced noticeable chlorosis. Garland et al. (2004) found hydroponic systems containing typical per capita surfactant production rates triggered reduced growth of lettuce plants, but wheat was not affected. Misra et al. (2010) irrigated tomato plants with various forms of greywater and tap water and found that the plants did not exhibit signs of toxicity.

The nutrient content in greywater can potentially benefit plant production, but these results are also varied. Misra et al. (2010) found that tomatoes irrigated with greywater had significantly higher stem and leaf biomass and greater uptake of 7 out of 10 nutrients. However, Finley et al. (2009) investigated lettuce, peppers, and carrots and found no significant difference in dry crop weight between greywater and tap water irrigation source. It is likely that the mixed findings regarding studies of greywater irrigation are due in part to the extremely variable content and concentrations of greywater sources.

Hygienic concerns are extremely important when considering wastewater irrigation for edible plant crops. The World Health Organization produces guidelines for these matters (WHO 2006). Finley et al. (2009) found no significant difference in fecal coliform levels on crop surfaces between plants irrigated with raw greywater, treated greywater, and tap water. The conclusion to this, and additional greywater studies, is that the actual application process of the water introduces the most risk for pathogen contact with plant surfaces (*Ibid*; Ledin et al. 2001). It is recommended that greywater is applied directly to the soil and plant roots, avoiding contact with leaves and edible surfaces. Application of wastewater to plants is also an action which puts humans at risk for contact with pathogens in the water.

The use of greywater for irrigation also presents risks to the growing media. Misra et al. (2010) found that greywater and water containing surfactants caused a reduction in capillary rise in soils, and thus reduced soil water retention. Wiel-Shafran et al. (2006) found a lower pH in soil irrigated with greywater (due to increased microbial respiration), increased bacterial populations, and accumulation of anionic surfactants leading to a reduction of capillary rise.

2.3.8 Vegetation- design implications for the vegetated wall structure

The incorporation of vegetation into biological filters introduces a positive effect from microbial root zone interactions in the filter media. Root development inside the filter material also introduces an element of vulnerability to the system, as roots can create open channels and hydraulic short circuiting inside the filter (Torrens et al. 2009a). To reduce this vulnerability, the root system can be kept largely separated from the filter media, with a system put in place to transport water from the filter core to the plants. Even without root zone interactions, the direct supply of greywater from inside the filter to the vegetation introduces the possibility for effects such as decrease in water volume through transpiration, uptake of nutrients, and temperature buffering. All of these effects may be dependent on plant species and development stage, which is outside the scope of this project.

3. METHODOLOGY

3.1 System design

The treatment system consists of a settling tank followed by the experimental intermittent media filter wall. As seen in Figure 2, the wall was divided into three separate sections.

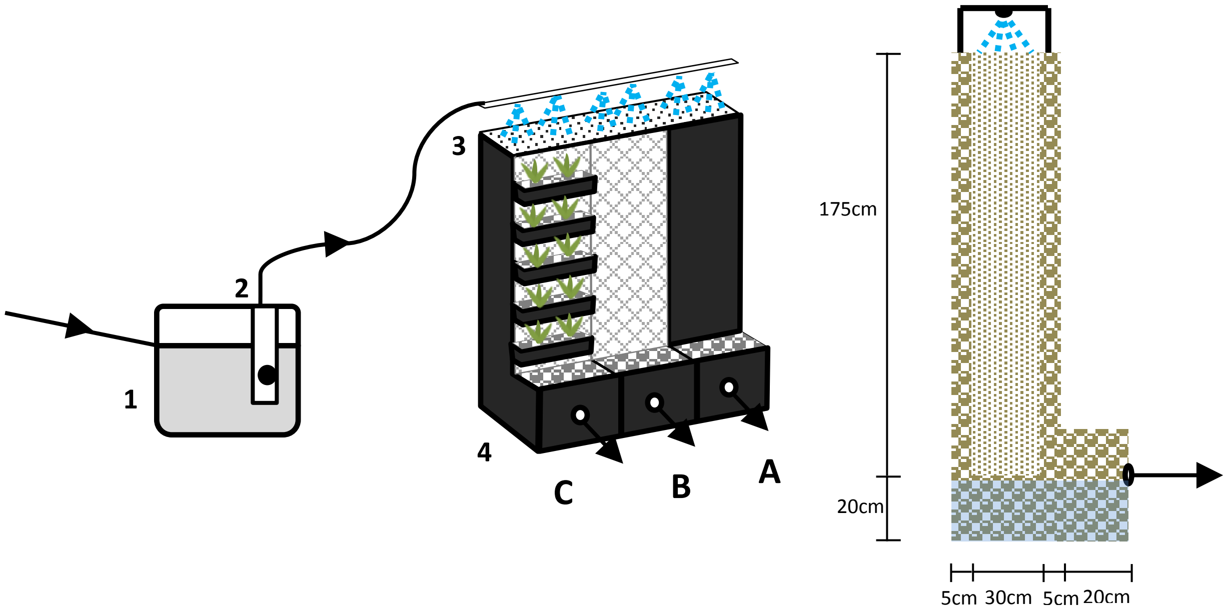


Figure 2 – System design and cross section details. 1) Settling tank 2) pump 3) dosing surface 4) outlets from sections A, B, and C

The system was positioned next to the Fløy IV building, part of the Mathematical Sciences and Technology (IMT) department on the Norwegian University of Life Sciences (UMB) campus. The walls are freestanding facing westward and constructed using waterproof plywood. The bottom drainage portion is lined with plastic (PE) liner. All sections of the wall are filled with lightweight expanded clay aggregates (LECA). The core of the wall is filled with NR 2-4mm Filtralite (Weber, Saint-Gobain) to a depth of 175 cm, which comprises the filter volume. This is sandwiched between a 5cm layer of NR 10-20 mm Filtralite (Weber, Saint-Gobain), with a nylon netting preventing mixing of the two LECA components. A 20cm depth of NR 10-20mm Filtralite is used at the bottom drainage section.

The filter media in Section A is completely enclosed in plastic liner and plywood walls. The filter media in Sections B and C are enclosed in a polyester/PVC geotextile grid with openings of 3mm x 3mm (Telenet from Teletextiles). These sections have plywood walls only on two sides, while the front and back are supported with a 10cm x 10cm steel grid. A photo of the filter wall with the three different sections is shown in Figure 3.

Figure 3 – Completed filter wall (left) and detail of planter shelves with felt strip irrigation system (right)

Section C contains 27 strips of 40cm x 10cm synthetic felt (combination of ZinCo SSM45 and Jordan 1438010) which penetrate halfway into the filter media core, and protrude out of the front surface of the wall (see Figure 3). These are used to line the bottom of five planter shelves built onto Section C, installed at depths of 30cm, 60cm, 90cm, 120cm, and 150cm from the dosing surface. The felt strips direct water from the core of the filter to the base of the planter shelves, where the plant plugs were eventually placed.

3.2 Dosing

The greywater source used in this study was from a source separating system in the student dormitories Kaja at the University of Life Sciences in Ås, Norway (Jenssen 2002). Following a settling tank with approximate residence time of one day, the greywater was dosed onto the surface of the filter every 30 minutes using a timer. Application of the dose was performed using spray nozzles (Lechler SZ1 axial-flow full cone nozzles) two in each section for a total of six nozzles. The nozzles were mounted 8cm from the dosing surface, for a 28cm diameter circular wetted area from each nozzle.

Dosing began on 21/08/2012, with adjustments of the dosing duration ranging from 15 to 22 seconds, until on 10/9/2012 the dosing duration was fixed at 21 second doses delivered every 30 minutes. A test of the nozzles gave an approximate application rate of 0.06 liters/second per nozzle, or approximately 360 liters/day for all three sections combined. This was later confirmed with filling rates during sampling periods. Using the wetted area from each nozzle, the resulting loading rate is approximately 980 liters/m^2·d. When loading is calculated using the entire media surface area instead of wetted surface area, the resulting loading rate is approximately 670 liters/m^2·d.

Dosing was interrupted for approximately 48 hours from 30/10/2012 to 1/11/2012 due to freezing of the delivery pipes. The dosing was switched off and the pipes and settling tank re-routed to an indoor configuration before re-initiating dosing. After re-initiation, the dosage to the filters was inadvertently increased because the pipe was drastically shortened, which decreased head-loss and increased the pressure to the nozzles. The final sampling on 05/11/2012 was collected while this increased dosing regimen was in place.

3.3 Plants

Four species of plants were cultivated from seed starting 3/7/2012. These were “Amerikanischer brauner” lettuce, “Nores” spinach, “Half tall” leaf cabbage, and marigolds (Lord Nelson) and were planted into Jiffy7 42mm planting media plugs (Jiffy Group), and watered with tap water until transplantation. On 24/8/2012 the lettuce and marigold plugs were placed into the planter boxes of the treatment wall Section C in an alternating pattern. The other plant species had unsatisfactory growth and/or major pest invasion, so were not used further. After transplantation the only source of irrigation (apart from rainwater interception) came from the felt strips transporting water from inside the filter.

3.4 Sampling

Sampling ports were placed in each of the wall sections at 5cm depth, 15cm depth, 30cm depth, and 100cm depth from the dosing surface. Initially, these were 4.5cm diameter plastic funnels connected to plastic tubing, and placed into the center of the filter core during construction. However, many of these sampling ports were nonfunctional (possibly due to kinks in the plastic tubing). All ports except for the uppermost (5cm depth) were replaced with a V-shaped steel rod inserted through 2/3 of the thickness of filter media. These were connected to plastic tubes and directed water to the outlet pipe of the system.

The outlets from each section were placed at 20cm from ground surface, allowing this depth of saturated water level (175cm from dosing surface). The outlets led to the local sewer network. Due to a leakage in the lining of Section A the outlet was nonfunctional. However, a standing water level 2-3 centimeters below the outlet allowed for a siphon to be set up for collection of an outlet sample at Section A. For unknown reasons, this standing water level disappeared for approximately three weeks before reappearing, so three sampling events from the outlet at Section A are missing.

Effluent samples were collected over an eight week period between 13/9/2012 and 06/11/2012. All samples were collected in 1-liter plastic bottles. Five of the sampling events included only inlet and outlet samples, and these were collected as 1-liter grab samples. The inlet sample was always collected during the dosing event immediately following outlet collection. Two out of these five sampling dates lack data from Section A due the malfunction of the outlet

Due to the nature of grab samples, the day and time that samples are collected can influence results. All grab samples were collected between 8AM and 10AM, which could possibly exhibit differing patterns from samples collected in the afternoon or evening.

Three sampling events included effluent samples from all sample portals, in addition to the inlet and the outlets (note: one out of three of these events lack data from the outlet to Section A due to malfunction). Due to a great variability in flow rates from the sampling portals, the samples at 5cm, 15cm, and 30cm depth were left for 24 hours and still did not yield a full 1 liter sample. The samples at 100cm depth and outlets were taken as composite samples over the same 24 hour period, and inlet sample taken at the end of the 24 hour period.

Samples for bacterial analysis were taken on 29/11/12 and 30/11/12 from the three section outlets as well as the inlet. Samples were collected as grab samples in 15ml sterile tubes.

3.5 Analysis

All samples were taken immediately to the laboratory for analysis. BOD_5 was measured using the WTW OxiTop system according to the user instruction manual. Measurements of pH and temperature were made with Hanna Instruments HI 84431 Total Alkalinity meter. To measure the total suspended solids (TSS) samples of 100-200ml were vacuum filtered through 47µm glass microfiber filters (Whatman Cat No 1822-047) and stored in a 100°C oven for 24 hours before final weighing. The analysis for COD, total nitrogen, nitrate, and total phosphorus were all conducted using testing kits from Hach Lange (testing kits LCK 314 & LCK 614; LCK 138; LCK 339; and LCK 349, respectively). The cuvette samples were digested using the Hach Lange Thermostat LT200, and final values determined using the Hach Lange DR 2800 spectrophotometer.

Bacterial analysis was conducted using the INDEXX Colilert-18 Quanti-tray and Quanti-tray/2000 systems, with INDEXX DST powder nutrient reagent capsules. Serial dilution was performed with deionized filtered water (which was also run as a negative control) and sterile vessels. INDEXX Quanti Sealer model 2X was used to seal the Quanti-trays before incubation in a 35°C oven for 18-22 hours. A 6-Watt fluorescent UV lamp was used to read results.

Two salt tracer tests were conducted, on 09/11/12 and 13/11/12. During these tests 20ml of NaCl solution (electrical conductivity 200 mS/cm) was injected during one dosing session, and measurements of electrical conductivity were taken at the outlets using a WTW TetraCon 325 Conductivity Meter.

Statistical analysis was performed using the Minitab15 statistical package (Minitab Inc.).

4. RESULTS & DISCUSSION

4.1 General treatment efficiency

The characterization of settling tank effluent entering the filters and the corresponding treatment performance from each section is presented in Table 1. The organic strength of the settling tank effluent in terms of BOD_5 and COD is in agreement with values reported in literature (Eriksson et al. 2002).

Table 1 – System treatment performance: average outlet concentration (SD) and % removal

	pH	TSS		COD		BOD_5		Tot P		Tot N	
		mg/l	%	mg/l	%	mg/l	%	mg/l	%	mg/l	%
Settling Tank Effluent	7.1 (0.1)	39 (8.2)	-	241 (26.2)	-	129 (46.8)	-	1.15 (0.11)	-	12.7 (1.4)	-
Section A Effluent	7.1 (0.2)	2 (1.3)	95	43 (6.6)	82	6 (1.8)	95	0.33 (0.19)	71	8.8 (1.7)	31
Section B Effluent	7.5 (0.4)	4 (2.8)	90	35 (6.5)	85	4 (0.8)	97	0.36 (0.17)	69	8.8 (2.3)	31
Section C Effluent	7.6 (0.5)	2 (2.2)	95	29 (8.5)	88	2 (1.2)	98	0.26 (0.06)	77	8.4 (1.6)	34

The treatment efficiencies expected from similar intermittent media filters designed in Norway for the treatment of domestic greywater are found in Table 2 (Norsk Rørsenter 2006).

Table 2 – Expected treatment performance for biological greywater filters in Norway. Adapted from Norsk Rørsenter Miljøblad Nr. 60, 2006, translated from Norwegian.

	Concentration	%
BOD_7	<20 mg/l	>90
COD	<30 mg/l	60-90
Total N	<10 mg/l	>25
Total P	<0.5 mg/l	>75
E. coli	<1000 *E. coli* /100ml	>99

The results in Table 1 compare favorably with the expected treatment values in Table 2. All three sections showed excellent removal of BOD_5 with an average removal of 95%, 97%, and 98% for sections A, B, and C, respectively. Converting to BOD_7 values using the common conversion factor of 1.15, the BOD_7 effluent concentrations fall well below the <20 mg/l expectancy shown in Table 2. The average COD removals were 82% for Section A, 85% for Section B, and 88% for Section C, which fall within the expected efficiency of 60-90% removal for similar filters. However, section C is the only section which meets the <30 mg COD/l effluent average concentration value.

All three sections meet the expected removal and effluent concentration for total nitrogen, with an average reduction of over 30% for all filter sections. The conditions in typical single-pass intermittent filters are not ideal for denitrification, partially due to the lack of an available carbon source after nitrification (Føllesdal 2005). Recirculation is one method which can be employed to improve denitrification and thus total nitrogen removal (Crites and Tchobanologous 1998; USEPA 2002).

The effluent concentrations meet the expected value of <0.5 mg/l total phosphorus for all three sections, but Section C is the only section to meet the removal efficiency expectation, with an average reduction of 77%. The majority of phosphorus removal takes place by sorption to the granular media in the filter, which will decrease over time as the sorption sites on the media are used up (Heistad et al. 2009; Jenssen and Krogstad 2002). This is a major consideration for the lifetime expectancy of a filter. However, the average influent concentration is very low, at 1.15 mg/l, due to separation of the blackwater fraction and the use of non-phosphate containing cleaning products in Norway. This almost meets the regulation in some areas of Norway for a wastewater discharge limit of 1.0 mg/l for phosphorus, before any treatment.

The high level of treatment reached by the system is significant due to the extreme hydraulic load. The expected treatment efficiencies provided in Table 2 correspond to a recommended hydraulic loading rate of 100 - 250 $l/m^2 \cdot d$ and a filter surface area of approximately $4.5m^2$ to treat the greywater for one Norwegian household. The loading rate for the filters in this study was much greater, in the range of approximately 650 – 1000 $l/m^2 \cdot d$ (depending on calculation based on entire surface area or wetted surface area, respectively) and a filter surface area of $0.54m^2$. For per capita greywater production rates of around 100 liters per person per day (Jenssen 2002), a filter surface area slightly over $1m^2$ would be sufficient for a household of four persons. A small surface area per person makes this system an attractive candidate for urban development.

The high level of treatment reached by the system is also significant due to the early-phase sampling. The sampling period during this study was within the first eleven weeks of filter operation. Widrig et al. (1996) classified the first ten weeks of filter operation as the startup period, during which the biofilm inside the filter matures and develops, until treatment stabilizes. After the startup period, the treatment efficiency of organic matter (BOD and COD) is expected to improve (Widrig et al. 1996; Føllesdal 2005). The bacterial communities responsible for nitrogen transformations also take several weeks to establish in intermittent filters (Bahgat et al. 1999). Monitoring of the system over a longer time period is necessary to determine steady-state treatment capacity, in addition to evaluating the vulnerability to clogging, which is a process that develops over many months or years (Siegrist 1987; Widrig et al. 1996).

Due to difficulties obtaining the testing equipment, analysis for the presence of the bacterial indicator *E. coli* was only performed twice, towards the end of the study period. The results are shown in Table 3.

Table 3 – Bacterial analysis

	***E. coli* Concentration** (per 100ml)[a]	
	29/11/2012	30/11/2012
Settling Tank Effluent	920800	488000
Section A Effluent	> 2005	6240
Section B Effluent	> 2005	7820
Section C Effluent	1652	1780

a) MPN (most probable number)

The reduction of *E. coli* was nearly 2 log units (approx. 98%) for Sections A and B, and over 2 log units (>99%) for Section C. This is comparable to the removal efficiency reported in Table 2.

However, none of the filter sections reached the expected concentration of <1000 *E. coli* /100ml reported in Table 2, which is also the concentration limit suggested by the World Health Organization for unrestricted irrigation with greywater.

4.2 Treatment related to filter depth

4.2.1 Organic removal

Figure 4 shows the removal of COD by depth from each wall section. It is evident that all three sections exhibit similar removal as a function of depth, with the bulk of organic removal taking place in the upper 15cm of the filter. A similar pattern was also observed for TSS removal (not shown).

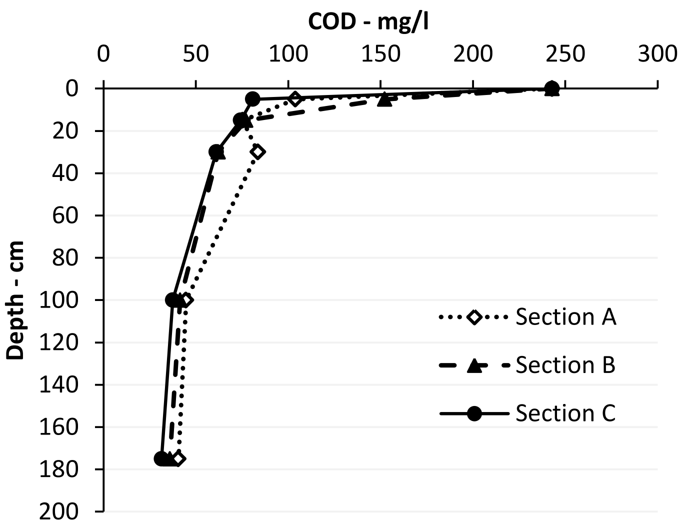


Figure 4 - COD removal with filter depth

An analogous pattern of organic removal has been observed in many investigations of intermittent media filters. The COD removal curve produced by Schwager and Boller (1997) during an investigation of an intermittent sand filter, shown in Figure 5, has a very close resemblance to the profile in Figure 4.

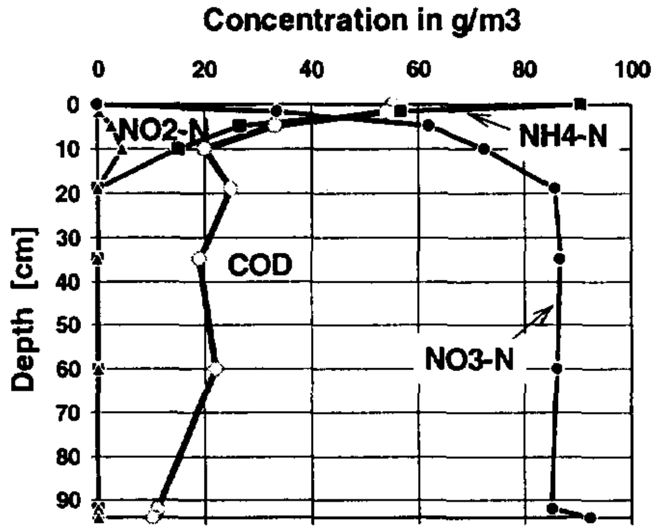
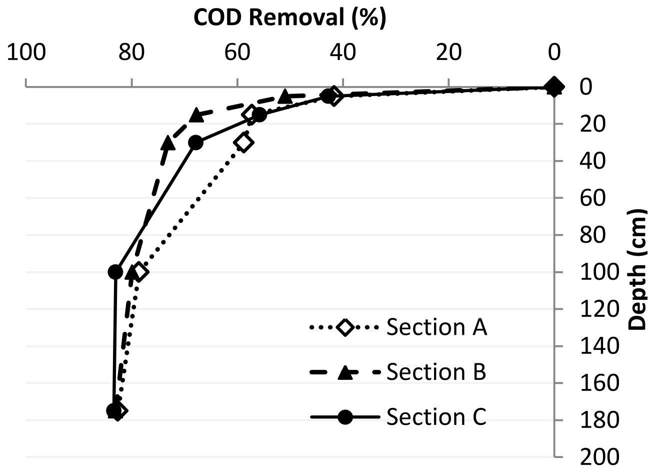


Figure 5 - COD removal and nitrogen transformations with filter depth, in Schwager and Boller 1997.

The explanations given for the high rate of removal at the filter surface are the higher biological activity in this zone due to better oxygen conditions and higher substrate availability (Petitjean et al. 2011; Schwager and Boller 1997; von Felde and Kunst 1997). Under this period of investigation it appears that the first 100cm of filter depth were sufficient to reduce the COD by around 80%, and the additional depth improved organic removal only slightly. This is in line with the concept of greater filter depth and diminishing returns described by researchers (Weaver et al. 1998).

A longer period of investigation may be necessary to fully evaluate the removal capacity related to depth for this filter wall configuration. Miller et al. (1994) reported that the percentage of solids accumulated at the surface of experimental filters decreased over time as the deeper sections became more biologically active. In addition, a wider dosing range may be necessary to fully evaluate possible benefits of the additional depth provided by the wall configuration. As stated previously, the final sampling date was collected during a period when the dosing rate was inadvertently increased three days prior to collection, from a wetted-area dosing rate of approximately 980 liters/m^2·day to a wetted-area dosing rate of approximately 1170 liters/m^2·day (as calculated from outlet flow rates). The COD removal curve for this final sampling date is shown in Figure 6(a) and the COD removal curve for the two full-scale sampling events during lower dosing range are shown in Figure 6(b).

a) b)



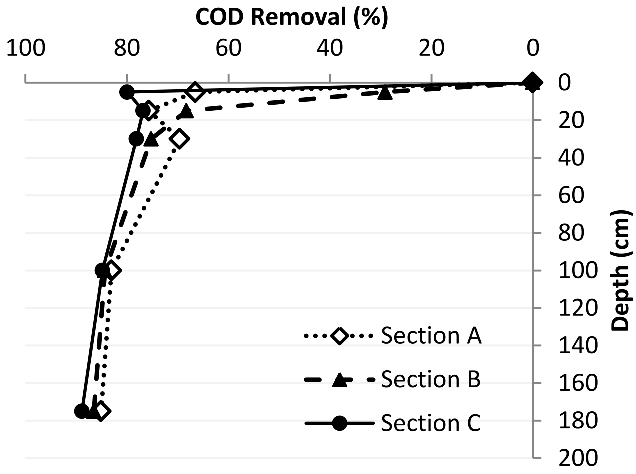


Figure 6 – a) COD removal 6 Nov. at higher dosing, and b) COD removal 25 Sept. & 15 Oct. at lower dosing range

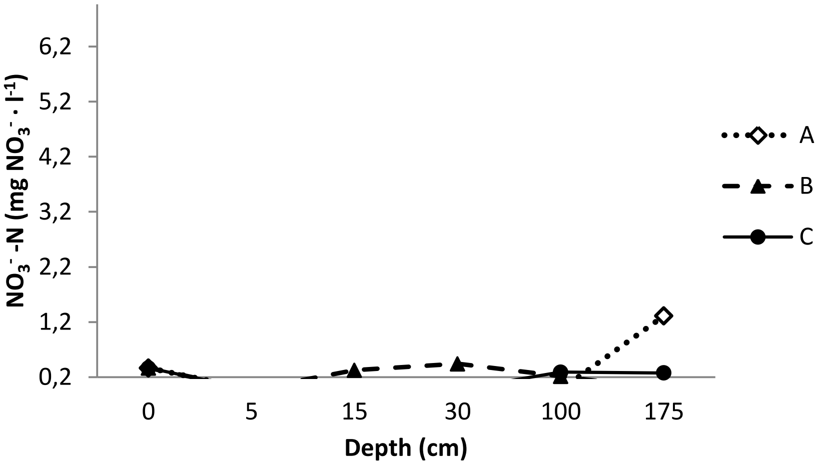
A comparison of these curves shows that the lower dosing range produces a steeper curve, with more of the removal taking place in the upper section of the filter. At the higher dosing range a greater percentage of removal takes place between the 30cm and 100cm depths than at the lower dosing. This phenomenon points to several possibilities regarding the role of depth in this filter. As the final sampling event was collected only three days after the increased dosing regimen, it is unlikely that the microbial community in the biofilm was fully adapted to the larger loading. Even with the sudden increase in hydraulic dose of nearly 20 percent, the removal percentages at the outlet of each section of the filter wall remained nearly constant. In this case the great depth implies a greater ability to buffer sudden fluctuations in dosing rates. This is a desirable attribute for applications of wastewater treatment with inherent variability, such as stormwater runoff.

Another possibility is that significant organic removal in the lower section of the filter (i.e. greater than 100cm) requires even higher dosing rates than those used in this study. From the comparison of the removal curves in Figure 6 it appears that the organic removal capacity at depths greater than 15cm was only activated at higher dosing. A gradual increase in hydraulic dosing to the system may reveal whether or not the deepest section of the filter can be activated in a similar manner.

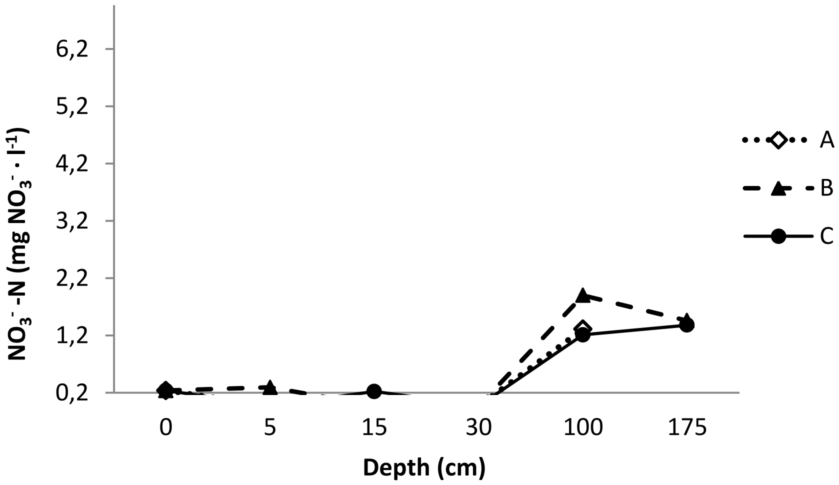
4.2.2 Nitrification

The development of nitrification by depth over the study period is shown in Figure 7.

a)



b)



c)

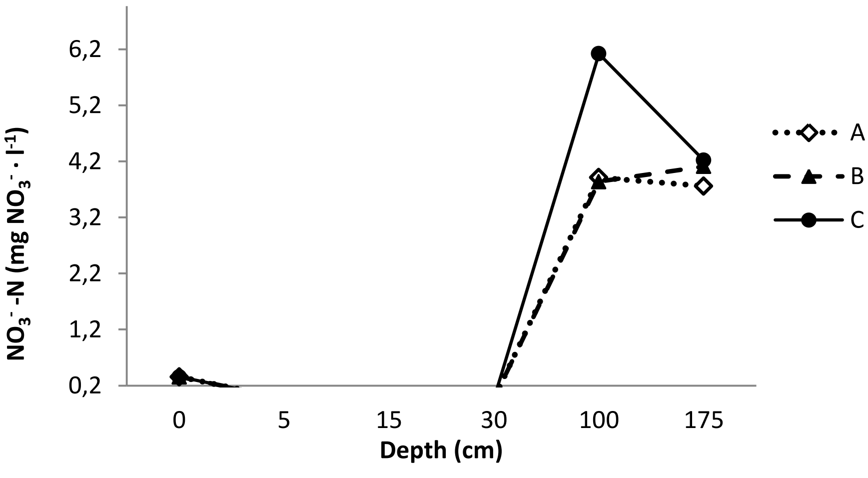


Figure 7 – Nitrate concentration over depth of filter on **a)** 25/09/12 **b)** 15/10/12 and **c)** 06/11/12

Clearly the nitrification process had not reached a steady state under the period of study, as shown by the large increase in nitrate levels during the final week of sampling. This is in line with the findings that nitrification can take on the order of 1-3 months for nitrifying bacteria to fully develop in intermittent media filters, with the higher end of the range for cold climate locations (Bahgat et al. 1999). This increase happened despite cold temperatures; the outlet temperatures measured in the final week were between 4 and 5.6 °C. Nitrifying bacteria have been reported to become nearly inactive below 5°C, and in fact Chen (2003) found that nitrification was completely inhibited in intermittent sand filters under the temperature of 15 °C. An investigation of the nitrification behavior in this system over the winter period would give better insight regarding the performance of the above-ground construction in colder climates.

The pattern of nitrification in relation to filter depth showed that increased nitrate levels did not appear before the 100cm depth sampling outlet for all three filter sections. This is a surprising finding and contrary to most studies of intermittent filtration, which describe the nitrification process as happening at the very surface of filters, and of importance to a depth of 20cm (VanCuyk et al. 2001; von Felde and Kunst 1997; Widrig et al. 1996; Schwager and Boller 1997). In Figure 5 from Schwager and Boller (1997), an increase in nitrate levels is seen to a depth of 20cm, but remains constant at greater depth.

The major factors affecting nitrification efficiency in biofilms include the load of organic matter, oxygen levels, and temperature (Føllesdal 2005; Boller et al. 1994a). Nitrification in biofilms only takes place with enough dissolved oxygen and after a substantial amount of the degradable organic fraction has been removed. Bahgat et al. (1999) reports that nitrifying bacteria (*Nitrobacter*) is a sensitive genus for oxygen concentrations, and claims this is the reasoning behind higher population counts at the filter surface than at greater depths. The development of nitrification in this system at deeper levels implies that oxygen concentrations were sufficient even at great depths to support nitrifying bacteria. An investigation of the dissolved oxygen gradient with depth inside the filter sections would be useful to support this theory.

It is likely that the organic matter was responsible for hindering nitrification in the upper sections of the filter. Although the pattern of organic removal was similar to the pattern reported by other researchers, the extremely high hydraulic load applied in this study also introduced a high organic load per unit surface area. With the average influent COD concentration of 241mg/l (see Table 1) and wetted surface area loading rate of 980 $l/m^2{\cdot}d$ the organic loading rate to the surface of the filters in this study was over 230g COD/ $m^2{\cdot}d$. For comparison, the organic loading rate to the surface of the experimental filter with removal curves shown in Figure 5 (Schwager and Boller 1997) was around 14g COD/ m^2·d (120mg COD/l septic tank effluent loaded at a rate of 120 l/m^2·d). This suggests that extremely high hydraulic and organic loading rates applied to single-pass intermittent media filters have a significant impact on the pattern of nitrification with filter depth. More specifically, the higher organic load may cause nitrification to be inhibited in the upper reaches of the filter, where previous studies have found most nitrification to occur (VanCuyk et al. 2001; von Felde and Kunst 1997; Widrig et al. 1996; Schwager and Boller 1997), requiring a greater filter depth to allow sufficient nitrification.

4.3 Containing wall material

4.3.1 Treatment implications

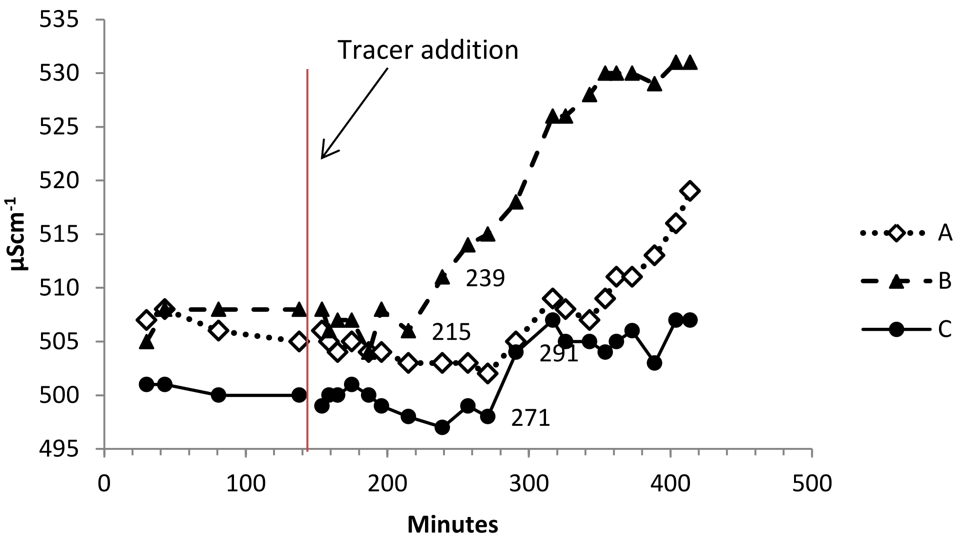
One-way analysis of variance (ANOVA) tests at 95% confidence interval were performed on the effluent values for each wall section to evaluate the influence of containing wall material. The results showed that Section C had a significantly lower mean COD and BOD_5 value than Section A (p value 0.018 and 0.013, respectively). While Section B had a lower mean COD and BOD_5 than Section A as shown in Table 1, the differences were not statistically significant. No significant differences were found for TSS, total phosphorus, total nitrogen, or pH (p values 0.398, 0.380, 0.910, and 0.137, respectively). There were too few trials of bacterial analysis to perform confident statistical comparisons, but as seen in Table 3, Section C showed greater removal of *E. coli* than Sections A and B in both trials.

Interpreting these results is difficult, as a complete separation of variables was not possible. It was believed that the more restrictive containing material used for Section A (plastic liner and plywood) versus the open material used for Sections B and C (geotextile grid) would create differing aeration patterns but this was not measured directly, for example with dissolved oxygen measurements inside the filters. Indirect indications of greater aeration are performance in terms of organic removal and nitrification— both aerobic processes with greater efficiency at higher oxygen levels. However, the removal efficiency can also be affected by confounding variables such as differing flow patterns or faulty construction.

The malfunction of the outlet at Section A meant that the saturated water level at the base of Section A was very slightly lower (2-3cm) than the other two wall sections. While this could potentially have an impact on effluent values any effect is believed to be negligible, and an investigation of the treatment over the depth of the filters shows that Section A exhibits lower removal throughout the filter, and not only at the outlet (see Fig. 4).

There is greater concern regarding confounding variables due to differences in flow patterns within the filters. The results from the NaCl tracer tests revealed that there are differences in the minimum retention times between the three filter sections. Figure 8 shows the results from the two trials.

a)



b)

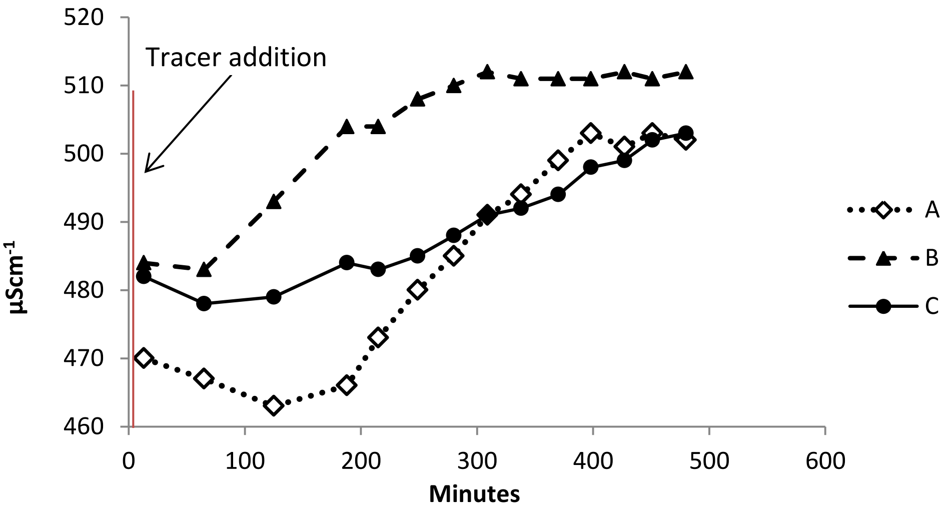


Figure 8 – Results from NaCl tracer tests performed on **a)** 09/11/12 and **b)** 13/11/12

From these graphs it appears that Section B shows signs of tracer appearance already after around 90 minutes from addition. Sections A and C take approximately 150-200 minutes to show signs of tracer breakthrough. The shorter minimum retention time in Section B may be due to short circuiting and preferential flow inside the filter bed. Shorter minimum retention times are correlated with reduced treatment efficiencies of both organics and bacterial indicators (Bancolé et al. 2003; Stevik et al. 1999b; Torrens et al. 2009a). It is tempting to assume that Section B, given an equal minimum retention time to that of Sections A and C, may have had slightly higher removal efficiencies. This would give a stronger statistical indication that the geotextile grid containing walls used in Sections B and C led to slightly better removal of organics than Section A's closed construction design. However, as only section C had statistically lower organic removal than Section A, a treatment effect of containing wall material cannot be stated with confidence.

There was no significant difference for total nitrogen removal between the three filter sections, and as the nitrification process had not reached a steady state under the period of investigation, a comparison of nitrate levels would be premature. However, as seen in Figure 7, all three sections exhibited increased nitrate levels beginning at a filter depth of 100cm. This is contrary to previous research which indicates that intermittent filters suffer from poor oxygenation along the depth gradient, which restricts aerobic bacterial activity to only the first few centimeters of the filter (Petitjean et al. 2011).

The containing material in Section A was designed to restrict contact with the atmosphere and aeration to a greater extent than the open design of Sections B and C. However, the treatment performance in terms of organic removal (BOD and COD) was only slightly less than Sections B and C, and in the case of Section B this difference was not statistically significant. Additionally the fact that Section A showed signs of nitrification even at great depth suggests that the containing material had little influence on the rate of aerobic bacterial processes.

In-depth examination regarding the impact of the "wall" design on aeration processes in intermittent filters is an interesting prospect for future research. It is possible that the material used to contain the filter media in this project had little effect on treatment, but the above-ground construction and the geometry of the filters may have played some role. Each filter section was constructed with only a 30cm thick filter media core sandwiched between 5cm layers of large-grained material as a capillary barrier. Although Section A was lined with plastic, it is possible that air flow occurred in the unsaturated capillary barrier regardless. A direct comparison of dissolved oxygen levels and treatment efficiencies for buried filters versus filters constructed above ground is one possibility. Additionally, an investigation of the impact of filter bed geometry is interesting, as the thickness of the filter bed may affect the longest pathway for oxygen to reach the very core of the filter.

4.3.2 Practical implications

Temperature monitoring of the effluent values during tracer tests showed that on 09/11/12 the temperature in section B was significantly highest (P=0.000), with average temperatures of 4.7, 5.5, and 4.6 for sections A, B, and C, respectively. This may be due to the shorter retention time in Section B allowing less exposure time for heat loss to the atmosphere (initial temperature in settling tank was 14°C). The higher temperature could also indicate that the placement of Section B in the middle allowed for greater buffering against temperature change than for the wall sections on the outsides.

On the second day of monitoring, 13/11/12, the system had been covered in a clear plastic sheet for several days to protect from frost, and Section A had significantly highest temperature (P=0.000) with temperatures of 6.2, 5.8, and 5.0 for A, B, and C, respectively. This indicates that the extra black colored plywood covering used on Section A may be better for holding solar radiative heat. The practical implication is that containing wall material choice impacts heat transfer characteristics. This has some impact on the temperature of the water inside the filter. If such a filter is designed to cover a significant portion of an outer wall on a building, this also impacts the heat transfer in the building.

Additional practical implications regarding containing wall material involve aesthetic considerations. A solid containing wall as used in Section A allows adjustment of color and material, such that the greywater filter can be disguised to blend into the appearance of the building it stands against. The open design of Sections B and C actually show the LECA filter material from the outside, which may or may not be desirable aesthetically.

Although Sections B and C were constructed with no solid barrier between the filter material and the atmosphere, there was no noticeable odor associated with the treatment wall structure. If this was monitored more closely and confirmed, the possibility for indoor installation could be considered.

4.4 Vegetation

4.4.1 Treatment implications

Statistical comparison of the effluent values for Sections B and C were used to evaluate the treatment implications of the vegetation and irrigation system built into Section C. While the mean effluent values for TSS, COD, BOD_5, total nitrogen and total phosphorus were all slightly lower for Section C compared to Section B (see Table 1), ANOVA test at 95% confidence interval determined that these differences were not significant. Additionally, the shorter breakthrough time exhibited by Section B (see Figure 8) is likely to influence the treatment capacity for Section B in a negative way. Therefore, the superior treatment in Section C cannot be attributed to the vegetation and irrigation system with confidence.

The two trials of bacterial analysis both showed that Section C had higher reduction of *E. coli* than Sections A and B (see Table 3). Studies have shown that bacterial removal in single-pass intermittent filters is strongly correlated to minimum and mean retention times in the filters (Bancolé et al. 2003; Stevik et al. 1999b; Torrens et al. 2009b). It is logical that Section B, which had the shortest minimum retention time, showed the lowest removal of *E. coli*. However, the minimum retention times in Sections A and C appeared to be similar, although Section C showed much greater reduction of *E. coli*.

According to the NaCl tracer tests, shown in Figure 8, Section C showed a more gradual increase in tracer concentration at the outlet of the filter than Sections A and B. This may indicate that Section C has a longer mean retention time, but the tracer tests were not conducted long enough to confirm this. The more gradual increase in concentration could be attributed to the displacement of the infiltrating water as the felt strip irrigation setup directs some water out of the core of the filter towards the plant roots and planting media.

The overall biomass of the vegetation during the course of this study was very small in comparison to the volume of water applied to the system. Although evidence of root growth could be seen, roots did not penetrate into the filter media. Especially due to the cooler fall and winter temperatures during the study period, the vegetation is not believed to have exerted any appreciable difference in transpiration of water or uptake of nutrients. Investigation into the treatment effects on such a system with much greater vegetation biomass and differing species of plant life is a large opportunity for further research.

4.4.2 Plant survival

Most of the plants survived throughout the period of study, to the end of November. This includes several nights of freezing in October, although the system was covered with a plastic sheet beginning in November. Some of the lettuce plants died over the course of the study, but there was no trend to survival according to location on the wall face vertically or horizontally. The marigold species all survived, although none flowered.

After several weeks a chalky, white substance appeared on the plant plug of one of the lettuce plants. This spread over the course of the study, appearing on several of the planting plugs as well as the surface of some of the LECA granules and the felt irrigation strips. An image of this is shown in Figure 9. The white substance never spread to plant leaves and did not appear to cause plant death, but identification by plant experts would determine the origin.

Figure 9 – Image of planter box, four weeks after system startup.

The watering system appeared to provide a sufficient amount of water to the roots of the plants, as there was no evidence of drying out of the Jiffy7 planting plugs. This should be tested also at warmer temperatures and higher transpiration rates, as the material type and surface area likely need to be optimized to vegetation water demand. The two different synthetic felt materials seemed to perform different functions for the plants. The thinner and more dense material (Jordan 1438010: 65% viscose, 25% polyester, and 10% PP) was better at transporting water from inside the filter to the base of the plant roots, while the thicker and less dense material (ZinCo SSM45: 100% PP) supported growth of plant roots inside the material, allowing the roots to spread inside the planter boxes. Evidence of root establishment inside the material can be seen in Figure 10 which is an image taken of the lining of the trays used to cultivate the seedlings.

Figure 10 – Root establishment inside synthetic felt material (ZinCo SSM45)

The main objective regarding vegetation in this research project was to determine effects on the quality of the treated effluent; therefore in-depth examinations of plant growth and survival were not included. Optimization of plant species with the nutrient, pH, and lighting characteristics of the system should be conducted by plant experts.

4.4.3 Practical implications

The incorporation of edible plant species (both lettuce and marigold) shows that wastewater treatment and agricultural production are possible to combine, even with a very small spatial footprint. The hygiene aspects of the plants were not examined in this study, but previous research suggests that most risk for microbial contamination of the plant surfaces occurs during application of irrigation water (WHO 2006; Finley et al. 2009; Ledin et al. 2001). This system supplies irrigation water directly to the roots, eliminating the risk for plant surfaces contacting the water, as well as for human contact during the irrigation process. Additional constituents of concern in greywater include heavy metals and pharmaceuticals, and uptake of these constituents by plants irrigated with greywater should be examined.

A practical benefit of the design of this wall system is the soil-less method of incorporating vegetation. In past studies the long term use of greywater for irrigation has shown harmful effects to native soils such as changes in pH and capillary rise and accumulation of heavy metals (Misra et al. 2010; Wiel-Shafran et al. 2006). This system puts greywater to use for a beneficial purpose of aesthetics and agricultural production, while avoiding the potential to pollute native soils.

The inclusion of vegetation into the filter also introduces an element of upkeep to the system. With die-off or harvesting, the plants would require replacement, and the ease of replacement must be considered, especially regarding the extent of root growth into the filter media. The inclusion of plants also increases the likelihood for public contact with the greywater if members of the public, especially children, feel inclined to touch the vegetation.

4.5 General practical observations

One goal for this study was to assess the general practical implications of this experimental vegetated wall wastewater filter.

The exposed nature of the filter wall system is vulnerable to outside temperature. In late October the overnight temperature in Ås, Norway reached -8°C which caused freezing of the distribution pipes and filter outlets. Insulated or heated pipes might be necessary for such a system in cold regions, and internal filter temperatures should be monitored over the duration of winter to establish the vulnerability to freezing. The effluent temperature was monitored in early November during the tracer testing, and it was observed that the initial temperature in the settling tank effluent of 14°C dropped to a minimum temperature of 4.0°C at one of the outlets. Steps to insulate the system may be desirable to guard against such a large heat loss in winter. However, in warm summer temperatures heat loss associated with evapotranspiration may be desirable due to the urban heat island phenomenon. The winter period is also a concern for the survival of the plant life due to freezing and lack of sunlight. The design for this system allows for relatively easy replacement of dead plants, but this may depend on the root penetration of the filter media.

The lack of excavation work associated with an above-ground filter structure is a benefit. This system was built on asphalt in a parking lot using only brief assistance with a forklift to raise the walls after filling. Another attribute to the system was the extremely small space requirement. The total area used was $1.2m^2$ for all three sections, but some of this area was used to create a larger saturated basin at the bottom of the structure for stability. This could potentially be removed if measures were taken to secure the structure to the building, for example. Then the total area would be about 0.8 square meters, treating a total of 360 liters per day. This space is immediately adjacent to the building, which in many circumstances is not in use anyway.

The intermittent media filter wall used only a very small spatial footprint, which is an attractive attribute for development in more urbanized regions. The possibility to extend the capacity of the filter wall by building upwards is an unanswered question. The wall in this study was two meters tall, which allowed easy access to the nozzles and dosing surface for maintenance and monitoring purposes. A taller structure would make access to the dosing surface more difficult, and maintenance work on plants at great heights would also need to be considered.

Most wastewater treatment systems are of the "out of sight, out of mind" variety, so public perception was an important theme regarding this very visible wastewater treatment structure. A very slight noise could be heard during the spray dosage every half hour. There was more noise associated with the pump inside the settling tank, but this would not be an issue with a normal underground septic tank. The cover over the spray nozzles and the NR 10-20mm Filtralite capillary barrier successfully eliminated public exposure to the untreated greywater. The only access points were at the outlets to the filter sections, and near the plants roots inside the planter shelves on Section C. The leakage in Section A introduced some water onto the pavement in the area where the system was constructed. However, this also indicates that above-ground construction makes malfunctions very obvious to the operator, and steps can be taken to fix the issue immediately.

Possibly the greatest success associated with this project was the positive overall public response to the treatment wall. There was much curiosity amongst students and faculty on the UMB campus, who seemed especially charmed by the link between wastewater and plant cultivation. This is in opposition to the flush-and-forget mentality which pervades the conventional wastewater treatment sector. The creation of a visible link between the wastewater produced in the home and living organisms which depend on that wastewater for survival may also serve as an incentive to preserve the quality of wastewater we release into the environment on a daily basis.

5. CONCLUSION

The main objective of this project was to evaluate the greywater treatment performance of an intermittent media filter constructed in the form of a wall. Additional goals were to evaluate the treatment pattern over the depth of the wall, examine the treatment effect of permeable containing walls, and examine the treatment effect of the vegetation/irrigation system applied.

The intermittent filter wall design showed good overall treatment performance despite a very large hydraulic loading. Average removal of solids (TSS) and organic compounds (BOD_5 and COD) was very high. Removal of nutrients in the form of total nitrogen and total phosphorus was somewhat lower but within the range expected from intermittent media greywater filters used in Norway (Norsk Rørsenter 2006). Removal of nutrients can also be improved, if necessary, with recirculation of the effluent and incorporation of specially engineered phosphorus sorbing media. The issue of pathogenic microorganisms should be investigated further, as the results from the very limited analysis of *E. coli* indicate that despite a 2 log unit reduction, the effluent concentrations are somewhat higher than the limit suggested by the World Health Organization for unrestricted irrigation with greywater (WHO 2006). The satisfactory treatment performance of organics and nutrients, despite high hydraulic loading and small spatial footprint, makes this design attractive for urban development.

Examination of removal patterns over the depth of the filter showed that the majority of organic removal (COD) takes place in the upper 15cm of the filter. However, there is some evidence that a sudden increase in hydraulic loading triggered greater removal of organics at depths over 15cm. Further investigation into removal of organics with depth at even larger hydraulic loading would reveal whether or not the deepest sections of the filter can be activated for organic removal.

The nitrification pattern over the depth of the filter showed that an increase in nitrate levels did not appear until the 100cm depth sampling outlet. This is most likely due to suppression of nitrification at the surface of the filter caused by high organic loading. The implication is that filters with high organic loading can compensate for the lost nitrification at the surface through greater filter depth.

The use of geotextile grid as a permeable containing wall showed little effect on treatment performance. Slightly greater average removal of organics was achieved by filter sections using the geotextile material, but confounding variables such as outlet malfunctions and differences in minimum retention times reduce the confidence in this finding.

The incorporation of lettuce and marigold plants onto a section of the filter did not produce significant effects on treatment performance. The planted filter section produced a differing pattern of recovery during a NaCl tracer test. This may be related to the displacement of water inside the filter due to the irrigation system. Any possible treatment effect of vegetation may be reliant on season, growth stage, and plant species, so this investigation was not considered an intensive study with regard to plant inclusion.

The findings in this study are based on a relatively small number of sample periods, all of which may be classified under the startup period for the treatment system. An extended study, covering all seasons throughout the year, would provide more information regarding the actual treatment capabilities. More thorough investigations of aeration patterns using dissolved oxygen measurements and gas tracer studies would be extremely useful to characterize the effect of the wall filter design. An intensive investigation and optimization of plant growth and survival is an opportunity for collaboration between water treatment experts and plant ecology experts.

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