NELSON MIGUEL GUERREIRO LOURENÇO

OPTIMIZATION OF CONTINUOUS VERMIFILTRATION PROCESSES FOR SMALL COMMUNITIES

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Declaração de autoria de trabalho

OPTIMIZATION OF CONTINUOUS VERMIFILTRATION PROCESSES FOR SMALL COMMUNITIES

Declaro ser o autor deste trabalho, que é original e inédito. Autores e trabalhos consultados encontram-se devidamente citados no texto constando da listagem de referências incluída.

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Para todos aqueles que de alguma forma, já não puderam ver este meu trabalho.

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RESUMO ALARGADO

Esta tese foca-se no estudo da optimização de um processo descentralizado de vermifiltração para tratamento de água residual urbana. O processo é avaliado tanto sob o ponto de vista técnico e ambiental numa perspectiva de sustentabilidade, no qual se incluem a optimização de um sistema de vermifiltração em pequena-escala, complementado com estudos de análise de ciclo de vida.

As tecnologias de tratamento de água residual incluem sistemas centralizados, mais comuns em zonas urbanas, e sistemas descentralizados, mais comuns em aglomerados populacionais dispersos e pequenas comunidades rurais. Os sistemas descentralizados têm vindo progressivamente a ser considerados como soluções mais sustentáveis. Muitos não requerem fornecimento de electricidade, operação dispendiosa ou sofisticada, sendo de fácil adaptação em diferentes contextos geográficos. Os sistemas secos controlados são aconselhados para regiões áridas e aglomerados dispersos sem sistemas centralizados de saneamento. Dado que diversas tecnologias descentralizadas secas e húmidas poderão ser importantes fontes de contaminação, encontram-se em aberto diversas oportunidades para investigação, incluindo, não apenas a conversão de sistemas rudimentares em sistemas controlados nas tecnologias secas, como também a inclusão de sistemas de tratamento secundário como completamento aos sistemas de tratamento primário nas tecnologias húmidas.

A vermifiltração combina filtração com vermicompostagem, tratando-se de uma tecnologia sustentável e de baixo custo que apresenta elevadas eficiências de tratamento, mesmo quando sujeita a reduzidos caudais de água residual. Tem sido aplicada com sucesso em habitações, pequenas ETAR, tanto para águas residuais urbanas como industriais.

Numa primeira fase, procedeu-se à optimização do sistema de vermifiltração. O procedimento envolveu a identificação das melhores variáveis hidráulicas, densidade de minhocas, e configuração do sistema, na avaliação de um sistema unitário e um sistema sequencial. As eficiências óptimas de tratamento foram obtidas para um tempo de retenção hidráulico de 6 horas, um caudal hidráulico de 0.89 m^3 m^{-2} dia^{-1}, e 177.6 g CBO m^{-2} dia^{-1} de taxa de carga orgânica. As melhores eficiências foram obtidas para uma densidade de minhocas de 20 g L^{-1} tendo sido atingidos valores para CBO_5, CQO, SST e NH_4^+ de 97.5%, 74.3%, 98.2% e 88.1%, respectivamente. O sistema sequencial

permitiu o aumento significativo das eficiências de tratamento quando comparado com o sistema unitário, tendo-se obtido eficiências de tratamento de 98.5% for CBO_5, 74.3% for tCOD, 96.6% for TSS, and 99.1% for and NH_4^+.

Posteriormente, de modo a avaliar o meio de enchimento mais adequado, duas alternativas foram testadas, especificamente vermicomposto e serradura. As eficiências de tratamento foram de 91.3% para CBO_5, 87.6% para CQO, 98.4% para SST, e 76.5% para NH_4^+ em VE, e 90.5% para CBO_5, 79.7% para CQO, 98.4% para SST, e 63.4% para NH_4^+ em SE. As minhocas contribuíram para reduzir as remoções de NH_4^+ e Azoto Total, e para aumentar a concentração de NO_3^- no efluente tratado. Comparativamente à água residual bruta, o vermicomposto contribuiu para aumentar a concentração de Fósforo Total. Em todos os tratamentos, as eficiências foram ainda insuficientes para cumprir a regulamentação da EU para descargas de água residual em meios aquáticos sensíveis (Azoto e Fósforo Totais) e as orientações da USEPA e OMS para irrigação (coliformes fecais). Ainda assim, todos os tratamentos removeram ovos de helmintes.

Desenvolveu-se uma análise de ciclo de vida por forma a comparar os sistemas de vermifiltração com outras tecnologias alternativas. Este procedimento incluiu um sistema de filtração lenta, um leito de macrófitas e um sistema de lamas activadas. O inventário de ciclo de vida permitiu identificar que os recursos materiais foram mais usados durante a fase de construção comparativamente a qualquer outra fase. Dadas as pequenas populações servidas, a quantidade de materiais usados por habitante foi mais elevada relativamente às quantidades encontradas para outras infraestruturas de maior dimensão. A electricidade foi o recurso mais utilizado durante a fase de operação, tendo sido um resultado expectável. Quando comparada com os leitos de macrófitas, a vermifiltração permitiu obter importantes benefícios ambientais na maioria das categorias de impacte, em particular durante a fase de construção. Comparativamente à filtração lenta, a vermifiltração originou a melhoria das categorias de impacte acidificação e eutrofização, ao mesmo tempo que originou a deterioração das restantes. A vermifiltração pode apresentar-se como uma melhor solução sob o ponto de vista ambiental que os sistemas de leitos de macrófitas e lamas activadas, fruto dos melhores resultados obtidos na maioria das categorias de impacte. Em todas as soluções de tratamento os impactes durante a fase de construção ultrapassaram os impactes das restantes fases, devido ao pequeno número de habitantes servidos, não atingindo economias de escala.

Durante a elaboração desta tese, diversas questões ficaram por responder, nomeadamente: i) o impacte das variáveis climáticas na dinâmica do tratamento e das eficiências, e que afectam o rendimento e a resiliência do processo segundo diferentes inputs de carga orgânica e temperaturas; qual o tipo de meio de enchimento mais adequado para o processo, em particular o que melhor contribui para a economia circular; e iii) a aplicabilidade, tanto do ponto de vista técnico como legal, do vermicomposto no solo, como fertilizante, aquando e futuras análises de ciclo de vida, deve ser incluída nas fronteiras do sistema.

Palavras-chave: vermifiltração, água residual, tratamento de efluentes, sustentabilidade ambiental, ciclo de vida.

ABSTRACT

This thesis studies the optimization of decentralized vermifiltration processes to treat urban wastewater. The process is evaluated on both technical and environmental sustainability perspectives, which included the optimization of small-scale vermifiltration processes, complemented with life cycle assessment studies.

Wastewater treatment technologies include conventional centralized systems, typically used on urban areas, and decentralized systems, more common in sparse dwellings and small communities of rural areas. Decentralized systems are being progressively considered as more sustainable solutions. Many do not require electricity supply, expensive or sophisticated operation, and are easy to adapt in different geographic contexts. Dry controlled systems are suited for arid regions and for disperse dwellings without centralized sanitation. Since several dry and wet decentralized technologies may be important sources of contamination, many opportunities are still open for research, including, not only the conversion of rudimentary systems into controlled systems in dry technologies, but also the inclusion of secondary treatment systems in a complement to primary treatment systems in wet technologies.

Vermifiltration combines filtration with vermicomposting, being a low-cost and sustainable secondary treatment technology with high efficiencies even with small wastewater flows. It has been successfully applied in households, small WWTPs, and on both urban and in some industrial wastewaters.

The optimization of the vermifiltration system was conducted first. The procedure evolved finding the best set of hydraulic variables, earthworm stocking density, and system configuration, when evaluating a single-stage reactor and a four-stage reactor. The optimal treatment efficiencies were obtained for 6 hours of hydraulic retention time, 0.89 m^3 m^{-2} day^{-1} of hydraulic loading rate, and 177.6 g BOD_5 m^{-2} day^{-1} of organic loading rate. An earthworm stocking density of 20 g L^{-1} proved to be the best optimal condition with treatment efficiencies for BOD_5, tCOD, TSS and NH_4^+ of 97.5%, 74.3%, 98.2% and 88.1%, respectively. Four-stage reactor significantly improved treatment efficiencies when compared to single stage reactor with efficiencies reaching 98.5% for BOD_5, 74.3% for tCOD, 96.6% for TSS, and 99.1% for and NH_4^+.

Further, in order to evaluate the best vermifilter packing, two alternatives were assessed, namely vermicompost and sawdust. Treatment efficiencies were 91.3% for

BOD_5, 87.6% for COD, 98.4% for TSS, and 76.5% for NH_4^+ in VE, and 90.5% for BOD_5, 79.7% for COD, 98.4% for TSS, and 63.4% for NH_4^+ in SE. Earthworms contributed to reduce NH_4^+ and TN removal, and to increase NO_3^- concentration in treated effluent. Comparing with raw wastewater, vermicompost contributed to increase TP concentration. In all treatments, efficiencies were still short to attain the EU regulation for wastewater discharges discharges in sensitive water bodies (TN and TP) and USEPA and WHO guidelines for irrigation (faecal coliforms). Even so, all treatments fully eliminated helminth eggs.

A full life cycle assessment study was made to benchmark vermifiltration systems against other technical alternatives. This included slow rate filtration, constructed wetland, and activated sludge. The lifecycle inventory showed that more material resources are used during construction than in any other phase. Given the small served population used, the material intensity per user was higher than that found in other larger facilities. Electricity was the resource more used during operation, which was an expectable result. Vermifiltration when compared with constructed wetlands brought important environmental benefits in most impact categories, in particular during the construction phase. Compared to slow rate filtration, vermifiltration resulted in the improvement of impact categories acidification and eutrophication, but in the deterioration of the remaining. Vermifiltration would be a better environmental solution than constructed wetlands and activated sludge, as shown by better results in most impact categories. In all treatment solutions the impacts during construction outweigh those of the other phases, due to the small number of served inhabitants, not attaining economies of scale.

During the making of this thesis several research questions were left unanswered, namely: i) the impact of climatic variables in the treatments' dynamic and efficiencies, affecting performance and resilience of vermifiltration at various input load rates and temperatures; ii) which filter packing is best suited for vermifiltration, in particular contributing to the circular economy; and iii) the technical and legal applicability of the vermicompost to soil, as fertilizer, should be included in the boundaries of future life cycle assessment.

Keywords: Vermifiltration, wastewater, optimization, life cycle assessment.

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Abbreviations

APY	Aqua-privy
BOD	Biochemical Oxygen Demand
C:N	Carbon to Nitrogen ratio
COD	Chemical Oxygen Demand
CT	Composting toilets
CW	Constructed wetlands
DCS	Dry controlled systems
DHP	Dehydrating pits
DP	Dry pits
DRS	Dry rudimentary systems
DT	Dry technologies
EU	European Union
FA	Fossa alterna
fp-WSP	Facultative pond - wastewater stabilization ponds
fws-CW	Free water surface constructed wetland
GHG	Greenhouse gases
HLR	Hydraulic loading rate
HRT	Hydraulic retention time
hssf-CW	Horizontal subsurface flow constructed wetland
int-SF	Intermitent sand filters
mp-WSP	Maturation ponds
MPN	Most Probable Number
OLR	Organic loading rate
PBF	Packet bed filters
p.e.	Populational Equivalent

PFL Pour flush latrine

PT Peat filters

PTS Primary treatment systems

rec-SF Recirculating sand filters

RS Dry rudimentary systems

SF Sand filters

SKW Soakaways

SP Single pits

SS Septic systems

ST Septic tanks

STS Secondary treatment systems

SWIS Subsurface wastewater infiltration systems

TN Total Nitrogen

TP Total Phosphorus

TSS Total Suspended Solids

UN United Nations

US United States

VCT Vermicomposting toilets

VCFT Vermicomposting flushing toilets

VF Vermifilters

VIP-dry Dry ventilated improved pit

VIP-wet Wet ventilated improved pit

vssf-CW Vertical subsurface flow constructed wetland

vssf-CW+hssf-CW Vertical subsurface flow constructed wetland and horizontal subsurface flow

W:D Wet to dry ratio

WP	Wet pits
WSP	Wastewater stabilization pounds
WT	Wet technologies
WWTP	Wastewater Treatment Plant

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Chapter 1
General Introduction

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1.1. Work justification

Due to increasing world population, natural resources are under increasing pressure which fosters wastewater reuse planning and emphasizes on the decentralized sanitation, especially in rural areas where high wastewater collection and treatment costs does not justify the installation of conventional solutions (Prasad and Kumar, 2012).

Demand for basic wastewater treatment has been increasing over the last decades due to population growth and current socio-economic development. Since 1990, wastewater treatment coverage has increased by 20% in most developing regions, and this track is expected to continue, following United Nations' very ambitious Sustainable Development Goal of achieving access to adequate and equitable wastewater treatment and hygiene for all by 2030 (United Nations, 2020). Currently, thousands of million people still lack access to proper wastewater treatment, and over 80% of the world's wastewater, near 2.0 million tons annually, is discharged into the environment without treatment, threatening ecosystems and human health (Massoud et al., 2009).

The foremost wastewater treatment goal is protection of public health through control of pathogens in order to prevent transmission of water-borne diseases and eutrophication of surface water (Pundsack et al. 2001; Zhang et al., 2008).

There is recent interest in environmentally safe and economically viable small-scale wastewater treatment technologies for onsite wastewater treatment. Such interest is reflected in the number of articles published about decentralized wastewater treatment technologies, which has been growing steadily since the mid-90s (Figure 1.1); as well as by the endorsement of international organizations, such as UNESCO (2017) and OECD (2013).

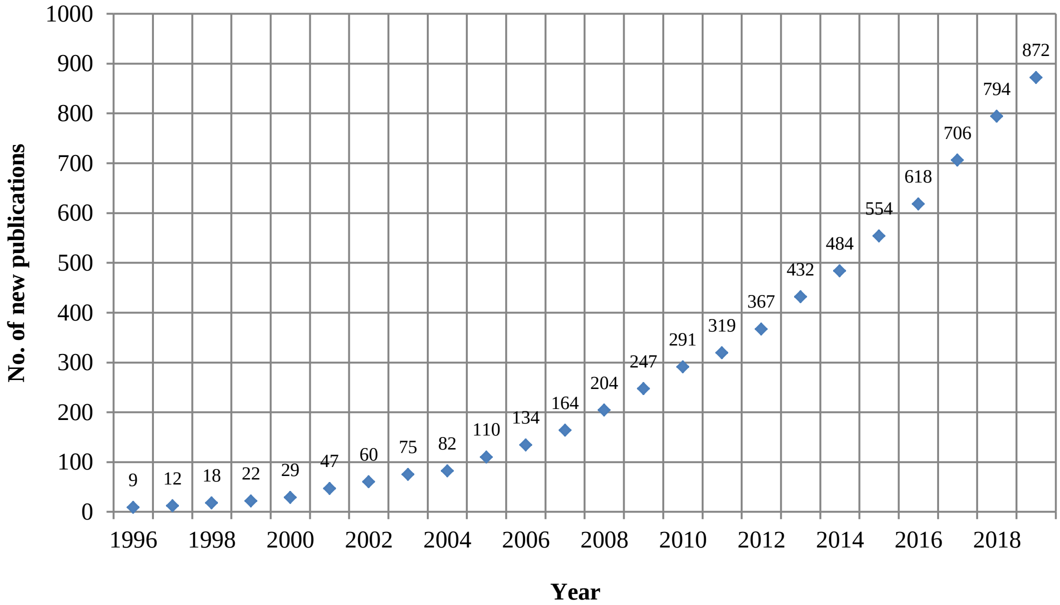

Figure 1.1 – New publications about decentralized wastewater treatment technologies in ISI journals in the period 1995-2020 (Web of Science, Clarivate Analytics® 2020).

Decentralized Wastewater Treatment Plants (WWTP) involve the collection, treatment, disposal and reuse of wastewater from households, clusters of homes and isolated communities, at or near the point of generation (Li *et al*., 2009). Previous studies have already focused on decentralized wastewater treatment technologies, including dry wastewater treatment (Hill and Baldwin, 2012), composting toilets (Anand and Apul, 2014), septic systems (Tchobanoglous and Crites, 2003), sand filters (USEPA, 2002), peat filters (Corley et al., 2006), vermifilters (Samal, 2018), constructed wetlands (Vymazal, 2011), or wastewater stabilization ponds (Alexiou and Mara, 2003), and also on comparing dry and wet technologies (Tilley et al., 2014). However, only a few have discussed specific technological requirements, efficiencies, costs, advantages for households and small clusters, and global environmental impacts.

According to the United Nations (UN) World Commission on Environment and Development, environmental sustainability is about acting in a way that ensures future generations have the natural resources available to live an equal, if not better, way of life as current generations (United Nations, 2020). Environmental sustainability is defined as responsible interaction with the environment to avoid depletion or degradation of natural resources and allow for long-term environmental quality. The practice of environmental sustainability helps to ensure that the needs of today's population are met without jeopardizing the ability of future generations to meet their needs (Study.com, 2020).

1.2. Vermifiltration

Vermifiltration is a low-cost wastewater treatment process which is based on the same oxidation reactions, biodegradation and microbial stimulation by enzymatic action also present in vermicomposting and in trickling filters. It has been described as a viable alternative to treated domestic wastewater in small clusters with good applicability in developing countries and in remote locations.

In vermifiltration, dissolved and suspended organic and inorganic solids are trapped by adsorption and stabilization through complex biodegradation processes that take place in the filter packing, being subsequently used by microorganisms (Sinha et al., 2008). Earthworms and microorganisms cooperate to ingest and biodegrade organic wastes and contaminants present in wastewater. Their action improve filter permeability, increasing the degradation of the organic matter (Sinha et al., 2008; Arora et al., 2014a), hence promoting high removal efficiencies of BOD_5, COD and TSS from wastewater (Sinha et al., 2008). Earthworms's mechanical action creates aerobic conditions inside the reactor which help prevent the formation of odors.

Applications of vermifiltration include small pilot-scale tests, households and small WWTP, opening new opportunities for treating domestic, urban wastewater and some industrial wastewater due to the low cost and sustainable nature (Sinha *et al*., 2008). It is also accepted that VF can be more cost-effective when compared with conventional WWTP (Samal et al., 2017).

1.3. Aims and outline

This thesis presents results from studies on vermifiltration as a nature based solution for small decentralized WWTP. The purpose of the research was to support the leveraging of technical activities of a private company, which supported the study both materially and with dedication time.

The technology was evaluated technical and environmentally, in pursue of the following research questions:

a) Are (dry and wet) decentralized WWTPs environmentally safe, socially acceptable in small rural communities?
b) What are the most important parameters for sizing the vermifiltration process, and during its operation?
c) What is the importance of filter packing in vermifiltration to treat domestic and urban wastewater?
d) What are the main environmental impacts of vermifiltration when compared to other technologies, in a context of small communities?

The thesis is divided into seven chapters, including a general introductory chapter (Chapter 1), State of the Art (Chapter 2), three main chapters (Chapter 3 to Chapter 5), and a final concluding chapter (Final Remarks). Supplemental materials associated with the main chapters are compiled at the end of the document (Appendix A). The following chapters are outlined below.

Chapters 2 to 5 reflect the content of scientific articles published in international journals with refereeing.

Chapter 2: State of the Art.

This Chapter includes the review the current available decentralized wastewater treatment technologies, comparing dry and wet solutions, discussing their operational requirements, applicability, effluent quality, efficiencies, environmental impacts, costs, challenges, and their advantages and implementation difficulties.

Related article:

Lourenço, N., Nunes, L. M. (2020). Review of dry and wet decentralized sanitation technologies for rural areas: applicability, challenges and opportunities. *Environmental Management*, **65**, pp. 642-664. https://doi.org/10.1007/s00267-020-01268-7.

Chapter 3: Optimization of a vermifiltration process for treating urban wastewater.

This Chapter focuses on the optimization of the hydraulic variables and earthworm abundance of a small-scale vermifiltration process. Parameters include the hydraulic retention time, the hydraulic loading rate, and recirculation ratio, organic loading rate, earthworm abundance and reactor type. The study included the evaluation of single, two-stage, and four-stage vermifilters.

Related article:

Lourenço, N., Nunes, L. M. (2017). Optimization of a vermifiltration process for treating urban wastewater, *Ecological Engineering*, **100**, pp. 138-146. https://doi.org/10.1016/j.ecoleng.2016.11.074.

Chapter 4: The role of filter packing in vermifiltration.

This Chapter focuses on the evaluation of the performance of a small-scale vermifiltration for the treatment of urban wastewater, studying sawdust and vermicompost as organic filter packing materials. This process comprises only a single stage vermifilter.

Related article:

Lourenço, N., Nunes, L. M. (2017). Is filter packing important in a small-scale vermifiltration process of urban wastewater? *International Journal of Environmental Science and Technology*, **14**(11), pp. 2411-2422. https://doi.org/10.1007/s13762-017-1323-1.

Chapter 5: Life cycle assessment to compare nature-based solutions for wastewater treatment in small communities.

This Chapter shows results of a life cycle inventory and impact assessment used to compare vermifiltration against other unit processes in three small WWTPs.

Related article:

Lourenço, N., Nunes, L. M. (2020). Life cycle assessment to compare nature-based solutions for wastewater treatment in small communities, under review.

Chapter 6: Final remarks.

This Chapter includes a discussion of the results from the preceding chapters and adds some recommendations for future research.

Chapter 2
State of the Art

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2.1. Introduction

Wastewater treatment technologies vary from conventional centralized systems, typically used on urban areas (Tchobanoglous and Crites, 2003), to decentralized systems, more common in sparse dwellings and small communities of rural areas (Tilley et al., 2014). Centralized wastewater treatment involves the transportation of wastewater and excreta from the site of production to the site of treatment, usually far away from the place of treatment (Zeeman et al., 2002). Such systems are expensive to build and to operate, consuming considerable amounts of energy, thus, being inappropriate for small communities resulting in unsustainable long-term costs for the population. Decentralized wastewater treatment (or on-site sanitation) may be defined as the treatment and disposal, at or near the source, of relatively small amounts of wastewater and excreta, originating from single households or small dwellings, not served by a central sewer system (Capodaglio, 2017). Is a common choice in areas not covered by centralized wastewater treatment and is being progressively considered as more sustainable technology with a prediction to serve 5 thousand million by 2030 (Strande, 2014), being relatively easy to manage and integrate (Zeeman et al., 2002). As a promising alternative to conventional wastewater treatment, decentralized alternatives work as a closed loop, in which human metabolites are considered valuable resources. Is has been gaining attention in the EU, where 23% of households are estimated to use a decentralized wastewater treatment technology (EEA, 2013) and, in the US where near 25% of the population has been served by decentralized wastewater treatment for over a decade (Capodaglio, 2017). It is also prevalent in other developed countries (Schaider et al. 2017) and these numbers are expected to rise in the future (Somlai-Haase et al., 2017).

Decentralized wastewater treatment technologies vary, in increasing level of complexity, from: dry technologies (DT), including i) dry rudimentary systems (RS) and ii) dry controlled systems (DCS); and wet technologies (WT), including iii) primary treatment systems and iv) secondary treatment systems (Figure 2.1). The technologies most used are, by far, the DT as an alternative to WT (Scott, 2002).

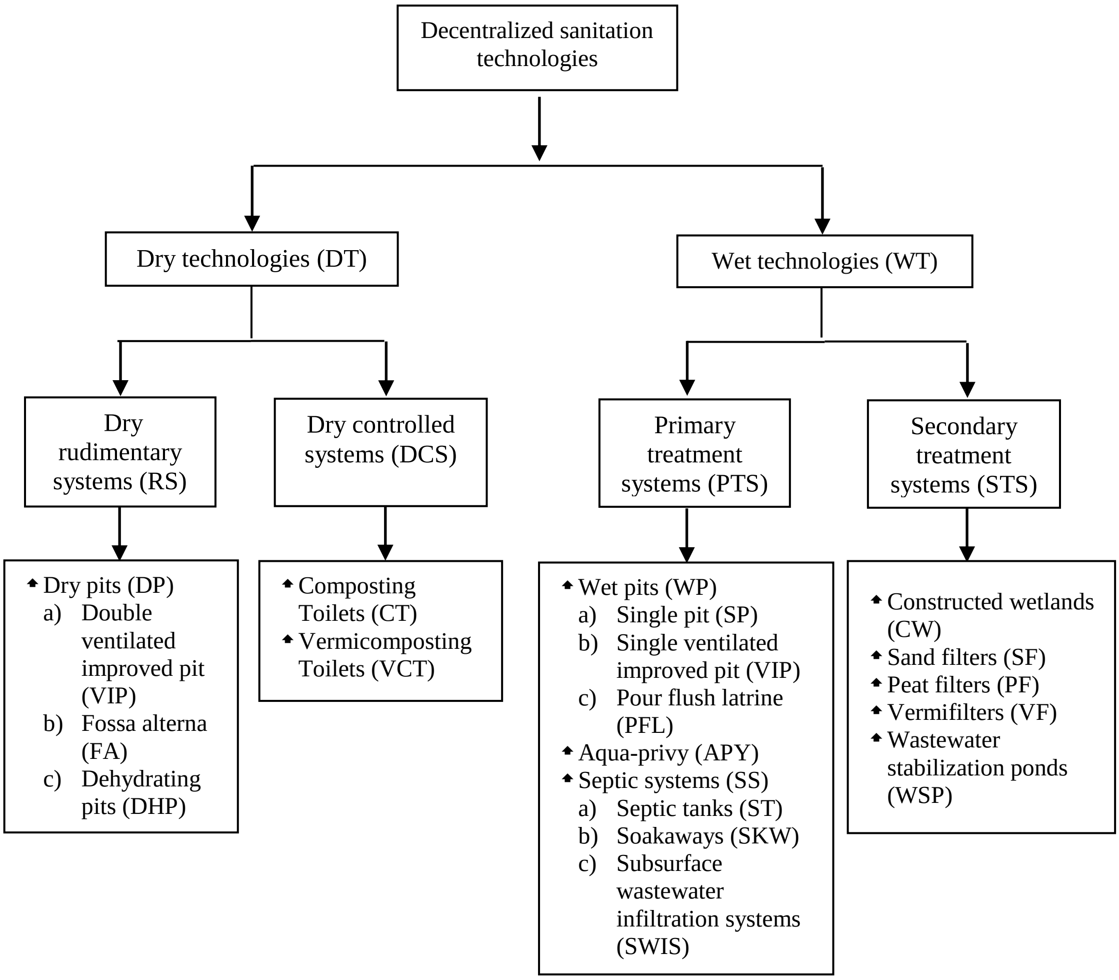

Figure 2.1 – Types of decentralized wastewater treatment technologies.

2.2. Dry technologies

A broad variety of dry wastewater treatment technologies (DT) is currently available, being particularly suited for arid regions and/or regions without central water supply or sewerage systems (Berger, 2011), in both developed and developing countries (Kaczala, 2006). DT are further divided into dry rudimentary systems (DRS), the most common form of wastewater treatment in developing countries, and dry controlled systems (DCS).

2.2.1. Dry rudimentary systems

The DRS discussed in this section include only the dry pits (DP) which are divided into i) single or double dry ventilated improve pit (VIP-dry) for collection, storage and partial treatment of excreta, producing pit humus; ii) fossa alterna (FA) or double vaults, for collection, storage and treatment of excreta and other organic wastes, producing pit humus; and iii) Dehydrating pits (DHP) (with one or two dehydration vaults), for collection, storage, and dehydration of excreta, producing dry feces (Tilley et al., 2014).

DP are excavated structures, either lined or unlined (both are also known as vaults (Berger, 2011)). Pit depth will depend on soil physical properties and groundwater depth. When full, pits are emptied, often by hand, or are abandoned. A toilet or another type of pedestal may be fitted over the hole and a structure may be built for privacy (Flores, 2010). Together, DP and wet pits (discussed later) are currently used by 1.77 thousand million people (Reid et al., 2014) on households and for communal service in rural and periurban areas of Northern, Central and Southern Africa (WHO/UNICEF, 2017), as well as in some countries of South America and Eastern Europe (von Muench and Rieck, 2011).

In DP, urine and water percolate into the soil through the bottom and side walls, while aerobic degradation occurs at the surface (Buckley et al., 2008), and anaerobic below the surface (Chaggu, 2004). Pits can be either single - for up to 20 users, or communal - for more users (Hansch, 2003).

In DHP urine-diverting can be present. Unlike VIP-dry, FA avoid the contamination of the end-product with fresh feces (Berger, 2011). Double VIP-dry are more appropriate than single VIP-dry for denser, periurban areas since allows it to be used continuously permitting a safer and easier emptying (Tilley et al., 2014). Examples are the Vietnamese double vault toilet, which has been used widely since 1954, with adaptations in Mexico and Guatemala (Scott, 2002).

In FA degradation must occur at a minimum of 20 °C for a period of one to two years to assure elimination of pathogens (WHO, 2006) (Table 2.1). Degradation is performed by the addition of lime, ash or earth into the storage chamber after each use (Kaczala, 2006).

DP do not provide enough treatment efficiency for safe disposal, therefore requiring an adequate subsequent treatment, particularly if the materials are used for fertilization. Some simple treatments have proven to be sufficient for the elimination of viruses and fecal coliforms, as, e.g., by solar exposure at a pH between 9 and 11 (Stenström et al., 2011); or by mixing wood ashes with feces at a ratio 1:3 under a pH between 9 and 10

(Schönning and Stenström, 2004). Unpleasant odors may nonetheless be present, along with insects and other disease vectors.

Sizing of DP is made considering a desirable emptying period (usually 1 to 2 years, depending on climate type), an average number of users (usually 5 users per system to 6 users per system), a plan area (generally circular or rectangular) (m^2) (Tilley et al., 2014), and estimating the average production of faeces and urine per user per day (128 g wet weight and 0.6 to 2.6 L, respectively) (Rose et al., 2015). Construction is difficult or impractical in rocky places. DP have a low installation cost, require little maintenance, and use few or no water to work (Reid et al., 2014). Well-constructed and well-maintained systems can last up to 20 years (Hansch, 2003).

Construction costs are in between 200€ and 600€ for double VIP-dry (WHO, 2017a); 15€ for FA (Menter, 2016); and between 50€ to more than 500€ for DHP. Typical operation costs are difficult to find in literature but, for DHP, costs have been indicated around 24€ $system^{-1}$ $year^{-1}$ (in Burkina Faso and in Peru) (von Muench and Rieck, 2011) (Table 2.1). The production of GHG in RS is not documented in the literature.

2.2.2. Dry controlled systems

The DCS discussed in this section include i) composting toilets (CT), and ii) vermicomposting toilets (VCT).

2.2.2.1. Composting toilets

CT are the most common type of DCS used in locations where centralized systems are unavailable or water is scarce (Redlinger et al., 2001). In general, CT can be considered a dry system not requiring connections to water and wastewater infrastructures. Most of the studies on CT are for richer countries such as Canada, United States, United Kingdom, Australia, Sweden, and Finland. In parallel, one of the most significant urban-scale applications of CT has been tried in Mexico (Anand and Apul, 2014). Most commercial CT are technically more evolved and share a higher capital cost than other DT (Table 2.1).

Unlike the DP discussed in the earlier section, CT convert feces into a dry, odorless compost, avoiding the issues related to odors, insects, and sludge disposal. Digestion occurs inside a vault placed under the toilet or in a separated location (Jenkins, 2005). Commercial models are typically made of plastic, ceramic, fiberglass or concrete, which is typically used in own-made household systems. Systems include both manual and automatic systems, electric and non-electric systems, and with or without urine diversion (Berger, 2009). Some systems can support water flush. Urine diversion can improve compost quality (Anand and Apul, 2014).

CT are divided into thermophilic composting (or hot composting) and mesophilic composting (slow or cold composting). Both rely on long retention times: in the order of months for hot composting and of years for cold composting. When properly operated the amount of compost is one tenths of the input volume of excreta. For adequate pathogen elimination and degradation of excreta, thermophilic composting should rely on emptying intervals of 1 to 2 years (WHO, 2006). In mesophilic systems, improved degradation is possible with the addition of earthworms, as in VCT-dry (Yadav et al., 2010).

2.2.2.2. Vermicomposting toilets

VCT include: i) single and multiple vaults/chambered toilets, ii) urine and non-urine separating toilets, and iii) electric and non-electric toilets (Anand and Apul, 2014), iv) manufactured and non-manufactured toilets (Berger, 2011). All models can receive excreta, kitchen and garden wastes, and cleaning products.

Operating conditions for both CT and VCT include control of oxygen, moisture (Azim, 2017), temperature (Scott, 2002), earthworm density (Lourenço, 2014), C:N ratio (Tilley et al., 2014), bulk density (Agnew and Leonard, 2003), and ammonia concentration (Domínguez and Edwards, 2004) (Table 2.1). For pathogen destruction, temperatures above 55 ºC during at least three days, a storage at 2-20 ºC for 1.5 to 2 years, or pH>9 for 6 months should be guaranteed (WHO, 2006). A C:N ratio of 15:1 and ammonia concentration less than 2% on the end-product show the desired maturation (Lourenço, 2014). Carbon-rich materials such as sawdust or dry leaves are added after each use to increase aerobic conditions, and to control leachate production

(Jenkins, 2005). Such materials create air pockets in excreta, improve the C:N ratio, and reduce potential odors.

The design of DCS is based on i) emptying intervals (time depending on the type of composting) or ii) equipment service time, which is based on the volume of the composting chamber, the average number of users, and the annual sludge produced per user (usually assumed to be 0.05 m^3). Commercial models usually report capacity by the number of users, number of uses, or intensity of use (daily or occasional) (Anand and Apul, 2014).

Interest for DCS has been growing due to a more controlled degradation of excreta than DP, when properly used, while allowing the recycling of nutrients (Torondel, 2010), being an interesting alternative to conventional flushing toilets (Jenkins, 2005). DCS reduce the production of hydrogen sulfide, ammonia, nitrous oxide, and volatile organic compounds, being therefore good candidates to attract financing based on greenhouse gases mitigation opportunities (Reid et al., 2014). Maintenance is critical to ensure proper operation. Unfortunately, engineered systems rely more on experience than on science, resulting in a lack of standardized or established design guidelines (Anand and Apul, 2014). Many commercial DCS have been reported to fail to attain temperatures above 55 °C (Jenkins, 2005) and excess moisture can cause anaerobic conditions and impede decomposition. Hydrogen sulfide, ammonia, and volatile organic compounds produced can lead to unpleasant odors (Anand and Apul, 2014). Overall, VCT-dry require less maintenance than CT. In general, CT require daily, weekly, or monthly maintenance, while maintenance for VCT is typically annual (Hill and Baldwin, 2013).

Installation costs for DCS are higher than for flush toilets, which act as a strong barrier against its use (Anand and Apul, 2014). Operation costs, on the contrary, can be significantly lower once toilets have been acclimated and optimized. Construction costs of owner-built toilets are usually lower than manufactured ones (Berger, 2011). Typical costs vary in the range of 1,500€ (manual operation) to 2,500€ (with automatic operation) (Anand and Apul, 2014). Costs for commercial systems range from 2,000€ for intermittent use models to more than 8,000€ for community models (in Germany). Costs for multiple-vault commercial toilets vary between 800€ (in Mexico) and 3,000€ (in Norway) (Berger, 2011). Operation costs range from 0.31€ $user^{-1}$ $year^{-1}$ (CT) to

0.02€ user^{-1} year^{-1} (VCT) (Table 2.1). GHG emissions from DCS are not documented in the literature.

Table 2.1 – Technical comparison between the different dry technologies (DT).

			Double VIP	*Fossa alterna*	Dehydrating pits	Composting toilets	Vermicomposting toilets
Construction		**Surface required (m^2 user^{-1})**	≥0.60 or 1.00-1.80 (CAWST, 2011); 3.0 (Flores, 2010)	0.07-0.10 (Morgan, 2007)	1.00-10.00 (von Muench and Rieck, 2011)	0.06-0.42 (Kaczala, 2006)	0.08 (Yadav et al., 2011)
		Users system^{-1}	4.0-8.0 (Harvey et al., 2002)	≤6.0 (WHO, 1992)	1.0-10.0 (von Muench and Rieck, 2011); 20 (Kaczala, 2006)	2.0-8.0 (Anand and Apul, 2014); 8.0-10.0 (Kaczala, 2006)	1.0-6.0 (Anand and Apul, 2014)
		Earthworm density (kg m^{-2})	Not applicable	Not applicable	Not applicable	Not applicable	1.00-16.00. (Lourenço, 2014); 0.25-5.00 (Yadav et al., 2011)
		Depth (m)	≥1.50 (CAWST, 2011); ≥3.00 (Tilley et al., 2014); ≥2.00 (WHO, 2017b)	2.50 (WHO, 1992); ≤1.50 (Tilley et al., 2014)	1.00 (Kaczala, 2006)	2.90 (WHO, 1992)	0.60 (Yadav et al., 2010); 0.09 (Yadav et al., 2011)
Operation	**Parameters**	**Storage time (years)**	1.5-2.0 (WHO, 2006); 1.0 (CAWST, 2011)	0.5-2.0 (Stenström et al., 2011); ≥1.0 (warm climate) (Tilley et al., 2014)	1.0-2.0 (WHO, 2006); 0.6 to ≥1.0 (warm climates) and 1.5-2.0 (cold climates) (Tilley et al., 2014); ≤0.5 (Phi et al, 2004); 0.5 (Chien et al., 2001); 0.8 (Harada et al., 2006)	1.0-2.0 for emptying intervals (WHO, 2006); 10.0 (life cycle – no emptying) (Anand and Apul, 2014)	≤10.0-20.0 (Anand and Apul, 2014)
		Oxygen (%)	Not applicable	Not applicable	Not applicable	15-20 (Mustin, 1999; Azim, 2017)	>55-65 (Domínguez and Edwards, 2004)
		Moisture (%)	Not applicable	Not availabe	Not applicable	40-65 (Azim, 2017); 45-70 (Tilley et al., 2014); 50-60 (Anand and Apul, 2014)	75-90 (Lourenço, 2014); 50-90 (Domínguez, 2004); 60-65 (Yadav et al., 2010)
		Temperature (°C)	≥20.0 for 1.5-2.0 years for pathogen destruction (WHO, 2006)	20.0-30.0 (Stenström et al., 2011); 25.0-30.0 for 2.0 years (CAWST, 2011)	Not available	>50.0 (Epstein, 1997); 55.0 for 7 days to sufficient pathogen removal (WHO, 2006); 50.0-60.0 for pathogen and helminth destruction (Scott, 2002); 40.0-65.0 (Anand and Apul, 2014)	20-25 (Lourenço, 2014); 15-25 (Domínguez and Edwards, 2004); 25 (Yadav et al., 2010);
		C:N ratio	Not applicable	Not available		25:1 (Tilley et al., 2014); 25-35 (Anand and Apul, 2014)	20:1-25:1 (Lourenço, 2010; Lourenço, 2014)
		Bulk density (kg m^{-3})	Not applicable	Not applicable		100-400 (Agnew and Leonard, 2003)	1,200 (Yadav et al., 2010)
		Ammonia (mg g^{-1})	Not applicable	Not applicable		Not applicable	<1 (Domínguez and Edwards, 2004)
	Efficiencies	**Fecal coliforms (Log$_{10}$)**	Not available	6.0 (Stenström et al., 2011)	7.0 (Wang et al., 1999)	>8.0 (total coliforms) (Yadav et al., 2010)	9.0 (Yadav et al., 2010)
		Helminths (%)	Not available	100 (Stenström et al., 2011)	99 (Wang et al., 1999); 100 (Chien et al., 2001); 85 (Winblad and Simpson-Hébert, 2004); 100 (Harada et al., 2006)	Not available	Not available
		Helminths (Log$_{10}$)	Not available	Not available	Not available	>2.0 (within 2 hours at 50 ºC and 50% moisture) (Darimani et al., 2015); 1.2-3.9 (Keim, 2015)	
		Viruses (Log$_{10}$)	Not available	≥4.0 (Stenström et al., 2011)	7.0 (Wang et al., 1999); 8.0 (Chien et al., 2001)	Not available	
Costs	**Construction**	**€ system^{-1}**	250.00-560.00 (South Africa) (Guerreiro, 2015); 170.00-520.00 (Double VIP) (WHO, 2017a); 360.00 (Kenya) (Ulrich et al., 2016); 120.00-500.00 (World) (von Muench and Rieck, 2011)	15.00 (Menter, 2016); 3.80-5.80 (Morgan, 2007); 120.00 (China) and 580.00 (South Africa) (von Muench and Rieck, 2011)	50.00-≥500.00 (von Muench and Rieck, 2011); 570.00-810.00 (South Africa) (Winblad and Simpson-Hébert, 2004); 33.00-123.00 (Guatemala), 28.00 (China) and 30.00 (World) (Winblad, 2002); 122.00 (Mexico) (Peasey, 2000); 215.00 (Nepal) (Lamichhane, 2007)	1,500.00-2,500.00 (Anand and Apul, 2014); 2,000.00 (intermittent models)-≥8,000.00 (community models) (Germany); 800.00 (Mexico)-3,000.00 (Norway) (multiple-vault commercial toilets) (Berger, 2011); 812.00-2,300.00 (Kaczala, 2006)	Not available
		€ user^{-1}	Not available	0.73 (Menter, 2016)	2.00 (von Muench and Rieck, 2011); 108.00-853.00 (Australia) (Kaczala, 2006)	120.00-1,290.00 (Anand and Apul, 2014); 133.00-1000.00 (Kaczala, 2006)	
	Operation	**€ system^{-1} year^{-1}**	Not available	Not available	75.00 (World) (Ulrich et al., 2016); 24.00 (Burkina Faso and Peru) (von Muench and Rieck, 2011); 15.00 (Nepal) (Lamichhane, 2007)	2,150.00 (Hill and Baldwin, 2012)	220,00 (Hill and Baldwin, 2012)
		€ user^{-1} year^{-1}	2.15-7.30 (IRC, 2012)	0.02 (Menter, 2016)	Not available	0.31 (Hill and Baldwin, 2012)	0.02 (Hill and Baldwin, 2012)

2.3. Wet Technologies

WT are here divided into primary treatment systems (PTS), and secondary treatment systems (STS). Such systems are discussed separately in the following sections. Detailed information is provided in Table 2.2 and Table 2.3.

2.3.1. Primary treatment systems

The PTS discussed in this section include i) wet pits (WP), which are divided into single pits (SP), wet ventilated improved pits (VIP-wet), and pour flush latrines (PFL); ii) aqua-privy (APY); and iii) septic systems (SS), which are divided into septic tanks (ST), soakaways (SKW), and subsurface wastewater infiltration systems (SWIS).

2.3.1.1. Wet pits

WP are excavated structures like DP which act as simple collection and disposal devices (Jenkins, 2005). All models use small amounts of water for flushing and cleaning (Torondel, 2010). WP are the most common of all pit designs, being commonly used by rural or periurban regions of Southern Asia where water for flushing is available (WHO, 2017a). WP are only suitable for locations with deep water tables where flooding is not expected. While SP and VIP-wet are used for storage and leaching of excreta, PFL are used for storage, leaching, and dewatering of excreta (Tilley et al., 2014). In all PT, excreta undergo mainly anaerobic degradation (Stenström et al., 2011). Once operational time is reached, pits are sealed and replaced by a new one (WHO, 2017a). WP depend exclusively on gravity to work.

Design is based on the extent of operation time and estimated annual volume of wastewater (Harvey et al., 2002), plan area (generally circular or rectangular), soil depth, soil infiltration rate (CAWST, 2011), and sludge production (40 L user^{-1} year^{-1} to 60 L user^{-1} year^{-1}, and up to 90 L user^{-1} year^{-1} if dry cleaning materials are commonly used). Depending on the number of users, WP can last up to 20 or more years (Tilley et al., 2014) (Table 2.2).

Like DP, WP will not provide sufficient treatment time (Stenström et al., 2011) being, therefore, a potential source of contamination to groundwater with nitrates, phosphates, and pathogens (Graham and Polizzotto, 2013). WP located in soils with insufficient permeability will rapidly fill up as the applied recharge surpasses the infiltration rate. Unpleasant odors and insects are more noticeable than on DP. Again, construction is difficult or impractical in rocky and sandy soils (Flores, 2010).

The construction cost for single and community WP is highly variable. Costs for SP have been reported between 65€ and 90€ (Ethiopia), with a global average of 80€. VIP systems cost in the range of 35€ (in Ethiopia) and 190€ (Ghana). PFL cost in the range of 25€ to 80€ (in Ghana) (Table 2.1). WP can be accounted as sources of methane when temperature and retention time are inappropriate (IPCC, 2006). Typical values for methane emissions can reach 640 mg user^{-1} year^{-1} to 1,510 mg user^{-1} year^{-1} (Reid et al., 2014) (Table 2.2).

2.3.1.2. Aqua-privy

An APY is a modified type of ST where excreta fall directly from a submerged pipe into the tank (Torondel, 2010). To prevent odors, the system includes two buried components: i) a closed tank, and ii) a soakaway (SKW) (WHO 1992). The tank retains both dense settled sludge and less dense flocculent material from influent wastewater, holding long enough the solids in order for them to settle (Stenström et al., 2011). Digestion of sludge is anaerobic in the tank (CAWST, 2011). SKW usually receives the effluent from the tank and allows it to infiltrate into the soil (WHO, 2017a). Solids must be removed every 1 to 5 years (UNICEF, 2016).

APY are mainly used in rural or periurban areas where water is available (WHO, 2017a). Typical uses include regions with deep water tables where flooding is not expected, and permeable soil is available. The life operation period of a SKW will vary according to the type of soil in which it is built (WHO, 2017b). The design of APY should predict a minimal volume of 1.0 m^3 (WHO, 2017b).

Compared to WP, APY are less susceptible to clogging with cleaning materials and have fewer problems with odors and insects (WHO, 2017b). APY are also easier to empty, reducing health hazards since contact with sludge is reduced. Comparing with septic tanks (discussed below), APY need less flush water since the toilet is found directly on top of the tank (Reed et al., 2014) but, comparing with WP, more is required to operate. Sludge must be removed regularly from the tank, usually every three years

(WHO, 2017b). Insects and odors may still be present if insufficient flush water is applied (Sasse, 1998).

Treatment efficiencies are of about 30% to 40% for BOD and 50% for TSS (Tilley et al., 2014) (Table 2.2).

APY have higher installation costs than VIP or PFL (WHO, 1992) but are less expensive than ST. For instance, in India, installation costs for a family familiar APY may vary from 7€ $user^{-1}$ to 8€ $user^{-1}$ (Indiamart, 2018) (Table 2.2). Greenhouse gas emissions from APY are not documented in the literature.

2.3.1.3. Septic systems

A SS consist typically of two buried components: i) the ST, with residence times long enough for solids to settle; and ii) the soil dispersal system, commonly a SKW or a SWIS. SS are considered the simplest form of decentralized WT used in developed countries (Schaider et al. 2017) being preferred over other on-site treatment methods for long-term domestic use (Tchobanoglous and Crites, 2003). In some Eastern EU countries, ST still supplies as much as 70% of wastewater treatment, sometimes as a pre-treatment. As a result of strict EU legislation, existing SWIS formerly used are gradually being dismissed in some EU countries (Capodaglio et al., 2017).

Like the APY, digestion in ST is essential anaerobic (Schaider et al. 2017). The presence of two or more compartments will enhance the settling of solids and removal of BOD and TSS. Treatment efficiencies will be lower in colder temperatures due to a decrease of microbiological activity (Tchobanoglous and Crites, 2003).

The design of a ST should consider the volume occupied by wastewater, number of users, average daily flow (100 L $user^{-1}$ day^{-1}), hydraulic retention time (HRT), sludge production rate (25 to 60 L $user^{-1}$ day^{-1}), emptying frequency (2 years), and a typical time for digestion of solids (60 days) (Paixão, 2004) (Table 2.2).

SS can be significant contributors to nutrient pollution into coastal waters and water streams in general, although this contribution is often underestimated. Also, Climate Change may exacerbate the discharge of organic pollutants from SS, particularly in coastal zones with rising water tables (Schaider et al. 2017). Nevertheless, when proper operated, ST is recognized as having environmental, public health and economic benefits for small clusters, when compared to DP and WP (Truhlar et al., 2016). ST also represent an upgrade over APY.

The following typical treatment efficiencies have been reported: BOD (30% to 40%) (Tilley et al., 2014), COD (48% to76%) (Nguyen et al., 2006), TSS (50%) (Tilley et al., 2014), TN (18% to 34%), TP (26% to 36%) (Nasr and Mikhaeil, 2013), and fecal coliforms (<0.1 Log10 to 0.3 Log10) (USEPA, 2002) (Table 2). Generally, the longer the value of the HRT the higher the BOD, COD and TSS removal efficiencies. Overall nitrogen and phosphorous removals are low and dependent on the type of tank, HRT and anaerobic activity. Significant improvement can be achieved when integrating in-tank baffles or increasing the number of chambers (Nasr and Mikhaeil, 2014).

Typical construction costs for a ST with SKW vary in the range of 79€ user^{-1} to 570€ user^{-1} (in Africa) to 115€ user^{-1} (in Asia) (NESC, 2004), but a ST with a SWIS upgrade can vary in the range of 1,250€ system^{-1} to more than 6,500€ system^{-1} (Municipality of Anchorage, 2017). Emptying a ST has been reported to vary between 40€ to 80€ (NESC, 2004) every 2 to 5 years (Table 2.2). As ST usually require a post-treatment, costs and complexity of the system tend to increase (Capodaglio et al., 2017). ST are significant sources of greenhouse gases (Leverenz et al., 2011) but frequent removal of sludge can reduce these emissions since septic conditions are avoided (IPCC, 2006). Typical emissions for methane on ST vary from 11 g user^{-1} day^{-1} (Truhlar et al., 2016) to 27.1 g user^{-1} day^{-1} (USEPA, 2018), and equal 0.005 g user^{-1} day^{-1} for nitrogen oxide (Diaz-Valbuena et al., 2011) (Table 2.2).

SKW and SWIS are passive and effective systems which allow wastewater to infiltrate into the soil, reduce BOD and remove smaller solids (USEPA, 2002). Typical SWIS include soil absorption beds, infiltration mounds, trenches, and leaching chambers (Collado and Díez, 2010). Absorption beds are sometimes preferred in more permeable soils, but trenches provide more surface area for soil absorption (Robbins and Ligon, 2014). These systems can be continuous or have intermittent operation mode (Li et al., 2015). The design of SKW should account for an adequate slope (Reed et al., 2014), a typical depth of 4 m (Oxfam, 2008), and a minimum distance of 3 m from the tank. Trenches should be 15 m to 30 m long with open-jointed 0.1 m diameter pipes laid on rocks, broken bricks or gravel. Typical surface required is 0.09 m^2 user^{-1} (WHO, 2017b) (Table 2.2).

SWIS should be used as long there is adequate soil permeability (20 L m^{-2} day^{-1} to 50 L m^{-2} day^{-1}) (Oxfam, 2008), natural self-purification of the soil (Collado and Díez, 2010), adequate topography (slopes with more than 20% grade), and adequate depth to groundwater (more than 1.5 m). Design is based on the hydraulic loading rate (HLR), organic loading rate (OLR) (ranging from 1.3 g of BOD m^{-2} day^{-1} in clay soils to 7 g of BOD m^{-2} day^{-1} in sandy soils (USEPA, 2002)), wastewater flow, and on the number of users. Trenches have typically a depth of 0.2 m to 0.5 m and should be excavated in series 2 m apart or twice the depth if it is greater than 1 m (Oxfam, 2008). A minimum depth of 0.6 m to 2.1 m of unsaturated soil is needed to complete pathogen elimination (USEPA, 2002).

SWIS are known for their simplicity, stability and low cost, working effectively in almost all climates. SWIS are an effective technology with low operation requirements. The assimilative capacity of many soils helps degrade non-recalcitrant pollutants found in domestic wastewater. SWIS are not ideal for more than 20 users requiring large areas when the soil has a low hydraulic capacity (USEPA, 2002). The life cycle may become reduced due to significant levels of settleable solids, greases, and fats in the effluent, lack of maintenance, or improper design (USEPA, 2000).

Treatment efficiencies will depend on loading rates, dosing system, retention time, soil hydraulic properties, soil pH, air temperature and rainfall. Typical treatment efficiencies are: 80.5% for BOD, 74.5% for COD and 72% for TSS (Howarth et al., 2002). Continuous operation increases removal efficiencies for BOD, COD, TSS, TN, and TP comparing with intermittent operation (Li et al., 2015) (Table 2.2). Even so, nutrient removal is low and variable, decreasing with the increase in the age of the system (Robertson, 2010) due to the accumulation of solid particles, organic matter, and biofilm (USEPA, 2002).

Typical construction costs of a single SWIS can range from 2,500€ $system^{-1}$ to 18,000€ $system^{-1}$ (Lesikar and Persyn, 2000). However, the complete removal and replacement of an existing SWIS can cost five to ten times more than a new one. In Europe, the operation of SWIS is in between 120€ $user^{-1}$ to 250€ $user^{-1}$; in the US, costs are in the range of 80€ $user^{-1}$ to 240€ $user^{-1}$ (NESC, 2004). Over a 20-year period, the total operating costs come to 250€ $year^{-1}$ (USEPA, 2000) (Table 2.2). Greenhouse gas emissions from SWIS are not documented in the literature.

Table 2.2 – Technical comparison between the different wet technologies – primary treatment systems.

				Construction wetlands (CW)	Facultative wastewater stabilization ponds (fp-WSP)	Maturation wastewater stabilization ponds (mp-WSP)	Intermittent sand filters (int-SF)	Recirculating sand filters (rec-SF)	Peat filters (PF)	Vermifilters (VF)
Construction	HRT (hour)			96.0 (fws-CW) (Oliveira, 2007); ≤144.0 (fws-CW) (USEPA, 2002); 1.2-15.0 (fws-CW), 21.6-120.0 (hssf-CW), and 18.0 (vssf-CW) (Zhang et al., 2009); 3.0-30.0 (fws-CW) (Greenway, 2005); 24.0-192.0 (Wu et al., 2016)	30.0-180.0 (USEPA, 2002); 10.0-40.0 (Stenström et al., 2011); 24.7 (Gikas and Tsihrintzis, 2014); 21.3 (Hernandez-Paniagua et al., 2014)	30.0-180.0 (USEPA, 2002); 8 (Chalatsi and Gratziou, 2016); 13.5 (Gikas and Tsihrintzis, 2014); 8.9 (Hernandez-Paniagua et al., 2014)	Not available	Not available	Not available	1.0-3.0 (Sinha et al., 2008); 6.0-9.0 (Xing et al., 2005); 2.0-8.0 (Lourenço and Nunes, 2017a); 0.5 (Sinha et al., 2014)
	HLR (m^3/m^2.day)			0.10 (vssf-CW) (Sasse, 1998); 0.64 (fws-CW), 0.64 (hssf-CW), 0.40-0.80 (vssf-CW), and 0.36-0.58 (vssf-CW+hssf-CW) (Zhang et al., 2009); 0.40-0.74 (Greenway, 2005)	0.02-0.07 (Chalatsi and Gratziou, 2016); 0.08 (Hernandez-Paniagua et al., 2014)	0.02-0.07 (Chalatsi and Gratziou, 2016); 0.08 (Hernandez-Paniagua et al., 2014)	0.04-0.08 (USEPA, 2002); Lesikar and Persyn, 2000); 0.04 (Siervers, 1998); 0.08-0.20 (USEPA, 1999a)	0.60 (Lesikar and Persyn, 2000); 0.10-0.20 (USEPA, 2002); 0.12-0.20 (USEPA, 1999b)	0.20 (Corley et al., 2006); 0.03-0.01 (Pundsack et al., 2000; Patterson, 1999)	0.2-4.0 (Li et al. 2009); 3 (Manyuchi et al., 2013); 2.0-3.0 (Xing et al., 2005); 0.67-2.67 (Lourenço and Nunes, 2017a); 1.5 (Kumar et al., 2015)
	OLR (g BOD/m^2.day)			6.0 (Galvão, 2009) (hssf-CW); 6.2 (hssf-CW) (Oliveira, 2007); 4.5-5.9 (USEPA, 2002); 8.0 (hssf-CW), and 20.0-40.0 (vssf-CW) (Sasse, 1998)	2.9-4.9 (Chalatsi and Gratziou, 2016); 2.2-6.7 (USEPA, 2002); 2.0 (Hernandez-Paniagua et al., 2014)	1.3 (Hernandez-Paniagua et al., 2014)	25.0 (USEPA, 2002); 2.4-9.8 (USEPA, 1999a)	10.0-40.0 (USEPA, 2002); 9.8-39.1 (USEPA, 1999b)	Not available	5.5-22.2 (Lourenço and Nunes, 2017a); 7.4 (Lourenço and Nunes, 2017b)
	Surface required (m^2/user)			5.0 (Galvão, 2009) (fws-CW); 5.0 (Sasse, 1998); 0.8-3.5 (Seco, 2008); 2.0-4.0 (Gkika et al., 2014); 6.0-28.0 (Mara, 2006a)	2.1-3.2 (Chalatsi and Gratziou, 2016); 1.7-4.1 (Gikas and Tsihrintzis, 2014); 3.8-7.4 (Mara, 2006a); 2.7 (Cyprus) (Mara, 2006a)	Not available	Not available	Not available	Not available	0.6 (Soto and Tohá, 1998); 0.5 (Sinha et al., 2014)
	Dosing frequency (No. times/day)			Not applicable	Not applicable	Not applicable	12-48 (USEPA, 1999a); 12-24 (USEPA, 2002); 4-18 (Wiesman, 2016)	48-120 or more (USEPA, 1999b); 12-120 (USEPA, 2002); ≥48 (Lesikar and Persyn, 2000)	Not available	Not available
	Recirculation ratio			Not applicable	Not applicable	Not applicable	Not applicable	3:1-5:1 (USEPA, 2002)	Not applicable	0.2-0.8 (Lourenço and Nunes, 2017a); 0.7 (Lourenço and Nunes, 2017b)
	W:D (hour)			Not applicable	Not applicable	Not applicable	Not applicable	Not applicable	Not applicable	1-3 (Bhise and Anaokar, 2015); 3-9 (Wang et al., 2014)
	Bed or packing depth (m)			0.60-1.50 (USEPA, 2002); 0.60 (hssf-CW) (Andreo-Martínez et al., 2016)	2.40-2.50 (Chalatsi and Gratziou, 2016); 1.00-2.00 (Abdullahi et al., 2014); 1.00-3.50 (Orumieh and Mazaheri, 2015); 1.70 (Hernandez-Paniagua et al., 2014)	0.90-1.50 (USEPA, 2002); 1.50 (Chalatsi and Gratziou, 2016); 0.70 (Hernandez-Paniagua et al., 2014)	0.60-0.90 (USEPA, 2002)		0.30-1.20 (Corley et al., 2006); 1.50 (Kõiv et al., 2009); 0.60 (Patterson, 2004)	0.10 (Arora and Kazmi, 2015); 0.15 (Kumar et al., 2015); 0.20 (Lourenço and Nunes, 2017a); 0.40 (Nie et al., 2015); 0.40-0.80 (Wang et al., 2014)
	Media size (mm)			Not applicable	Not applicable	Not applicable	0.30-0.50 (Crites and Tchobanoglous, 1998); 0.25-0.75 (USEPA, 1999a)	0.25-0.30 (Boyle, 1995); Darby et al, 1996); 1.00-3.00 (USEPA, 1999b)	Not applicable	Not applicable
	Media uniformity coefficient			Not applicable	Not applicable	Not applicable	<4 (USEPA, 1999a; USEPA, 2002)	<4 (USEPA, 1999b)	Not applicable	Not applicable
	Earthworm abundance (g/L)			Not applicable	Not applicable		Not applicable	Not applicable	Not applicable	10-40 (Zhao et al., 2010; Tomar and Suthar, 2011; Lourenço and Nunes, 2017a); 20 (Lourenço and Nunes, 2017b); 2.5-5.0 (Soto and Tohá, 1998); 25 (Manyuchi et al., 2013); 30 (Kumar et al., 2014)
	L:W			<1:1 (fws-CW) (Sundaravadivel and Vigneswaran, 2001)	2:1-3:1 (Quiroga, 2018)		Not applicable	Not applicable	Not applicable	Not applicable
Operation	Efficiencies	BOD (%)		63.0 (Song et al., 2006); 91.0 (Vymazal, 2009); 80.0-93.0 (Vergeles et al., 2015); 70.0-95.0 (Seco, 2008); 69.0-86.0 (Zhang et al., 2009); 54.2-57.1 (Licata et al., 2017)	75.0-95.0 (USEPA, 2002); 79.0 (Mburu et al., 2013); 80.0 (Orumieh and Mazaheri, 2015)		99.0 (Cagle and Johnson, 1994; Siervers, 1998)	98.0 (Piluk and Peters, 1994); 94.0 (Roy and Dube, 1994); 97.0 (USEPA, 1999b)	98.0 (Corley et al., 2006); >90.0 (Gunes and Ayaz, 1998)	86.0 (Arora et al., 2016); 97.5 (Lourenço and Nunes, 2017a); 99.0 (Soto and Tohá, 1998); 91.0-98.0 (Xing et al., 2005); >90.0 (Sinha et al., 2008); 99.0 (Cardoso-Vigueros et al., 2013)
					75.0 (Orumieh and Mazaheri, 2015)	70.0 (Orumieh and Mazaheri, 2015)				
		COD (%)		77.3 (Vergeles et al. 2015); 63.0-77.0 (Song et al., 2006); 84.0 (Vymazal, 2009); 65.0-91.0 (Seco, 2008); 67.0-89.0 (Zhang et al., 2009); 72.0-79.2 (Licata et al., 2017)	62.0 (Mburu et al., 2013); 78.0 (Orumieh and Mazaheri, 2015)		Not available	Not available	86.0 (Corley et al., 2006)	74.3 (Lourenço and Nunes, 2017a); 81.0-86.0 (Xing et al., 2005); 80.0-90.0 (Sinha et al., 2008); 92.0 (Cardoso-Vigueros et al., 2013); 72.0-88.0 (Furlong et al., 2014)
					69.0 (Orumieh and Mazaheri, 2015)	68.0 (Orumieh and Mazaheri, 2015)				
		TSS (%)		72.1 (Vergeles et al. 2015); 60-67 (Song et al., 2006); 89.0 (Vymazal, 2009); 55.0-97.0 (Seco, 2008); 87.0-99.0 (Zhang et al., 2009); 64.5-69.5 (Licata et al., 2017)	>90 (USEPA, 2002); 67 (Mburu et al., 2013); 74 (Orumieh and Mazaheri, 2015)		78.0 (Cagle and Johnson, 1994); 93.0 (Siervers, 1998)	89.0 (Piluk and Peters, 1994); 96.0 (Roy and Dube, 1994); 76.0 (USEPA, 1999b)	96.0 (Corley et al., 2006); >90.0 (Gunes and Ayaz, 1998), 43.0 (Patterson, 2004)	0 (Arora et al., 2016); 98.2 (Lourenço and Nunes, 2017a); 95.0 (Soto and Tohá, 1998); 97.0-98.0 (Xing et al., 2005); 90.0-95.0 (Sinha et al., 2008); 97.0 (Cardoso-Vigueros et al., 2013)
					70.0 (Orumieh and Mazaheri, 2015)	79.0 (Orumieh and Mazaheri, 2015)				
		NH_4^+ (%)		35.0-45.0 (Song et al., 2006); 19.0 (Vymazal, 2009); 17.0-53.0 (Zhang et al., 2009); 57.5-59.7 (Licata et al., 2017)	>80.0 (USEPA, 2002); 95.0 (Kayombo, 2005)	Not available	Not available	80.0 (USEPA, 1999b)	99.0 (Corley et al., 2006); 96.0 (Patterson, 2004)	90.0 (Arora et al., 2016); 88.1 (Lourenço and Nunes, 2017a); 76.0 (Kumar et al., 2015); 98.0 (Cardoso-Vigueros et al.,2013)
		TN (%)		9 (Vergeles et al. 2015); 34-72 (Seco, 2008); 31-59 (Zhang et al., 2009); 81-85 (Greenway, 2005); 51.9-54 (as TKN) (Licata et al., 2017)	60 (USEPA, 2002); 80 (Kayombo, 2005); 22 (Orumieh and Mazaheri, 2015)	22 (fp-WSP); 23 (mp-WSP) (Orumieh and Mazaheri, 2015)	39.0 (Cagle and Johnson, 1994); 18.0-33.0 (USEPA, 2002); 26.0 (Siervers, 1998)	53.0 (Piluk and Peters, 1994); 47.0 (Roy and Dube, 1994); 40.0-60.0 (USEPA, 2002)	22.0-67.0 (Gunes and Ayaz, 1998); 54.0 (Patterson, 2004)	60.2 (Wang et al., 2011); 46.0-73.0 (Cardoso-Vigueros et al., 2013)
		TP (%)		22.0-31.0 (Song et al., 2006); 19.0 (Vymazal, 2009); 30.0-95.0 (Seco, 2008); 62.0-97.0 (Zhang et al., 2009); 20.0-28.0 (Greenway, 2005); 35.1-36.4 (Licata et al., 2017)	50.0 (USEPA, 2002); 19.0 (Orumieh and Mazaheri, 2015)	25.0 (fp-WSP); 21.0 (mp-WSP) (Orumieh and Mazaheri, 2015)	Not available	>20.0-30.0 (USEPA, 2002)	75.0 (Patterson, 2004)	82.0 (Wang et al., 2013)
		Fecal coliforms	%	82-99.9 (Greenway, 2005); 65 (hybrid CW) to 99 (hssf-CW) (Vymazal, 2005); 44.4-96 (Hinds et al., 2012); 63.5-99.2 (hssf-CW) (Wu et al., 2016); 89.8-90.5 (hssf-CW) (Licata et al., 2017)	99 (Tyagi et al., 2016)		99.9 (Cagle and Johnson, 1994); 99.9 (Siervers, 1998)	99.6 (Piluk and Peters, 1994); 97.3 (Roy and Dube, 1994); 99.0-99.9 (USEPA, 2002)	>99.0 (Patterson, 1999; Pundsack et al., 2000); >99.9 (Gunes and Ayaz, 1998)	Not available
			Log_{10} reduction	1.0-2.0 (Oliveira, 2007); 0.5-3 (WHO, 2006); 2.0-3.0 (USEPA, 2002) (Garcia et al., 2013)	2.0-3.0 (USEPA; 2002)	Not available	2.0 (Cagle and Johnson, 1994); 2.8 (Siervers, 1998)	2.4 (Piluk and Peters, 1994); 1.6 (Roy and Dube, 1994); 2.0-3.0 (USEPA, 2002)	Not available	2.8 (Arora et al., 2016); 2.7 (Arora et al., 2014); 3.4 (Kumar et al., 2015); 3.7 (Lourenço and Nunes, 2017b)
					2-3.5 (Tyagi et al., 2016)					
		Helminth eggs	%	99.0 (Reinoso et al. (2008), Barbagallo et al., (2011))	>99 (Campos et al., 2002); 100 (Tyagi et al., 2016)	Not available	Not available	Not available	Not available	Not available

2.3.2. Secondary treatment systems

STS reviewed include i) constructed wetlands (CW), ii) packed bed filters (PBF), in which are included sand filters (SF) and peat filters (PF); iv) vermifilters (VF); and v) wastewater stabilization ponds (WSP).

2.3.2.1. Constructed wetlands

CW replicate the functions of natural wetlands involving wetland vegetation, soils, and their associated microbial assemblages (USEPA, 2004a). Macrophyte plants can develop abundant rhizomes being adapted to live in saturated soils. Such plants use organic and inorganic substances present in wastewater for their metabolism (Scholz and Lee, 2005). CW have been used worldwide (Salas, 2005), but mostly in Mediterranean countries (Rodrigues, 2004), North America (USEPA, 2002) and China (Zhang et al., 2009). In Europe, most of them have been designed for 500 users (Correia et al., 2016).

Typical types of CW include free water surface (fws-CW), vertical subsurface flow (vssf-CW), horizontal subsurface flow (hssf-CW) (Vymazal, 2011), and vssf-CW+hssf-CW (Vymazal, 2007). Typical fws-CW are shallow and low flow velocity wetlands with areas of open water and floating and submerged and/or emergent macrophytes (Kadlec and Wallace, 2008). The vssf-CW are flatbeds (USEPA, 2000) filled with gravel topped with sand or other porous filter material (Davis et al., 2000). Beds are fed intermittently with large batches, flooding the surface. Wastewater percolates down through the bed where is collected by a drainage network at the bottom. The bed drains completely, which allows air to refill it. In hssf-CW wastewater is fed using an inlet which flows slowly under the surface of the bed to the outlet, where it is collected before leaving through a water level control structure. Most of the bed is anoxic/anaerobic due to permanent saturation (Mander et al., 2014).

The combination vssf-CW+hssf-CW includes parallel vertical subsurface flow beds followed by two or three horizontal subsurface flow beds in series (Vymazal, 2007). When the production of macrophytes is the main goal, the system can also be followed by a larger fws-CW (Maddison et al., 2009). Both vssf-CW and hssf-CW are capable of being operated under colder conditions than those for fws-CW (Mander and Jenssen, 2003). However, fws-CW can be adapted to larger wastewater flows than vssf-CW and hssf-CW (Ghermandi et al., 2007).

Configuration of CW depend on land morphology, but CW are usually rectangular (Duarte et al., 2010). Sizing methods are based on hydraulics and local hydrology, being dependent on influent flow rate and BOD, ambient temperature, and bed properties, namely granulometry, porosity and hydraulic conductivity (Sundaravadivel, M., Vigneswaran, S. (2001)) (Table 2.3).

CW have good landscape integration, being odor and insect free (Mano et al., 2003). The production of sludge is low (Galvão and Matos, 2004). If properly planned and maintained, they can provide water for reuse, work as wildlife habitats, and public use benefits (USEPA, 2004a). CW are simple to maintain, but treatment performance is strongly influenced by weather (USEPA, 2000). The best prospects for successful treatment should be in warmer regions (Stein and Hook, 2005).

Some plant species have shown to remove several pollutants including heavy metals (Gill et al., 2014), hydrocarbons (Wallace et al., 2011), polycyclic aromatic hydrocarbons (Warężak et al., 2016) and polychlorinated biphenyls (Collins et al., 2006), which are transformed to less soluble forms or become inactive (USEPA, 2000).

CW can occupy large areas with values ranging from 5 m^2 $user^{-1}$ to 6 m^2 $user^{-1}$ on hssf-CW to 1 m^2 $user^{-1}$ to 3 m^2 $user^{-1}$ on vssf-CW (Cooper, 2005) (Table 2.3). Between one and two years is needed to achieve acclimation. A typical operational concern is the clogging of the bed (Seco, 2008). It occurs in hssf-CW rather than in vssf-CW (Vohla et al., 2011) typically after five to fifteen years of operation (Duarte et al., 2010). The main causes of clogging are the accumulation of solids within bed media pores, inadequate design, excessive loading of organic matter, and suspended solids. The accumulation of solids may result from biofilm growth, vegetal debris, roots and rhizomes development, wastewater solids entrapment, and chemical precipitation (Pedescoll et al., 2009). Maintaining a constant vegetation biomass is usually the major operational task (Sundaravadivel and Vigneswaran, 2001). In hssf-CW nitrification is limited but denitrification often occurs (Vymazal and Kröpfelova, 2008), and removal of phosphorous is usually low (Mander et al., 2014).

Both fws-CW and hssf-CW are designed to remove BOD, COD, and TSS (Nilsson, 2012) but fws-CW is mostly used for tertiary treatment (Kadlec and Wallace, 2008).

The use of vssf-CW+hssf-CW can increase removal efficiencies, especially for TN (Tunçiper, 2009). In vssf-CW the removal of BOD, COD, and TSS is higher (Vymazal and Kröpfelova, 2008) than in hssf-CW. Also, vssf-CW can provide better oxygen transfer into the bed, improving nitrification (Cooper, 2005). Effluents from CW may require proper disinfection before reusing (USEPA, 2002).

Typical reported efficiencies are: 91% for BOD, 84% for COD, 89% for TSS (Vymazal, 2009), 81% to 85% for TN (Greenway, 2005), 20% to 30% for TP (Vymazal, 2009), and 82% to 99.9% for fecal coliforms (Greenway, 2005) (Table 2.3). The introduction of epigenic earthworms in CW is gathering recent interest contributing to decrease clogging and accelerate the decomposition of organic pollutants (Xu et al., 2013). It also helps to improve the conversion of suspended solids into soluble organic forms (Tomar and Suthar, 2011).

Construction costs of CW depend on the occupied land area, excavation, bedding materials, pumps and piping, and vegetation (Gunes et al., 2011). Reported costs are of 0.20€ $user^{-1}$ m^{-2} for vssf-CW and 0.29€ $user^{-1}$ m^{-2} for fws-CW) (Taylor et al., 2010). Operation costs are related to personnel, energy consumption (USEPA, 2000), and the transport of the vegetation to landfill (Machado et al., 2007). Reported costs are 0.17€ m^{-2} $year^{-1}$ to 0.28€ m^{-2} $year^{-1}$ for fws-CW, and 0.21€ m^{-2} $year^{-1}$ to 0.34€ m^{-2} $year^{-1}$ for vssf-CW (Table 2.3). The emission of greenhouse gases by CW are from 1.8 mg m^{-2} $hour^{-1}$ (fws-CW) (Johansson et al., 2003) to 6.4 mg m^{-2} $hour^{-1}$ (hssf-CW) (Mander et al., 2014) for methane; and from 0.031 mg m^{-2} $hour^{-1}$ (fws-CW) (Johansson et al., 2003) to 0.42 mg m^{-2} $hour^{-1}$ (vssf-CW) for nitrous oxide (Mander et al., 2014) (Table 2.3).

2.3.2.2. Packed bed media filters

PBF, which include sand filters (SF) and peat filters (PF), are discussed here. PBF are aerobic and fixed film filtration systems where wastewater is intermittently loaded by gravity in small doses into the upper surface of a filter packing allowing it to percolate through the system. The main treatment is accomplished by microorganisms attached to the filter packing. Wastewater applied must receive initial treatment, usually using a ST (USEPA, 2002). Unsaturated flow conditions and air movement are facilitated between doses (Headley, 2006). The process is based on a combination of physical, chemical and biological interactions (Wang, 2015). Filters can be placed above ground or at ground level. A range of materials has been successfully used including inorganic materials such as gravel, rock, silica sand, crushed glass and plastic, or organic materials such as peat (Headley, 2006).

SF are one of the oldest wastewater treatment methods, with several models extensively tested and used (USEPA, 2002). Washed, graded sand is the most common packing material, but gravel can be also used under the sand to support the media and surround the drain system. Wastewater is applied either by gravity or pressure.

Clogging in SF occurs when solids accumulate in the bed, mainly when primary treatment is inefficient or there is excessive microbial growth in the filter (Lesikar and Persyn, 2000). Odors from open filters may require the cover of the filter or the installation on isolated places. Weekly maintenance is required for the media, pumps and controls, and design must address extremely cold temperatures (Solomon et al., 1998). For filters working with high HLR sand packing should be replaced every 2 to 5 years (Gustafson et al., 2001).

SF are very reliable in the treatment of BOD, TSS, and fecal coliforms. If properly designed and operated SF can produce effluents with the desirable quality for discharge standards (USEPA, 2002). Nearly complete ammonia removal is achieved, but due to the higher and coarser packing media used, elimination of fecal coliforms is less efficient in recirculating systems than on intermittent ones (USEPA, 2002). Typical efficiencies are: >98% for BOD, 78% to 96% for TSS, 18% to 60% for TN and 1.59 Log_{10} to 3.00 Log_{10} for fecal coliforms (Table 2.3).

Filter media is the most expensive component and one of the most significant factors affecting the cost of SF (Wiesman, 2016). Reported costs for filter media are in the range of 90€ m^{-2} to 135€ m^{-2}. Typical values for GHG emissions are of 0.0072 g user-1 day^{-1} for methane and 0.006 g $user^{-1}$ day^{-1} for nitrous oxide (Truhlar et al., 2016).

PF are used for domestic wastewater treatment since the early 70s in USA, Canada, Australia, Ireland, and Spain. Models typically range from simple gravity feed filters to modular recirculation filters. Microbial populations use the organic materials in peat as source of nutrients (Patterson, 1999). Such systems are effective in conditions when loadings are seasonal or intermittent (USEPA, 2002). Sphagnum peat (Kõiv et al., 2009) and peat moss (Headley, 2006) are the most common packing materials used. PF can be run intermittently or in recirculation, in this last case to enhance nitrogen removal.

Typical sizing parameters for PF include HLR and packing bed depth (Corley et al., 2006 (Table 2.3). High peat depths are required to remove suspended solids, improve nitrification, remove phosphorus and ensure enough pathogen removal (Kõiv et al., 2009).

As in SF, a primary treatment should precede PF. The bed should be replaced every 8 years to 15 years (Corley et al., 2006). PF are passive and robust systems under variable loadings, having little maintenance (USEPA, 2002). The environmental impacts of peat mining and extraction are the main obstacles to this technology (Headley, 2006).

Treatment efficiencies vary according to the bulk density of peat, peat variety, and OLR. Typical values are: 98% for BOD, 86% for COD, 96% for TSS, 54% for TN, 75% for TP and >99% for fecal coliforms (Patterson, 2004).

Current prices for Sphagnum peat are of 10.55€ m^{-3} (Jasinski, 2017). Overall, construction costs vary in the range of 8,300€-12,500€ per system. Operation costs are usually depended on the replacement of the bed, but currently, there is no cost information in the literature (Table 2.3). Reported GHG emissions from peat mining are 6.8 g m^{-2} of methane and 210 g m^{-2} of nitrous oxide (Couwenberg, 2009).

2.3.2.3. Vermifilters

VF are packed filters which contain epigenic earthworms (Figure 3.1). Such systems offer favorable conditions to stimulate and accelerate microbial degradation and increase the permeability of the filter. Earthworms ingest wastewater solids, converting them into vermicasts (Garg et al., 2006). Physical, chemical and biological properties of pollutants are changed due to earthworm intermediation (Liu et al., 2012).

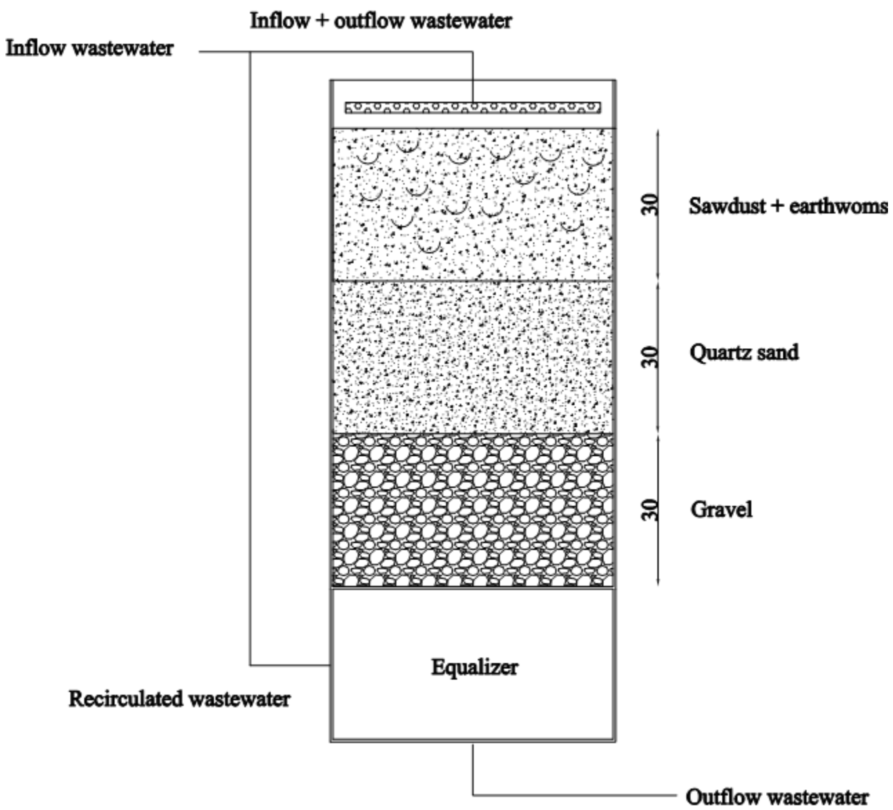

Figure 2.2 – Side view of a typical VF.

Figure 2.3 – Epigenic earthworms (*Eisenia* spp.) used in VFs.

VF are widely used to treat wastewater in households and small communities (Sinha et al., 2014) in Europe and South America. VF have been used also as a PTS, receiving the designation of vermicomposting flushing toilets (VCFT) (Table 2.3) (Howard and Ripley, 2016), but are mainly used as secondary treatment (Lourenço and Nunes, 2017b), usually advisable in order to decrease clogging (Lourenço and Nunes, 2017a).

VF are simple structures with paralelipipedic (Kumar et al., 2014) or cylindrical shape (Taylor et al., 2003). The following parameters are used to size the systems (Table 2.3): flow type (subsurface vertical flow and subsurface horizontal flow) (Samal et al., 2017), HRT (Gupta and Petla, 2015), HLR, OLR (Sinha et al., 2010), wet to dry ratio (W:D), bed material (organic and inert packing), and earthworm abundance (Lourenço and Nunes, 2017a). The design includes continuous or intermittent flow and subsurface vertical and subsurface horizontal flow. A W:D period is usually adopted to remove anaerobic biomass and restore the hydraulic capacity of the filter (Samal et al., 2017). Common packing materials include, as organic packing, vermicompost, sawdust (Lourenço and Nunes, 2017a), wood chips (Furlong and Enrique-Hernández, 2017), bark, peat, straw (Li et al., 2009), and garden soil (Singh et al., 2017); and as inert packing, gravel, quartz sand (Lourenço and Nunes, 2017b), river gravel, mud balls, glass balls (Kumar et al., 2015), clay pebbles (Furlong and Enrique-Hernández, 2017), ceramsite, and coal (Wang et al., 2010a).

VF are considered sustainable and resilient systems (Furlong and Enrique-Hernández, 2017), having low energy consumption (Sinha et al., 2008), not producing sludge (Kumar et al., 2014). No costly machinery or skilled labor are required. The operation includes the periodical refilling of packing material with carbon-rich materials and the manual aeration of the bed. Being aerobic systems, VF can significantly reduce the production of unpleasant odors and the presence of insects. The presence of vermicasts has been reported to increase the concentration of total dissolved organic matter, reducing treatment efficiencies (Lourenço and Nunes, 2017b). One solution adopted is to keep a steady abundance of earthworms and harvest the vermicasts every three-four months (Singh et al., 2017).

Typical treatment efficiencies are: 99% for BOD, 92% for COD and 98% for TSS. The use of vertical VF and fourth-stage VF can increase ammonia removal to 83% (Lourenço and Nunes, 2017a) (Table 3.1).

Typical construction costs are between 100€ user^{-1} and 150€ user^{-1}, while operation costs decrease when increasing the number of users, with typical values around 0.05€ m^{-3} year^{-1} (Sinha et al., 2014). GHG from VF are 0.35 g m^{-2} hour^{-1} to 77 g m^{-2} hour^{-1} for nitrous oxide. VF can typically act as a sink for methane (0.58 g m^{-2} hour^{-1} to 1.71 g m^{-2} hour^{-1}) but when high organic loading rates are applied, methane emissions up to 11.2 g m^{-2} hour^{-1} were reported (Luth et al., 2011).

2.3.2.4. Wastewater stabilization ponds

WSP are earth structures capable of containing wastewater, built to facilitate the occurrence of biological processes with or without the presence of oxygen, and involving both algae and bacteria. Such systems have been extensively used for the last 50 years particularly in small towns of Europe, Israel, India, South Africa, and North and South America (Howard and Ripley, 2016).

WSP comprise one or a sequence of different types of man-made ponds. Usually, the first pond is anaerobic and used for wastewater with high organic loads, followed by a facultative stabilization pond (fp-WSP). Both ponds may need to be complemented by maturation ponds (mp-WSP) depending on the required final effluent quality (Varón and Mara, 2004).

The fp-WSP have a typical depth of 1.1 m to 1.5 m (Glaz et al., 2016), being designed typically for BOD removal. These systems occupy more surface area than mp-WSP (Gikas and Tsihrintzis, 2014). Because of the lower organic loadings received, fp-WSP are well oxygenated (Peña and Mara, 2004) promoting adequate algae growing. The mp-WSP are shallower than fp-WSP having a typical depth of 0.7 m (Hernández-Paniagua et al., 2014), being particularly suited for pathogen removal. Size and number of mp-WSP depend on the required bacteriological quality of the final effluent.

Sizing is based on HRT, HLR and OLR. Typical HRT values include 24 days to 81 days (Glaz et al., 2016) for fp-WSP and 8.9 days for mp-WSP; typical HLR values are 0.02 m^{3} m^{-2} day^{-1} to 0.07 m^{3} m^{-2} day^{-1} both for fp-WSP and mp-WSP (Chalatsi and Gratziou, 2014), and typical OLR values are 80 kg BOD hectare^{-1} day^{-1} to 400 kg BOD hectare^{-1} day^{-1} for fp-WSP, and 12.9 kg BOD hectare^{-1} day^{-1} for mp-WSP (Hernández-Paniagua et al., 2014) (Table 2.3). Maintaining an adequate algal population will generate the oxygen needed by bacteria to remove BOD. Algal growth is affected by the availability of nutrients, pH, light intensity, temperature, and biotic factors such as algae density (Abdel-Raouf et al., 2012).

WSP are simple, reliable and low maintenance solutions (Alexiou and Mara, 2003), having low construction and operating costs, long operation life, being capable of handling well fluctuating hydraulic and organic loads (Yi et al., 2009). WSP do not need to be aerated, and do not require electromechanical equipment (Peña and Mara, 2004). In general, WSP occupy more area per user than other STS, requiring 1.7 m^{-2} $user^{-1}$ to 4.1 m^{-2} $user^{-1}$ (Gikas and Tsihrintzis, 2014) (Table 3.1). WTS can often attract and develop insects (Stenström et al., 2011), and produce odors. The main obstacle to the use of this technology, in particular fp-WSP, is the presence of high concentration of dissolved solids in the treated wastewater, mostly due to the high concentration of algae (Kaya et al., 2007). WSP do not meet effluent criteria consistently throughout the year, often requiring additional treatment to attain discharge standards (Massoud et al., 2009).

Efficiencies are highly dependent on sunlight, wind, temperature, rainfall, and evaporation (Tadesse et al., 2004), but increase with longer HRT and high HLR (Orumieh and Mazaheri, 2015). The highest removals of BOD and COD occur in warmer climates due to the increase of biological processes and number of sunlight hours. The fp-WSP systems are usually more effective at removing BOD than mp-WP (Mozaheb, 2010) but mp-WP can make a significant contribution to nutrient removal and pathogen destruction (Naddafi et al., 2009).

Costs are dictated by the cost of land, accounting for 60% of total investment needs. Typical surface area requirements are between 2 m^{-2} $user^{-1}$ and 7 m^{-2} $user^{-1}$ (Table 2.3). Typical construction costs vary from 105€ $user^{-1}$ (in France) to 343€ $user^{-1}$ (in Germany) (Mara, 2006), with typical operating costs varying from 0.50€ $user^{-1}$ $year^{-1}$ (in Germany) (Berland and Cooper, 2001) to 4€ $user^{-1}$ $year^{-1}$ (in France) (Mara, 2006). WSP are improbable sources of GHG, but if poorly designed or managed both fp-WSP and mp-WSP can produce methane (Heubeck and Craggs, 2010). In these cases, typical values are of 25 mg m^{-2} $hour^{-1}$ for methane, and between 5.0×10^{-4} mg m^{-2} $hour^{-1}$ and 4.0×10^{-2} mg m^{-2} $hour^{-1}$ for nitrous oxide (Hernández-Paniagua et al., 2014).

Table 2.3 – Technical comparison between the different wet technologies – secondary treatment systems (STS).

<table>
<tr><th colspan="4"></th><th>Construction wetlands (CW)</th><th>Facultative wastewater stabilization ponds (fp-WSP)</th><th>Maturation wastewater stabilization ponds (mp-WSP)</th></tr>
<tr><td rowspan="13">Construction</td><td colspan="3">HRT (hour)</td><td>96.0 (fws-CW) (Oliveira, 2007); ≤144.0 (fws-CW) (USEPA, 2002); 1.2-15.0 (fws-CW), 21.6-120.0 (hssf-CW), and 18.0 (vssf-CW) (Zhang et al., 2009); 3.0-30.0 (fws-CW) (Greenway, 2005); 24.0-192.0 (Wu et al., 2016)</td><td>30.0-180.0 (USEPA, 2002);
10.0-40.0 (Stenström et al., 2011);
24.7 (Gikas and Tsihrintzis, 2014);
21.3 (Hernandez-Paniagua et al., 2014)</td><td>30.0-180.0 (USEPA, 2002);
8 (Chalatsi and Gratziou, 2016);
13.5 (Gikas and Tsihrintzis, 2014);
8.9 (Hernandez-Paniagua et al., 2014)</td></tr>
<tr><td colspan="3">HLR (m^3 m^{-2} day^{-1})</td><td>0.10 (vssf-CW) (Sasse, 1998); 0.64 (fws-CW), 0.64 (hssf-CW), 0.40-0.80 (vssf-CW), and 0.36-0.58 (vssf-CW+hssf-CW) (Zhang et al., 2009); 0.40-0.74 (Greenway, 2005)</td><td>0.02-0.07 (Chalatsi and Gratziou, 2016); 0.08 (Hernandez-Paniagua et al., 2014)</td><td>0.02-0.07 (Chalatsi and Gratziou, 2016);
0.08 (Hernandez-Paniagua et al., 2014)</td></tr>
<tr><td colspan="3">OLR (g BOD m^{-2} day^{-1})</td><td>6.0 (Galvão, 2009) (hssf-CW); 6.2 (hssf-CW) (Oliveira, 2007); 4.5-5.9 (USEPA, 2002); 8.0 (hssf-CW), and 20.0-40.0 (vssf-CW) (Sasse, 1998)</td><td>2.9-4.9 (Chalatsi and Gratziou, 2016);
2.2-6.7 (USEPA, 2002);
2.0 (Hernandez-Paniagua et al., 2014)</td><td>1.3 (Hernandez-Paniagua et al., 2014)</td></tr>
<tr><td colspan="3">Surface required
(m^2 $user^{-1}$)</td><td>5.0 (Galvão, 2009) (fws-CW); 5.0 (Sasse, 1998); 0.8-3.5 (Seco, 2008); 2.0-4.0 (Gkika et al., 2014); 6.0-28.0 (Mara, 2006)</td><td>2.1-3.2 (Chalatsi and Gratziou, 2016);
1.7-4.1 (Gikas and Tsihrintzis, 2014); 3.8-7.4 (Mara, 2006);
2.7 (Cyprus) (Mara, 2006a)</td><td>Not available</td></tr>
<tr><td colspan="3">Dosing frequency (No. times day^{-1})</td><td>Not applicable</td><td>Not applicable</td><td>Not applicable</td></tr>
<tr><td colspan="3">Recirculation ratio</td><td>Not applicable</td><td>Not applicable</td><td>Not applicable</td></tr>
<tr><td colspan="3">W:D (hour)</td><td>Not applicable</td><td>Not applicable</td><td>Not applicable</td></tr>
<tr><td colspan="3">Bed or packing depth (m)</td><td>0.60-1.50 (USEPA, 2002); 0.60 (hssf-CW) (Andreo-Martínez et al., 2016)</td><td>2.40-2.50 (Chalatsi and Gratziou, 2016);
1.00-2.00 (Abdullahi et al., 2014);
1.00-3.50 (Orumieh and Mazaheri, 2015);
1.70 (Hernandez-Paniagua et al., 2014)</td><td>0.90-1.50 (USEPA, 2002);
1.50 (Chalatsi and Gratziou, 2016);
0.70 (Hernandez-Paniagua et al., 2014)</td></tr>
<tr><td colspan="3">Media size (mm)</td><td>Not applicable</td><td>Not applicable</td><td>Not applicable</td></tr>
<tr><td colspan="3">Media uniformity coefficient</td><td>Not applicable</td><td>Not applicable</td><td>Not applicable</td></tr>
<tr><td colspan="3">Earthworm abundance (g L^{-1})</td><td>Not applicable</td><td colspan="2">Not applicable</td></tr>
<tr><td colspan="3">L:W</td><td><1:1 (fws-CW) (Sundaravadivel and Vigneswaran, 2001)</td><td colspan="2">2:1-3:1 (Quiroga, 2018)</td></tr>
<tr><td colspan="3"></td><td></td><td colspan="2"></td></tr>
<tr><td rowspan="13">Operation</td><td rowspan="13">Efficiencies</td><td colspan="2" rowspan="2">BOD (%)</td><td rowspan="2">63.0 (Song et al., 2006); 91.0 (Vymazal, 2009); 80.0-93.0 (Vergeles et al., 2015); 70.0-95.0 (Seco, 2008); 69.0-86.0 (Zhang et al., 2009); 54.2-57.1 (Licata et al., 2017)</td><td colspan="2">75.0-95.0 (USEPA, 2002); 79.0 (Mburu et al., 2013);
80.0 (Orumieh and Mazaheri, 2015)</td></tr>
<tr><td>75.0 (Orumieh and Mazaheri, 2015)</td><td>70.0 (Orumieh and Mazaheri, 2015)</td></tr>
<tr><td colspan="2" rowspan="2">COD (%)</td><td rowspan="2">77.3 (Vergeles et al. 2015); 63.0-77.0 (Song et al., 2006); 84.0 (Vymazal, 2009); 65.0-91.0 (Seco, 2008); 67.0-89.0 (Zhang et al., 2009); 72.0-79.2 (Licata et al., 2017)</td><td colspan="2">62.0 (Mburu et al., 2013); 78.0 (Orumieh and Mazaheri, 2015)</td></tr>
<tr><td>69.0 (Orumieh and Mazaheri, 2015)</td><td>68.0 (Orumieh and Mazaheri, 2015)</td></tr>
<tr><td colspan="2" rowspan="2">TSS (%)</td><td rowspan="2">72.1 (Vergeles et al. 2015); 60-67 (Song et al., 2006); 89.0 (Vymazal, 2009);
55.0-97.0 (Seco, 2008); 87.0-99.0 (Zhang et al., 2009); 64.5-69.5 (Licata et al., 2017)</td><td colspan="2">>90 (USEPA, 2002); 67 (Mburu et al., 2013);
74 (Orumieh and Mazaheri, 2015)</td></tr>
<tr><td>70.0 (Orumieh and Mazaheri, 2015)</td><td>79.0 (Orumieh and Mazaheri, 2015)</td></tr>
<tr><td colspan="2">NH_4^+ (%)</td><td>35.0-45.0 (Song et al., 2006); 19.0 (Vymazal, 2009); 17.0-53.0 (Zhang et al., 2009); 57.5-59.7 (Licata et al., 2017)</td><td>>80.0 (USEPA, 2002); 95.0 (Kayombo, 2005)</td><td>Not available</td></tr>
<tr><td colspan="2">TN (%)</td><td>9 (Vergeles et al. 2015); 34-72 (Seco, 2008); 31-59 (Zhang et al., 2009); 81-85 (Greenway, 2005); 51.9-54 (as TKN) (Licata et al., 2017)</td><td>60 (USEPA, 2002); 80 (Kayombo, 2005); 22 (Orumieh and Mazaheri, 2015)</td><td>22 (fp-WSP); 23 (mp-WSP)
(Orumieh and Mazaheri, 2015)</td></tr>
<tr><td colspan="2">TP (%)</td><td>22.0-31.0 (Song et al., 2006); 19.0 (Vymazal, 2009); 30.0-95.0 (Seco, 2008); 62.0-97.0 (Zhang et al., 2009); 20.0-28.0 (Greenway, 2005); 35.1-36.4 (Licata et al., 2017)</td><td>50.0 (USEPA, 2002); 19.0 (Orumieh and Mazaheri, 2015)</td><td>25.0 (fp-WSP); 21.0 (mp-WSP)
(Orumieh and Mazaheri, 2015)</td></tr>
<tr><td rowspan="3">Fecal coliforms</td><td>%</td><td>82-99.9 (Greenway, 2005); 65 (hybrid CW) to 99 (hssf-CW) (Vymazal, 2005); 44.4-96 (Hinds et al., 2012); 63.5-99.2 (hssf-CW) (Wu et al., 2016); 89.8-90.5 (hssf-CW) (Licata et al., 2017)</td><td colspan="2">99 (Tyagi et al., 2016)</td></tr>
<tr><td rowspan="2">Log_{10} reduction</td><td rowspan="2">1.0-2.0 (Oliveira, 2007); 0.5-3 (WHO, 2006);
2.0-3.0 (USEPA, 2002) (Garcia et al., 2013)</td><td>2.0-3.0 (USEPA; 2002); 0.8-1.0
(Liu et al., 2017)</td><td>Not available</td></tr>
<tr><td colspan="2">2-3.5 (Tyagi et al., 2016)</td></tr>
<tr><td>Helminth eggs</td><td>%</td><td>99.0 (Reinoso et al. (2008), Barbagallo et al., (2011))</td><td>>99 (Campos et al., 2002);
100 (Tyagi et al., 2016)</td><td>Not available</td></tr>
</table>

Table 2.3 – Technical comparison between the different wet technologies – secondary treatment systems (STS) (cont.).

			Construction wetlands (CW)	**Facultative wastewater stabilization ponds (fp-WSP)**	**Maturation wastewater stabilization ponds (mp-WSP)**
Costs	**Construction**	**€ $user^{-1}$**	43.00-650.00 (Seco, 2008); 270.00-378.40 (Gkika et al., 2014); 362.00-392.00 (Masi et al., 2006); 34.00-103.00 (Platzer et al., 2002); 580.00 (vssf-CW) and 730.00 (hssf-CW) (Mara, 2006); 171.00 (France) (vssf-CW), and 588.00 (vssf-CW) to 735.00 (hssf-CW) (Germany) (Mara, 2006)	93.50 (France) (Berland and Cooper, 2001); 340.00 (Mara, 2006); 105.00 (France) and 343.00 (Germany) (Mara, 2006)	
		€ $user^{-1}$ m^{-2}	0.20 (vssf-CW) to 0.29 (fws-CW) (Greece) (Taylor et al., 2010)	Not available	Not available
		€ m^{-2}	52.00-442.00 (Seco, 2008); 237.00-277.00 (Russeau et al., 2004); 101.00-129.00 (Masi et al.2006); 22.00-229.00 (Platzer et al., 2002)	Not available	Not available
		€ $system^{-1}$	50,000.00-90,000.00 (including septic tank) (Guerreiro, 2015)	Not applicable	Not applicable
	Operation	**€ m^{-3} $year^{-1}$**	0.01-0.07 (Crites and Ogden, 1998); 0.12-0.25 (Gkika et al., 2014); 0.17-0.28 (fws-CW), and 0.21-0.34 (vssf-CW) (USEPA, 2000); 0.03 (fws-CW) to 0.11 (vssf-CW) (Greece) (Taylor et al., 2010); 0.01-0.03 (Zhang et al., 2009); 0.63 (hssf-CW) (Mara, 2006)	$0.04x10^{-1}$ (Muga and Mihelcic, 2008)	Not available
		€ $user^{-1}$ $year^{-1}$	6.50-13.90 (Gkika et al., 2014); 1.20 (fws-CW), and 6.96 (vssf-CW) (Taylor et al., 2010); 26.00-71.00 (Spain) (Puigagut et al., 2007); 7.3-7.7 (Italy) (Masi et al., 2006); 3.5 (Mara, 2006); 5 (France), 0.73 (vssf-CW) (France), and 0.64 (hssf-CW) to 0.74 (vssf-CW) (Germany) (Mara, 2006)	0.50 (Germany) (Berland and Cooper, 2001); 0.58 (Mara, 2006); 0.98 (Germany) to 4.00 (France) (Mara, 2006)	
GHG emissions	**Methane ($mg\ m^{-2}\ hour^{-1}$)**		1.8-7.7 (fws-CW) (Johansson et al., 2003); 5.4 (vssf-CW) (Søvik et al., 2006); 5.9 (fws-CW), 2.9 (vssf-CW), and 6.4 (hssf-CW) (Mander et al., 2014)	Not available	25.0 (Hernandez-Paniagua et al., 2014)
	Nitrogen oxide ($mg\ m^{-2}\ hour^{-1}$)		0.031-0.192 (fws-CW) (Johansson et al., 2003); 0.150-0.420 (vssf-CW) (Inamori et al., 2008); 0.130 (fws-CW), 0.140 (vssf-CW), and 0.240 (hssf-CW) (Mander et al., 2014)	$5.0x10^{-4}$-$4.0x10^{-2}$ (Hernandez-Paniagua et al., 2014)	

Table 2.3 – Technical comparison between the different wet technologies – secondary treatment systems (STS) (cont.).

				Intermittent sand filters (int-SF)	Recirculating sand filters (rec-SF)	Peat filters (PF)	Vermifilters (VF)
Construction	HRT (hour)			Not available	Not available	Not available	1.0-3.0 (Sinha et al., 2008); 6.0-9.0 (Xing et al., 2005); 2.0-8.0 (Lourenço and Nunes, 2017a); 0.5 (Sinha et al., 2014)
	HLR ($m^3\ m^{-2}\ day^{-1}$)			0.04-0.08 (USEPA, 2002); Lesikar and Persyn, 2000); 0.04 (Siervers, 1998); 0.08-0.20 (USEPA, 1999a)	0.60 (Lesikar and Persyn, 2000); 0.10-0.20 (USEPA, 2002); 0.12-0.20 (USEPA, 1999b)	0.20 (Corley et al., 2006); 0.03-0.01 (Pundsack et al., 2000; Patterson, 1999)	0.2-4.0 (Li et al. 2009); 3 (Manyuchi et al., 2013); 2.0-3.0 (Xing et al., 2005); 0.67-2.67 (Lourenço and Nunes, 2017a); 1.5 (Kumar et al., 2015)
	OLR (g BOD $m^{-2}\ day^{-1}$)			25.0 (USEPA, 2002); 2.4-9.8 (USEPA, 1999a)	10.0-40.0 (USEPA, 2002); 9.8-39.1 (USEPA, 1999b)	Not available	5.5-22.2 (Lourenço and Nunes, 2017a); 7.4 (Lourenço and Nunes, 2017b)
	Surface required (m^{-2} $user^{-1}$)			Not available	Not available	Not available	0.6 (Soto and Tohá, 1998); 0.5 (Sinha et al., 2014)
	Dosing frequency (No. times day^{-1})			12-48 (USEPA, 1999a); 12-24 (USEPA, 2002); 4-18 (Wiesman, 2016)	48-120 or more (USEPA, 1999b); 12-120 (USEPA, 2002); ≥48 (Lesikar and Persyn, 2000)	Not available	Not available
	Recirculation ratio			Not applicable	3:1-5:1 (USEPA, 2002)	Not applicable	0.2-0.8 (Lourenço and Nunes, 2017a); 0.7 (Lourenço and Nunes, 2017b)
	W:D (hour)			Not applicable	Not applicable	Not applicable	1-3 (Bhise and Anaokar, 2015); 3-9 (Wang et al., 2014)
	Bed or packing depth (m)			0.60-0.90 (USEPA, 2002)		0.30-1.20 (Corley et al., 2006); 1.50 (Kõiv et al., 2009); 0.60 (Patterson, 2004)	0.10 (Arora and Kazmi, 2015); 0.15 (Kumar et al., 2015); 0.20 (Lourenço and Nunes, 2017a); 0.40 (Nie et al., 2015); 0.40-0.80 (Wang et al., 14)
	Media size (mm)			0.30-0.50 (Crites and Tchobanoglous, 1998); 0.25-0.75 (USEPA, 1999a)	0.25-0.30 (Boyle, 1995); Darby et al, 1996); 1.00-3.00 (USEPA, 1999b)	Not applicable	Not applicable
	Media uniformity coefficient			<4 (USEPA, 1999a; USEPA, 2002)	<4 (USEPA, 1999b)	Not applicable	Not applicable
	Earthworm abundance (g L^{-1})			Not applicable	Not applicable	Not applicable	10-40 (Zhao et al., 2010; Tomar and Suthar, 2011; Lourenço and Nunes, 2017a); 20 (Lourenço and Nunes, 2017b); 2.5-5.0 (Soto and Tohá, 1998); 25 (Manyuchi et al., 2013); 30 (Kumar et al., 2014)
	L:W			Not applicable	Not applicable	Not applicable	Not applicable
Operation	Efficiencies	BOD (%)		99.0 (Cagle and Johnson, 1994; Siervers, 1998)	98.0 (Piluk and Peters, 1994); 94.0 (Roy and Dube, 1994); 97.0 (USEPA, 1999b)	98.0 (Corley et al., 2006); >90.0 (Gunes and Ayaz, 1998)	86.0 (Arora et al., 2016); 97.5 (Lourenço and Nunes, 2017a); 99.0 (Soto and Tohá, 1998); 91.0-98.0 (Xing et al., 2005); >90.0 (Sinha et al., 2008); 99.0 (Cardoso-Vigueros et al., 2013)
		COD (%)		Not available	Not available	86.0 (Corley et al., 2006)	74.3 (Lourenço and Nunes, 2017a); 81.0-86.0 (Xing et al., 2005); 80.0-90.0 (Sinha et al., 2008); 92.0 (Cardoso-Vigueros et al., 2013); 72.0-88.0 (Furlong et al., 2014)
		TSS (%)		78.0 (Cagle and Johnson, 1994); 93.0 (Siervers, 1998)	89.0 (Piluk and Peters, 1994); 96.0 (Roy and Dube, 1994); 76.0 (USEPA, 1999b)	96.0 (Corley et al., 2006); >90.0 (Gunes and Ayaz, 1998), 43.0 (Patterson, 2004)	0 (Arora et al., 2016); 98.2 (Lourenço and Nunes, 2017a); 95.0 (Soto and Tohá, 1998); 97.0-98.0 (Xing et al., 2005); 90.0-95.0 (Sinha et al., 2008); 97.0 (Cardoso-Vigueros et al., 2013)
		NH_4^+ (%)		Not available	80.0 (USEPA, 1999b)	99.0 (Corley et al., 2006); 96.0 (Patterson, 2004)	90.0 (Arora et al., 2016); 88.1 (Lourenço and Nunes, 2017a); 76.0 (Kumar et al., 2015); 98.0 (Cardoso-Vigueros et al.,2013)
		TN (%)		39.0 (Cagle and Johnson, 1994); 18.0-33.0 (USEPA, 2002); 26.0 (Siervers, 1998)	53.0 (Piluk and Peters, 1994); 47.0 (Roy and Dube, 1994); 40.0-60.0 (USEPA, 2002)	22.0-67.0 (Gunes and Ayaz, 1998); 54.0 (Patterson, 2004)	60.2 (Wang et al., 2011); 46.0-73.0 (Cardoso-Vigueros et al., 2013)
		TP (%)		Not available	>20.0-30.0 (USEPA, 2002)	75.0 (Patterson, 2004)	82.0 (Wang et al., 2013)
		Fecal coliforms	%	99.9 (Cagle and Johnson, 1994); 99.9 (Siervers, 1998)	99.6 (Piluk and Peters, 1994); 97.3 (Roy and Dube, 1994); 99.0-99.9 (USEPA, 2002)	>99.0 (Patterson, 1999; Pundsack et al., 2000); >99.9 (Gunes and Ayaz, 1998)	Not available
			Log_{10} reduction	2.0 (Cagle and Johnson, 1994); 2.8 (Siervers, 1998)	2.4 (Piluk and Peters, 1994); 1.6 (Roy and Dube, 1994); 2.0-3.0 (USEPA, 2002)	Not available	2.8 (Arora et al., 2016); 2.7 (Arora et al., 2014a); 3.4 (Kumar et al., 2015); 3.7 (Lourenço and Nunes, 2017b)
		Helminth eggs	%	Not available	Not available	Not available	Not available

Table 2.3 – Technical comparison between the different wet technologies – secondary treatment systems (STS) (cont.).

			Intermittent sand filters (int-SF)	Recirculating sand filters (rec-SF)	Peat filters (PF)	Vermifilters (VF)
Costs	**Construction**	**€ $user^{-1}$**	Not available	Not available	Not available	85.00-122.00 (Soto and Tohá, 1998); 28.00-49.00 (land not included) (Sinha et al., 2014)
		€ $user^{-1}$ m^{-2}	Not available	Not available	Not available	Not available
		€ m^{-2}	Not available	Not available	Not available	Not available
		€ $system^{-1}$	5,800.00-12,500.00 (USEPA, 2002); 4,800 (Guerreiro, 2015)		170,00 (for peat prices) (Headley, 2006); 8,300.00-12,500.00 (USEPA, 2002)	0.04 (Sinha et al., 2014)
	Operation	**€ m^{-3} $year^{-1}$**	Not available		Not available	≤0.01 (Soto and Tohá, 1998); 0.05 (Sinha et al., 2014)
		€ $user^{-1}$ $year^{-1}$	40.00-250.00 (per system) (USEPA, 2002)		Not available	0.32-0.61 (Soto and Tohá, 1998)
GHG emissions	**Methane ($mg\ m^{-2}\ hour^{-1}$)**		0.3 (as mg $user^{-1}hour^{-1}$) (Truhlar *et al.*, 2016)		6.8 (as g m^{-2}) (Couwenberg, 2009)	0.58-1.71 (as a sink of methane), and 11.2 (emission) (Luth et al., 2011)
	Nitrogen oxide ($mg\ m^{-2}\ hour^{-1}$)		0.25 (as mg $user^{-1}$ $hour^{-1}$) (Truhlar *et al.*, 2016)		210.00 (as g m^{-2}) (Couwenberg, 2009)	0.35-77.00 (Luth et al., 2011)

2.4. Conclusions

Decentralized wastewater treatment should be economically affordable, environmentally sustainable, and socially acceptable for the management of excreta and wastewater. In general, decentralized wastewater treatment is typically used in sparse dwellings and small communities of rural areas, being progressively considered a more sustainable technology comparing with centralized wastewater treatment. Comparing with centralized wastewater treatment, decentralized wastewater treatment does not require energy or expensive or sophisticated operation, being easy to adapt to different geographic contexts.

A change to decentralized wastewater treatment could be essential to improve resiliency, efficiency, and environmental gains in rural areas, not only the recovery of resources in marginal lands, but also the demanding for treated drinking water and energy. A well-maintained wastewater treatment technology can contribute to the safe disposal of excreta and reuse of wastewater in agriculture avoiding the contamination of soil, surface water bodies and groundwater. Both Figure 2.4 and decision matrices shown in Tables 2.4 and 2.5 may help in choosing the most proper technology for future applications.

Dry controlled systems can be suited for arid regions or regions of rural areas without central water supply or sewerage, though its use has been gaining some momentum as more eco-houses are built in the developed countries (see decision Table 2.4). Although, currently, apart from the good level of knowledge regarding implementation and performance, technological transfer into practice from dry rudimentary systems and wet pits to dry controlled systems is still insufficient, and apart from the costs, general low awareness and the recognition of benefits persists (see decision Table 2.5). Even if every technology can theoretically be a sensible solution for sparse dwellings and small communities of different sizes and demographics, in general, the less developed the country, the more difficult is to implement dry technologies and dry controlled systems in particular. Unsurprisingly, dry controlled systems will still be more readily accepted in developed countries. Factors such as population status, level of development, environmental requirements (waste generated or production of greenhouse gases), weather, land availability, all can become crucial to select the more affordable technology (see decision Tables 2.4 and 2.5). Even if, in developed countries, mostly pre-existing centralized technologies and traditional technical approaches are the main obstacles to a more general diffusion of decentralized systems, dry controlled systems may be competitive alternatives.

Several dry controlled systems had been inappropriately dimensioned, born technologically obsolete, or been ineffective by inappropriate operation and maintenance, resulting in a lack of standardized or established design guidelines. Along with their higher construction and operating costs, it can explain why wet pits are still the main technology used in developing countries, and secondary treatment systems the main technology in developing countries. From all the technologies discussed, removal efficiencies varied widely. Also, climate change may difficult or change current efficiencies from wet technologies. Depending on each case, alternative concepts like constructed wetlands, packing bed filters, or wastewater stabilization ponds, can profitably be integrated in small dwellings or marginal lands of developing countries, as a part of more sustainable solutions (Tables 2.4 and 2.5). Such integration of current primary treatment systems with secondary treatment systems may increase efficiencies and reduce the sources pollution to soil and water.

Table 2.4 – Decision matrix for dry technologies.

		VIP	FA	DHP	CT	VT
Health and Hygiene		1	2	2	3	3
Environment and resources		1	2	2	3	3
Technology		4	4	4	3	3
Costs		4	3	3	2	2
Applicability	**Socio-economic**	4	3	3	1	1
	Dwelling size	1	3	3	2	2
	Climate	1	2	2	3	3

Legend: Very unfavorable (1), unfavorable (2), favorable (3), very favorable (4).

Table 2.5 – Decision matrix for wet technologies.

		WP	APY	SS	CW	PF	SF	VF	WSP
Health and Hygiene		1	2	3	3	4	3	4	4
Environment and resources		4	4	3	4	4	4	4	4
Technology		4	3	3	3	3	3	3	3
Efficiencies		1	2	3	2	4	4	4	3
Costs		4	3	3	3	2	2	3	3
Applicability	**Socio-economic**	4	4	3	3	3	3	3	3
	Dwelling size	4	4	3	3	3	3	4	4
	Climate	1	2	3	3	4	4	4	3

Legend: Very unfavorable (1), unfavorable (2), favorable (3), very favorable (4).

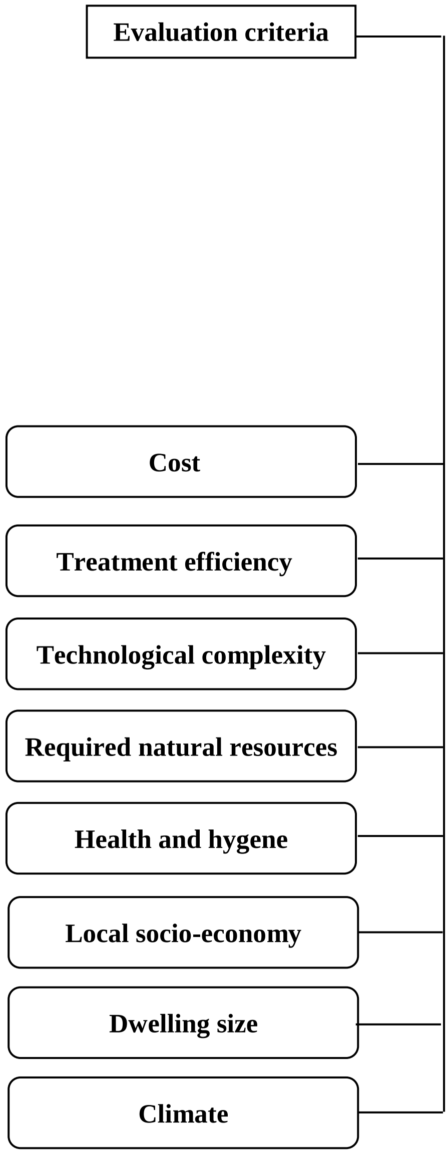

Figure 2.4 – Evaluation criteria for the different types of decentralized sanitation technologies.

Chapter 3

Optimization of a vermifiltration process for treating urban wastewater

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3.1. Introduction

Nearly 80% of water supplies used by society return as municipal wastewater carrying hazardous chemicals and high loads of organic matter (Sinha *et al*. 2008).

There is growing interest in developing environmentally safe and economically viable small-scale wastewater treatment technologies for onsite wastewater treatment, in particular for small communities and individual households. The most common configurations for onsite wastewater treatment are septic tanks, in some cases followed by passive soil absorption systems. When properly designed and installed these systems should provide sufficient treatment to prevent unacceptable groundwater contamination. Unfortunately, there are widespread evidences that frequently they do not meet this criterion. In the USA estimates indicate that only four to six percent of existing septic tanks are watertight (Nelson, 2005). Leaking septic tanks are in fact one the major widespread sources or contamination in many regions of the world (Fujita *et al*., 2013; Kuroda *et al*., 2012), and suspected of being in the origin of water-borne diseases (Borchardt *et al*., 2003).

Vermifiltration is a low-cost wastewater treatment process which is based on the same oxidation reactions, biodegradation and microbial stimulation by enzymatic action also present in vermicomposting and in trickling filters. In vermifiltration earthworms mechanical action creates aerobic conditions inside the reactor which help prevent the formation of odors. Dissolved and suspended organic and inorganic solids are trapped by adsorption and stabilization through complex biodegradation processes that take place in the filter packing, being subsequently used by microorganisms (Sinha *et al*., 2008). As in vermicomposting, in vermifiltration, epigenic earthworms convert the organic matter in wastewater into a suitable matrix filter – the vermicast. In vermicomposting, earthworms ingest near their own weight on organic matter each day and excrete 60% of this weight in castings (Lourenço, 2014). Their castings are rich in humus, nutrients and microorganisms, including nitrifying bacteria (Sinha, 2010) while having a hydraulic conductivity like sand with high adsorption properties. Earthworms and microorganisms cooperate to ingest and biodegrade organic wastes and contaminants present in wastewater. Their action improve filter permeability, increasing the degradation of the organic matter (Sinha *et al*., 2008; Arora *et al*., 2014b), hence promoting high removal efficiencies of BOD_5, COD and TSS from wastewater (Sinha *et al*., 2008).

In recent years, many studies on the physical process of vermifiltration have been published being extremely helpful for the sizing and managing of vermifilters (Xing, *et al*., 2011; Xing *et al*., 2010). The hydraulic retention time (HRT) and hydraulic loading rate (HLR) affect treatment and effluent quality. HRT determines the actual time the wastewater is in contact with the filter. It is proportional to the depth of the vermifilter (VF) witch may increase over time due to the accumulation of earthworm castings. HRT depends on wastewater flow rate, VF volume and type of material used. In principle, within certain ranges, the longer the wastewater remains inside the filter, the greater will be the BOD_5 and COD removal efficiency, but at the expenses of larger filter volumes. The main reason is that wastewater requires a certain contact time with the biofilm to allow for the adsorption, transformation, and reduction of contaminants (Hughes *et al*., 2006). HLR is an essential parameter in the design stage of all filters and determines the volume and amount of wastewater that a VF can reasonably treat in a given time. For a given system, higher HLR values will cause HRT to decrease and therefore reduce treatment efficiency. HLR may depend on parameters such as structure, effluent quality and filter packing bulk density, and method of effluent application (Siegrist, 1987). Common HRT values in vermifiltration systems range from 1 to 3 h (Sinha *et al*., 2008). As for HLR, the values commonly used have been between 0.2 $m^3\ m^{-2}\ day^{-1}$ (Li *et al.* 2009) and 3.0 $m^3\ m^{-2}\ day^{-1}$ (Manyuchi *et al*., 2013). Treatment efficiency is influenced by health, maturity and population abundance of earthworms. Abundance is a fundamental parameter for efficient running of vermifiltration (Li *et al*., 2009). Thus, to ensure an efficient treatment system, sufficient earthworm abundance is necessary. Different values are reported in literature usually in grams or number of individuals per volume of filter packing or surface area of filter packing. Common densities vary between 10 g L^{-1} and 40 g L^{-1} of *Eisenia fetida* (Bouché, 1972) per filter packing material (Tomar and Suthar, 2011; Zhao *et al*., 2010). Soto and Toha (1998) refer in their studies the use of 2.5 L^{-1} to 5.0 g L^{-1} with removal efficiencies of 99% for BOD and 96% for TSS. Manyuchi *et al*. (2013) used 25 g L^{-1} to treat domestic wastewater and obtained a removal of BOD_5 of 98% and COD of 70%. Kumar, *et al*. (2014) used a abundance of 30 g L^{-1} to treat domestic wastewater with BOD_5 and TSS final concentrations of 8.0±2.0 mg L^{-1} and 29±8.54 mg L^{-1}, respectively.

Over the past years, several studies have been made regarding the use of vermifiltration systems to increase treatment efficiency using different level stages (Wang *et al.*, 2011; Tomar and Suthar, 2011) as level stage systems provide excellent aerobic conditions for nitrification, increasing the removal rates for COD and NH_4^+.

Previous studies about vermifiltration systems have only primarily focused on the use of single tower VFs or combined vermifiltration processes in the treatment of different types of wastewater as very little work has been made so far using in-series VFs. Therefore, the objective of this study was to compare single stage VF with in-series VF stage alternative.

3.2. Material and methods

3.2.1. Reactor structure

Four filters without earthworms were tested in order to evaluate the best hydraulic properties on the removal of BOD_5, COD, TSS and NH_4^+ from urban wastewater. A vermifiltration process was carried to evaluate the best earthworm abundance on the removal of BOD_5, COD, TSS and NH_4^+ from urban wastewater based on the best hydraulic variables previously obtained, and using a single-stage VF. Yet, based on the best hydraulic variables and earthworm abundance, an in-series fourth-stage VF was further tested.

The reactor modules were made of PVC with a total volume of 25 L (Figure 3.1) following the treatment scheme found in literature (Taylor *et al.*, 2003; Lakshmi *et al.*, 2014). Experiments were made using vermicompost produced from municipal organic solid waste as the filtering material provided by a specialized company (FUTURAMB®). Vermicompost occupied the top 16.0 cm (average Ø < 0.1-3.0 mm), underneath which was installed an inert filter constituted of 6.0 cm of homogenized washed quartz sand (average Ø = 550 μm) on top of 7.0 cm of gravel (average Ø = 40 mm). The vermifilter was covered with a lid, leaving sufficient head space and opening as to allow natural aeration. An irrigation system was attached on top of the vermifilter made by a regular mesh. HDPE flexible pipes (Ø = 0.5 cm), separated 2.0 cm from each other, were used for wastewater irrigation and were kept 3.0 cm above the filter surface to ensure optimal wastewater distribution, the creation of drop-overflow and thereby increase aerobic conditions. Gravel applied on the bottom of the filter was separated from the equalizer by a stainless steel mesh (Ø = 0.4 cm). Quartz sand was separated from gravel and from vermicompost by a stainless steel mesh (Ø = 80 μm). Vermicompost physical-chemical characterization is shown in Table 3.1. The effluent from each VF was collected in the equalizer from where all samples were taken. From here, recirculation was made using a pump (Q_r) and mixed with raw wastewater as (Q_w) to be feed to the top of the filters ($Q_w + Q_r$).

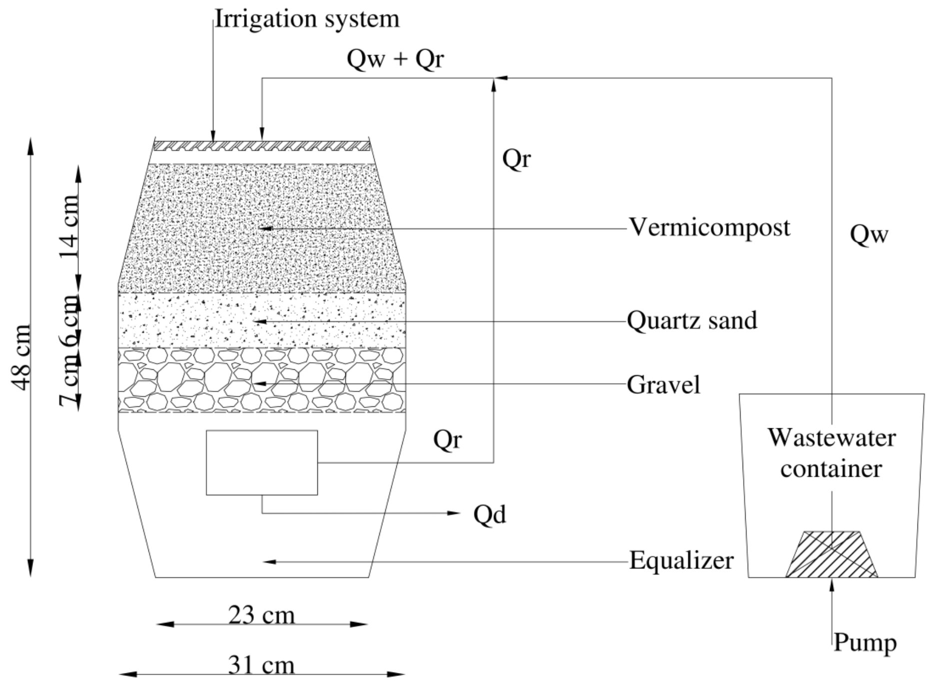

Figure 3.1 – Vermifilter unit design.

Table 3.1 – Characterization[1] of the vermicompost used as biological filter medium.

Parameter	Value
Bulk density (kg m^{-3})	600±0.00
Porosity (%)	73.7±0.30
Hydraulic conductivity (cm s^{-1})*	1.4±0.00
*p*H (H_2O)	6.82±0.01
EC (μS cm^{-1})	2,530±2.00
Organic matter (%)	56.48±0.01
TOC (%)	32.76±0.04
TN (%)	3.64±0.02
C/N ratio	9.0±0.03
TP (mg kg^{-1})	3,769±0.4
TK (mg kg^{-1})	7,150±0.08

[1] Mean concentration ± standard deviation.

[*] Obtained using a falling head permeability test according with Darcy's Law.

3.2.2. Acclimation of the vermifilter

Moisture content was held constant after placing the filters to field capacity following procedures used by the company that provided the earthworms, and the filter packing for acclimation of the earthworms. For this purpose filters were flushed with recirculating water for 30 days. After this time, VF was flushed and recirculated permanently for 45 days with wastewater collected from urban WWTP of Messines, Algarve, to allow the growth of heterotrophic microorganisms in the vermicompost. Each filter was fed, by pumping raw wastewater from a PVC container. The flow was also adjusted to permit the optimum moisture conditions for the survival of the earthworms.

3.3. Experimental design

a) Study of hydraulic variables

For this study, four HRT were tested: 2h, 4h, 6h and 8h (T2, T4, T6 and T8, respectively) (Table 3.2). Influent wastewater flow, Q_w and recycling flow, Q_r, were both adjusted to obtain a constant Q_{mix} equal to 1.50 $cm^3\ s^{-1}$, as this was the optimal flow for maintaining the ideal moisture in each filter. HRT was adjusted by changing the recirculation ration, $R = Q_r/Q_w$.

Table 3.2 – Hydraulic parameters for the different treatments.

Experiment	Q_{mix} ($cm^3\ s^{-1}$)	Q_r ($cm^3\ s^{-1}$)	Q_w ($cm^3\ s^{-1}$)	Q_r/Q_{mix}	HRT (h)	OLR (g BOD_5 day^{-1})	HLR ($m^3\ m^{-2}\ day^{-1}$)
T2	1.50	0.22	1.28	0.15	2	22.15	2.67
T4	1.50	0.86	0.64	0.57	4	11.08	1.33
T6	1.50	1.07	0.43	0.72	6	7.38	0.89
T8	1.50	1.18	0.32	0.79	8	5.53	0.66

Experiments were made for a period of 24 hours with continuous wastewater recirculation. Samples for chemical analysis were taken at the onset of the experiment from the influent wastewater and at the end of the treatment period from the treated effluent. HRT was measured in hours, OLR was measured in g BOD_5 day^{-1} and the HLR was measured in m^3 m^{-2} day^{-1} (equations (3.1), (3.2) and (3.3) respectively).

$$HRT = V/Q_w \qquad (3.1)$$

$$OLR = Q_w \times BOD_5 / A \qquad (3.2)$$

$$HLR = Q_w/A \qquad (3.3)$$

Where V (m^3) is the volume of the reactor, Q_w (m^3 day^{-1}) the influent wastewater flow rate, BOD_5 (mg L^{-1}) the organic matter concentration in influent wastewater and A (m^2) the reactor surface area.

b) Earthworm abundance study

Earthworm abundance was tested after all hydraulic parameters were optimized. Epigeic earthworm species are a key for decomposition of biomass due to their preference for organic rich substrates compared to the other species (Gajalakshmi *et al.*, 2001). In this study, earthworm's species chosen were *Eisenia fetida* (Bouché, 1972), in adult stage. The individuals were provided by a vermicomposting specialized company (FUTURAMB®) and previously installed on plastic boxes with coffee grounds at adequate moisture content for 15 days. No signs of disease and stress in the individuals were found.

Four treatments were made based on different earthworm abundance and according with an HRT of 6 hour: 10 (W10), 20 (W20), 30 (W30) and 40 (W40) g of earthworm per L of vermicompost. The different abundances were selected following the studies made by Tomar and Suthar (2011) and Zhao *et al*. (2010). The ratio of earthworms in the filter was also presented as weight of earthworms per weight of vermicompost (kg kg^{-1}) and volume of earthworms per volume of vermicompost (L L^{-1}) assuming a weight of 0.5 g and a volume of 1.0 cm^3 per earthworm respectively (Lourenço, 2014) (Table 3.3). A filter filled with the same packaging material, but without earthworms, acted as control (W0) (Figure A.1). Earthworms were placed on the top of the vermicompost and were allowed to install during an acclimation period of 15 days (Figure A.2).

Table 3.3 – Earthworm abundance in the experiments.

Experiment	Earthworm abundance (g L^{-1})	N.º kg^{-1}	N.º L^{-1}	kg kg^{-1} ratio	L L^{-1} ratio
W10	10	24	18	76	56
W20	20	48	36	38	28
W30	30	72	54	25	18
W40	40	96	72	19	14

Wastewater was applied continuously from a PVC wastewater container for 24 hours. All filters were frequently monitored for foul odors, smooth percolation of wastewater through the vermicompost, and clogging. General earthworm behavior, including agility, movement, stress and health conditions was also monitored. After this period, treated wastewater samples were collect for chemical analysis.

c) Sequential treatment study

In order to improve the quality of the treated wastewater and evaluate the earthworm influence in nitrification, the VF new configuration follow the studies made by Wang *et al*. (2011) who designed a sequential-stage VF to enhance the nutrients removal from rural domestic wastewater treatment. The VF consisted in a set of four equal reactors (A, B C and D) (Figure A.3), linked in sequence, with recirculation from the last back to the first (Figure 3.2). Since each reactor also had vermicompost as filtering material occupying the top 15.0 cm, with the total vermifiltration high *h* of the system being A=15 cm, A+B=30 cm, A+B+C=45 cm and A+B+C+D=60 cm. Homogenized washed quartz sand and gravel were kept with 6.0 and 7.0 cm high, respectively. The following hydraulic values were used during the experiment: Q_{mix}=1.5 $cm^3\ s^{-1}$; Q_r=1.07 $cm^3\ s^{-1}$; Q_w=0.43 $cm^3\ s^{-1}$; Q_r/Q_{mix}=0.72, NC=3.51, HRT=6h, HLR=0.89 $m^3\ m^{-2}$.day and OLR=177.6 g $BOD_5\ m^2\ day^{-1}$. Wastewater pumping and acclimation followed the procedures described for the first experiment.

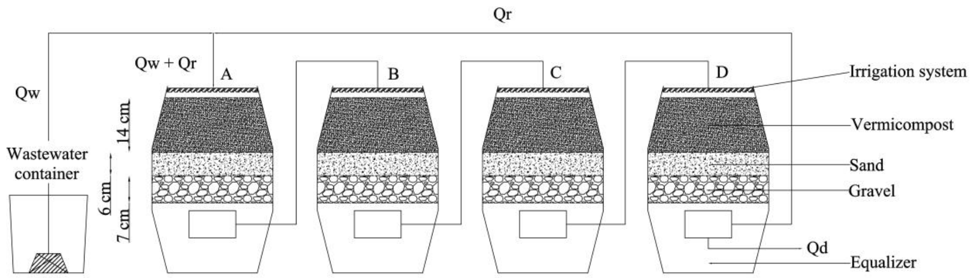

Figure 3.2 – Sequential vermifilter design.

3.4. Sampling and chemical analysis

For each treatment, samples were obtained at the beginning and at the end of the experiment. Samples of raw wastewater were taken from the feeding tank and samples of treated wastewater were taken from the equalizers (Figure 3.1 and Figure 3.2). All samples were analyzed immediately after sampling for BOD_5, COD and TSS. For this purpose, 5 L on each treatment were collected from the equalizer and three replicates were made for each parameter. *p*H and EC were analyzed using a HANNA HI98129 meter with a precision and range ±0.01 and 0.00-14.00 for *p*H, ±2% and 0.0-3,999 to μS cm^{-1} for EC and the later converted automatically by the equipment do TDS in the range 0-2,000 mg L^{-1}. BOD_5 was analyzed using an *OxiTop®-C respirometric system with incubation at constant temperature for 5 days (Eaton et al., 2005)* with a precision and range of ±1%. tCOD was analyzed with a photometer (NOVA 60, Merck) with a precision and range of ±5.0 mg L^{-1} and 25-1,500 mg L^{-1}) based on the permanganate method (Eaton *et al*., 2005). Dissolved COD (sCOD) was determined after filtration of the samples with a 40 μm filter paper and pCOD by the difference between tCOD and sCOD. TSS was determined by filtrating the sample through a Whatman® 40 μm cellulose filter paper, drying to a constant weight at 103 to 105 °C, and weighting (Eaton *et al*., 2005). NH_4^+ was quantified by photometry using a HANNA HI733 meter with a precision and range of ±0.1 mg L^{-1} and 0.0-99.9 mg L^{-1} respectively.

3.5. Statistical analysis

Statistical analysis was carried out for data interpretation. One-way analysis of variance (ANOVA), followed by Tukey test at a significance level of $\alpha = 0.05$ were made to check for differences between treatments. SPSS® 17.0 was used in the analysis.

3.6. Results and discussion

3.6.1. Wastewater characterization

The wastewater used in the study came from the urban WWTP of Messines, Algarve, with a served population of 6,000 inhabitants which receives wastewater from a combined sewage collection system designed to transport both rainwater and sewage together. All samples were collected on November 25th after the preliminary wastewater treatment. Wastewater used in the study was the same wastewater used in all experiments (Figure A.4). No rain was registered during the days before wastewater collection. Wastewater physical-chemical and microbiologic characterization is shown in Table 3.4.

Table 3.4 – Characterization of influent wastewater and comparison with the literature data.

Parameter	Value	Henze (2008)	USEPA (2002)	Metcalf & Eddy (2003)
*p*H	8.21±0.02	n.a.	n.a.	n.a.
EC (μS cm^{-1})	960±2.31	n.a.	n.a.	n.a.
BOD_5 (mg L^{-1})	93.3±2.89	350	221	190
tCOD (mg L^{-1})	251±2.31	750	580	430
sCOD (mg L^{-1})	102±2.00	300	n.a.	n.a.
pCOD (mg L^{-1})	149±1.15	450	n.a.	n.a.
TDS (mg L^{-1})	479±1.15	n.a.	n.a.	500
TSS (mg L^{-1})	94.57±0.67	400	243	210
NH_4^+ (mg L^{-1})	46.37±0.23	45	9	25
BOD_5/tCOD	0.37±0.02	0.47	0.38	0.44
sCOD/tCOD	0.41±0.01	0.40	n.a.	n.a.
pCOD/tCOD	0.59±0.00	0.60	n.a.	n.a.
tCOD/NH_4^+-N**	5.42±0.08	16.7	64.4	17.2

[1] Mean concentration ± standard deviation.
n.a.: Not available.
*Average concentration.
** NH_4^+ was converted into NH_4^+-N stoichiometrically.

The ratio BOD_5/COD is one important way to assess the biodegradability of wastewater, as in a raw urban wastewater the BOD_5/COD ratio varies between 0.3 and 0.8 (Tchobanoglous *et al.*, 2003). With a BOD_5/COD ratio of 0.5, wastewater will consider to be easily treatable by biological processes (Tchobanoglous *et al*., 2003). In a typical urban wastewater, BOD_5, COD and TSS have average concentrations of 350, 750 and 400 mg L^{-1} respectively (Henze and Comeau, 2008) (Table 3.4). Comparing our results with the results from literature (Table 3.4), BOD_5 (93.3±2.89 mg L^{-1}), tCOD (251±2.31 mg L^{-1}) and TSS (94.57±0.67 mg L^{-1}) were all lower. Also, according with literature, our wastewater proved to be lower than the 0.5 ratio appropriate to biological treatment. These two aspects could be justified by the sedimentation of solids in the PVC container during the study.

When treating wastewater, it is usually stated that C/N ratio in raw wastewater should be near 10:1 for aerobic treatment (USEPA, 1995; Maier, 1999; Tchobanoglous *et al*., 2003; Henze and Comeau, 2008). C/N ratio in wastewater is related with the COD/NH_4^+-N ratio as microorganisms can use COD as carbon compounds. Nitrogen is present mainly as organic nitrogen and NH_4^+. Cardoso-Vigueros *et al*. (2013) indicated that to increase N removal from wastewater by nitrification it should be studied the C/N ratio in raw wastewater. The C/N values obtained in our study were lower than 10:1 (Table 3.4). NH_4^+ concentration (46.4 mg L^{-1}±0.26) was similar with values referred by (Henze and Comeau, 2008) values (45 mg L^{-1}) as this supports that this parameter could be mainly from domestic sources (e.g. urine or cleaning agents). Hughes *et al*. (2008) reported that near 75% of the TN in a typical urban wastewater is NH_4^+ and the majority (70–90%) comes from urine, whilst the final 20% comes from cleaning agents, disinfectants and food wastes. Nutrient concentration (as carbon and nitrogen) reported in this study was lower than that reported by literature.

3.6.2. Study of hydraulic variables

In all treatments, biological organic matter removal (as BOD_5) from wastewater was generally positively affected by HRT, with 9.3 mg L^{-1}±0.58 in T2, 6.0 mg L^{-1}±0.00 in T4 and 3.3 mg L^{-1}±0.58 in T6. The highest BOD_5 removal efficiency was registered in T6 with 96.4%±0.62. Both concentration and removal efficiency in T6 attain EU standards for urban wastewater discharge (Directive 91/271/EEC, 21th May, 1991) of 25 mg L^{-1} and 70-90%, respectively.

tCOD was also positively affected by HRT with 86.7 mg L^{-1}±4.2 in T2 and 68.7 mg L^{-1}±4.2 in T4 but T6 and T8 didn't improve the wastewater treatment as tCOD final concentration in T6 was 70.0 mg L^{-1}±7.2 and 83.3 mg L^{-1}±4.2 in T8. sCOD register the lowest concentration and the best removal at T6 with 62.7 mg L^{-}±1.2 and 38.6%±1.14. tCOD removal efficiency was not as significant as that of BOD_5 since this parameter is related with the non-biological organic matter removal. pCOD was removed more efficiently than sCOD, as a significant amount of the sCOD was still present in wastewater. Anusha and Sundar (2015) studied the COD removal efficiency using a VF with gravel, sand and garden soil as filter packing and obtained the highest removal rates at 6h HRT with 65.9% as the values obtained in our study at HRT=4h, HRT=6h and HRT=8h (as tCOD) proved to be higher (72.7%±1.66, 72.2%±2.87 and 66.8%±1.65).

The lowest TSS values were obtained for HRT=6h (T6). BOD_5 and TSS removal could be explained by the fact that part of the solids was accumulated in vermicompost and digested by earthworms. Using vermicompost as filter packing, (Kumar *et al.*, 2014) studied the TSS removal in a VF and related the HRT with the lack of availability of time to degrade the solids in the reactor. This could justify the lower BOD_5, sCOD and TSS removal efficiencies in T2.

Typically, in a treated wastewater the BOD_5/COD ratio varies between 0.1 and 0.3 (Meireles, 2011). The lowest BOD_5/tCOD ratio was obtained in T6 and T8 with 0.05±0.01. This suggests that apparently all biological organic matter was removed in T6 and T8. Besides, the values obtained in all experiments led to a wastewater with a difficult subsequent biological treatment since the BOD_5/COD ratio obtained was already lower than 0.5 (Tchobanoglous *et al*, 2003).

The increase found in T8 (4.0 mg L^{-1}±0.00 for BOD_5, 83.3 mg L^{-1}±4.2 for COD and 0.3 mg L^{-1}±0.1 for TSS) may be due to some release of biodegradable and recalcitrant materials from the filter packaging itself.

Organic loading rate is defined as the application of soluble and particulate organic matter (as BOD_5) per unit area per unit time (Otis, 2001; Siegrist, 1987). As HRT increased from T2 to T8 OLR decreased from 533.1 g BOD_5 m^{-2} day^{-1} to 133.1 g BOD_5 day^{-1} (Table 3.2) due to the decrease of organic matter applied per unit of time. During the study, the best HRT obtained was 6 h (T6) with an OLR of 177.6 g BOD_5 m^{-2} day^{-1}.

Wastewater treatment at HLR of 2.67 m^3 m^{-2} day^{-1} had the lowest removal efficiencies (90.0%±0.62 for BOD_5, 65.5%±1.65 for tCOD and 95.2%±0.11 for TSS). This could be related to the fact that when the HRT decreased the HLR increased and so increased the volume of wastewater per unit of filter area. Kumar *et al*. (2014) illustrated the effect of hydraulic loading rates on the wastewater treatment in VFs working with hydraulic loading rates of 1.5, 2.0, 2.5 and 3.0 m^3 m^{-2} day^{-1}, having concluded that the optimum result was 2.5 m^3 m^{-2} day^{-1}.

Table 3.5 – Physical-chemical parameters in treated wastewater and treatment efficiencies.[1]

	Experiment				**EU regulation**	
Parameter	**T2**	**η (%)**	**T4**	**η (%)**	**C_{limit} (mg L^{-1})**	**η_{limit} (%)**
BOD_5 (mg L^{-1})	9.3^{c}±0.58	90.0^{a}±0.62	6.0^{b}±0.00	93.6^{b}±0.00	25	70-90
tCOD (mg L^{-1})	86.7^{b}±4.2	65.5^{a}±1.65	68.7^{a}±4.2	72.7^{b}±1.66	125	75
sCOD (mg L^{-1})	83.3^{b}±3.5	18.3^{a}±4.08	68.0^{a}±2.0	33.3^{b}±1.86	-	-
pCOD (mg L^{-1})	3.33^{a}±2.31	97.8^{a}±1.55	2.67^{a}±1.15	98.2^{a}±0.77	-	-
TSS (mg L^{-1})	4.5^{c}±0.1	95.2^{a}±0.11	1.8^{b}±0.15	98.0^{b}±0.16	35	90
BOD_5/tCOD	0.11^{b}±0.01	-70.9^{b}±3.25	0.09^{b}±0.01	-76.4^{b}±1.47	-	-
sCOD/tCOD	0.96^{a}±0.09	137.7^{a}±23.2	0.99^{a}±0.06	144.6^{a}±15.36	-	-
Parameter	**T6**	**η (%)**	**T8**	**η (%)**	**C_{limit} (mg L^{-1})**	**η_{limit} (%)**
BOD_5 (mg L^{-1})	3.3^{a}±0.58	96.4^{c}±0.62	4.0^{a}±0.00	95.7^{c}±0.00	25	70-90
tCOD (mg L^{-1})	70.0^{a}±7.2	72.2^{bc}±2.87	83.3^{b}±4.2	66.8^{ab}±1.65	125	75
sCOD (mg L^{-1})	62.7^{a}±1.2	38.6^{b}±1.14	78.0^{b}±0.00	23.5^{a}±0.00	-	-
pCOD (mg L^{-1})	7.33^{a}±7.02	95.1^{a}±4.70	5.33^{a}±4.16	96.4^{a}±2.79	-	-
TSS (mg L^{-1})	0.1^{a}±0.01	99.9^{c}±0.01	0.3^{a}±0.1	99.7^{c}±0.11	35	90
BOD_5/tCOD	0.05^{a}±0.01	-87.1^{a}±2.17	0.05^{a}±0.00	-87.1^{a}±0.63	-	-
sCOD/tCOD	0.90^{a}±0.09	122.2^{a}±22.86	0.94^{a}±0.05	131^{a}±11.3	-	-

1 Mean concentration ± standard deviation. Values followed by the same letter within each column are not different (ANOVA, dg=(3,8); Tukey's test, dg=(4,8); α=0.05).
C_{limit}: Concentration limit.
η_{limit}: Efficiency limit.

3.6.3. Earthworm abundance study

Earthworms showed good survival rates during all the experiment and no stress signals were found (Figure A.5), which could be due to the fact that epigenic earthworm species have natural ability to colonize organic wastes; high rates of consumption, digestion and assimilation of organic matter and tolerance to a wide range of environmental factors (Edwards *et al*, 2011). On experiments with 30 and 40 g L^{-1} (W30 and W40) the process was interrupted at the end of five hours due to the clogging of the VF. In such experiments, no samples were collected for analysis, as the total wastewater volume in the equalizer was not sufficient. This could be justified by the higher earthworm abundances in these experiments who produced large amounts of castings and reduced medium permeability since *Eisenia fetida* earthworm ingest near its own weight daily in waste (Lourenço, 2010; Appelhof and Fenton, 2003). Mucus production associated with water excretion in earthworm guts increases the activity of soil microorganisms – producing organic matter as vermicompost, which may have contributed to decrease filter permeability in W30 and W40.

Carbon is the principal component of organic substances found in wastewater and is biodegraded under aerobic conditions (Winkler, 1997). In wastewater, microorganisms can use COD as carbon compounds to build their cell structures and generate energy. Earthworms act together with aerobic microorganisms to accelerate and enhance the decomposition of organic matter (Daven and Klien, 2008) by promoting enzymatic activity and faster biochemical reactions. Arora *et al*. (2016) using an abundance of earthworms of 30.0 g L^{-1}, obtained removal efficiencies from 85.5% in BOD_5 and 77.8% in COD. Malek *et al*. (2012), with an abundance of 40 g L^{-1}, obtained removal efficiencies of 78-90% for COD and 82-96% for TSS. The results obtained showed efficiencies for BOD_5 of 98.6%±0.6 in W10 and 97.5%±0.6 in W20 and removal efficiency for tCOD of 66.0%±1.2 in W10 and 74.3%±1.2 in W20. TSS showed removal efficiencies of 99.5%±0.1 in W10 and 98.2% ± 0.1 in W20.

The increase in earthworm abundance contributed to reduce BOD_5 and TSS efficiencies but to increase tCOD removal efficiency ($p<0.05$). The presence of earthworms intensifies the organic loadings of wastewater since they increase the permeability of the filter. Some studies report that earthworm abundance is not a particularly relevant parameter when treating wastewater with low organic loads, as abundances may adjust naturally to influent organic loads (Reinecke and Viljoen, 1990). Results showed that pCOD was the main COD fraction removed by earthworms, which may be attributable to the ingestion and degradation of solid particles trapped in vermicompost (Sinha *et al*. 2010). tCOD, pCOD and sCOD fractions also decreased in W10, contributing to increase efficiencies. tCOD and pCOD fractions were significantly lower in W20 comparing to W10 (Table 3.6) (Tukey test, $p<0.05$) but no statistically significant difference was obtained for sCOD (Table 3.6) (Tukey test, $p>0.05$). Cardozo-Vigueros *et al*. (2013), obtained for an urban wastewater vermifiltration process obtained 99%, 92% and 97% for BOD_5, COD and TSS removal efficiencies, respectively. Also treating urban wastewater, Xing *et al*. (2005) reported removal efficiencies for BOD_5 between 91-98%, COD between 81-86% and TSS between 97-98%. Comparing with our results, BOD_5 and COD removal efficiencies were all higher. As for TSS removal efficiencies, our results proved to be higher for T6 and T8 (Table

3.5). Earthworms action improve filter permeability, increasing the degradation of the organic matter (Sinha *et al*., 2008; Arora *et al*., 2014b), promoting removal efficiencies of BOD_5 by over 90%, COD by 80–90% and TSS by 90–95% from wastewater (Sinha *et al*., 2008). The lower sCOD removal efficiency in W20, when compared comparing to W10, may be due to the dissolution of humic substances presented in vermicastings (Edwards *et al*., 2011).

NH_4^+ removal efficiency decreased as earthworm abundance increased from W10 to W20 (Table 3.6). This result may be related to the mineralization of the vermicompost, promoted by the earthworms and microorganisms, as they promote microbial activity. Besides, the major role of earthworms in nitrogen cycling lies in their ability to increase the mineralization rate of organic nitrogen (Binet and Trehen, 1992; Willems *et al*., 1996; Bansal and Kapoor, 2000; Lachnicht and Hendrix, 2001; Osler and Sommerkorn, 2007). According to Sinha *et al*. (2008), earthworms excrete polysaccharides, proteins, and other nitrogenous compounds as they mineralize the nitrogen in wastewater to make it available to plants as nutrients, while excreting NH_4^+ (Whalen *et al*., 2000).

Table 3.6 – Effluent parameters[1] for the different treatments.

Parameter	Experiment					
	W0	***η* (%)**	**W10**	***η* (%)**	**W20**	***η* (%)**
BOD_5 (mg L^{-1})	2.0^a±0.0	97.9^a±0.0	1.3^a±0.6	98.6^a±0.6	2.3^a±0.6	97.5^a±0.6
tCOD (mg L^{-1})	91.3^b±2.3	63.7^a±0.9	85.3^b±3.1	66.0^a±1.2	64.7^a±3.1	74.3^b±1.2
sCOD (mg L^{-1})	50.0^a±5.3	51.0^a±5.2	47.3^a±1.2	53.6^a±1.1	51.3^a±4.2	49.7^a±4.1
pCOD (mg L^{-1})	41.3^b±7.0	72.3^a±4.7	38.0^b±3.5	74.6^a±2.3	13.0^a±6.4	91.1^b±4.3
TSS (mg L^{-1})	0.5^a±0.06	99.4^b±1.8	0.4^a±0.06	99.5^b±0.1	1.7^b±0.08	98.2^a±0.1
NH_4^+ (mg L^{-1})	1.4^a±0.29	97.1^c±0.8	2.4^b±0.15	94.8^b±0.4	5.5^c±0.26	88.1^a±0.7

1 Mean concentration ± standard deviation. Values followed by the same letter within each column are not different (ANOVA, dg=(2,6); Tukey's test, dg=(3,6); α=0.05).

BOD_5/tCOD ratio was lower in treated wastewater in W0 and W10, due to the removal of BOD_5, but higher in W20, mainly due to the relatively higher efficiency in removing also tCQO (Table 3.7). pCOD fraction was significantly decreased by earthworm activity, as shown by an increase in sCOD/tCOD as earthworm abundance increased (Table 3.7). Earthworms improve the total specific surface area of filter packing, enhancing the ability to adsorb organic and inorganic substances from wastewater (Sinha, 2010).

In our study, VF contributed to decrease the tCOD/NH_4^+-N ratio, being the lowest value obtained in W20 with 15.1±1.6 (Tukey test, $p<0.05$, Table 7). The values of the ratio were lower than 57.14 which Wang *et al*. (2015) found to be the desirable tCOD/NH_4^+-N after an efficient nitrification process.

Table 3.7 – Effluent parameters[1] for different treatments.

Parameter	Raw wastewater	Experiment		
		W0	**W10**	**W20**
BOD_5/tCOD	0.37±0.02	0.02^a±0.00	0.02^a±0.01	0.04^b±0.01
sCOD/tCOD	0.41±0.01	0.55^a±0.07	0.56^a±0.03	0.80^b±0.09
pCOD/tCOD	0.59±0.01	0.45^b±0.07	0.44^b±0.03	0.20^a±0.09
sCOD/pCOD	0.68±0.01	1.25^a±0.36	1.25^a±0.14	5.04^a±3.72
tCOD/NH_4^+-N	6.97±0.11	89.6^b±22.00	45.8^a±2.17	15.1^a±1.56

1 Mean concentration ± standard deviation. Values followed by the same letter within each column are not different (ANOVA, dg=(2,6); Tukey's test, dg=(3,6); α=0.05).

In conclusion, treatment efficiencies in W10 and W20 though statistically different for all parameters, but BOD_5 and sCOD are for efficiencies and effluent quality well above EU standards for urban wastewater discharge (Table 3.6). Besides, the abundance of earthworms in the filters will eventually converge to a higher density if conditions are favorable, which seems to be the case given the results obtained for W20. For these reasons an abundance of 20 g L^{-1} was used in the subsequent analysis.

3.6.4. Sequential vermifiltration study

BOD_5 decreased from A to C with a decrease from 3.3±0.67 mg L^{-1} to 1.0±0.00 mg L^{-1} (Table 3.8). Along the sequence of filters, no differences were found in sCOD and pCOD fractions as it was obtained 39.3 mg L^{-1}±1.33 in A, 38.7 mg L^{-1}±1.76 in B, 42.0 mg L^{-1}±2.00 in C and 45.3 mg L^{-1}±1.76 in D, and 30.7 mg L^{-1}±7.57 in A, 20.0 mg L^{-1}±2.00 in B, 19.7 mg L^{-1}±4.04 in C and 20.0 mg L^{-1}±3.46 in D and (Table 3.8) (Tukey test, $p>0.05$). The efficiency for TSS in sequential system (96.6%±1.95) was lower comparing to single system (99.5%±0.1 for W10 and 98.2%±0.1 for W20).

NH_4^+ concentration decreased over the sequential of filters, as the final reduction efficiency obtained was 99.1%±0.0 in D (Table 3.8). Wang *et al.* (2011) also obtained a removal of NH_4^+ of near 98% after the VFs last stage. Nitrification is the process by which NH_4^+ or NH_3 are oxidized into NO_2^- by *Nitrosomonas* spp., and the NO_2^- further oxidized into nitrate NO_3^- by nitrite-oxidizing bacteria, often *Nitrobacter* spp.. A very important fraction of NH_4^+ is eliminated through adsorption by biomass and nitrification processes, carried out by aerobic autotrophic bacteria (Tunira and Saxena, 2010). At the same time, Gupta (2013) reported an increase in bacteria diversity in wastewater due to the presence of earthworms (*Eisenia fetida*), especially in response to nutrients in their castings. Wu *et al*. (2014) refer in their studies that nitrification was affected significantly by the aeration rates.

Table 3.8 – Treated wastewater parameters[1] with the different treatments.

VF	**BOD_5 (mg L^{-1})**	***η* (%)**	**tCOD (mg L^{-1})**	***η* (%)**	**sCOD (mg L^{-1})**	***η* (%)**
A	3.3^{b}±0.15	96.3^{a}±1.28	70.0^{b}±5.29	72.4^{a}±2.08	39.3^{a}±2.31	62.2^{a}±2.22
B	1.7^{ab}±0.58	98.2^{a}±0.64	58.7^{ab}±1.15	76.9^{b}±0.46	38.7^{a}±3.06	62.8^{a}±2.93
C	1.0^{a}±0.00	98.9^{b}±0.00	61.3^{ab}±4.62	75.8^{ab}±1.82	42.0^{a}±3.46	59.6^{a}±0.00
D	1.3^{a}±0.58	98.5^{b}±0.64	65.3^{b}±1.15	74.3^{ab}±0.45	45.3^{a}±3.06	56.4^{a}±0.00
VF	**pCOD (mg L^{-1})**	***η* (%)**	**TSS (mg L^{-1})**	***η* (%)**	**NH_4^+ (mg L^{-1})**	***η* (%)**
A	30.7^{a}±7.57	79.6^{a}±5.05	7.7^{c}±0.87	91.9^{a}±0.91	6.2^{c}±0.12	86.6^{a}±0.43
B	20.0^{a}±2.00	86.7^{a}±1.34	4.3^{ab}±0.28	95.5^{bc}±0.29	3.6^{b}±0.23	92.2^{b}±0.87
C	19.7^{a}±3.33	87.1^{a}±2.77	5.7^{b}±0.63	94.0^{b}±0.66	1.3^{a}±0.22	97.3^{c}±0.82
D	20.0^{a}±2.94	86.7^{a}±2.31	3.2^{a}±0.28	96.6^{c}±1.95	0.4^{a}±0.23	99.1^{c}±0.88

[1] Mean concentration ± standard deviation. Values followed by the same letter within each column are not different (ANOVA, dg=(3,8); Tukey's test, dg=(4,8); α=0.05).

Tomar and Suthar (2011) used a two stage vermifiltration process and reported that NH_4^+ concentration underwent a strong decline between 5 and 35 cm filter depth, and between 35 cm and 65 cm filter depth. Comparing with conventional VF models, in our study, the wastewater applied above the filter surface (from A to D) could have increased aerobic conditions, and promoted dissolved oxygen concentration with further nitrification.

BOD_5/COD ratio decreased along the sequence of filters which can indicate that organic fractions in wastewater were gradually removed. Starting from an initial BOD_5/COD ratio of 0.37 (Table 3.4), the values obtained led to a wastewater with a difficult subsequent biological treatment (0.05±0.02 in A, 0.03±0.01, 0.02±0.00 and 0.02±0.01 (Table 3.9). Results also showed that, along the sequence of reactors, pCOD/tCOD did not significantly decreased and the treatment was not improved (0.43±0.08 in A, 0.34±0.04 in B, 0.32±0.05 in C and 0.30±0.05 in D). The decrease in NH_4^+ concentration was accompanied by an increase in COD/NH_4^+-N ratio (11.48±1.03 in A, 21.11±2.26 in B, 64.07±15.09 in C and 135.83±41.25 in D) (Tukey test, $p<0.05$). Fang *et al*. (2010) refer that TN removal by nitrification may be enhanced when increasing the carbon to nitrogen (as COD/NH_4^+-N) ratio in wastewater. COD/NH_4^+-N increased along the reactors with 14.48±1.03 in A, 21.11±2.26 in B, 64.07±15.09 in C, ending with 135.83±41.25 in D (Table 3.9) indicating better nitrification conditions when compared to the single reactor system.

Table 3.9 – Chemical ratios[1] in the reactors.

Reactor	BOD_5/tCOD	sCOD/tCOD	pCOD/tCOD	COD/NH_4^+-N
A	0.05^{b}±0.02	0.57^{a}±0.08	0.43^{a}±0.08	14.48^{a}±1.03
B	0.03^{ab}±0.01	0.66^{a}±0.04	0.34^{a}±0.04	21.11^{ab}±2.26
C	0.02^{b}±0.00	0.69^{a}±0.05	0.31^{a}±0.05	63.82^{b}±15.45
D	0.02^{b}±0.01	0.69^{a}±0.05	0.31^{a}±0.05	135.83^{c}±41.25

[1] Mean concentration ± standard deviation. Values followed by the same letter within each column are not different (ANOVA, dg=(3,8); Tukey's test, dg=(4,8); α=0.05).

In what concerns the impacts of VF due to atmospheric emissions, published results are contradictory. NH_4^+ is generated by organic nitrogen mineralization leading to ammonia emissions which is the first inorganic nitrogen form produced during biological wastewater treatment (Henze, 2008). Wüst *et al*. (2009) refer that earthworms enhance nitrous oxide emissions through gut-associated denitrification. For Luth (2011) high earthworm population were associated with the absence of ammonia emissions and reduced nitrous oxide emissions. Apart from acting as direct sources or sinks of greenhouse gases, earthworms have an indirect influence on the net emission of the organic substrate through the microbial community (Binet *et al*., 1998; Aira *et al*., 2007).

3.7. Conclusions

The best removal efficiencies were obtained with a hydraulic retention time of 6 hours, 0.89 m^3 m^{-2} day^{-1} hydraulic loading rate and 177.6 g BOD_5 m^2 day^{-1}. Using traditional VF models, an earthworm abundance of 10 g L^{-1} proved to be the most effective, being also possible to attain the EU standards for urban wastewater discharge. Comparing with traditional VF models, the four-stage sequential vermifilter increased BOD_5 and NH_4^+ efficiencies and consequent nitrification.

Chapter 4

The role of filter packing in vermifiltration

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4.1. Introduction

Due to increasing world population, the natural resources are under increasing stress which fosters wastewater reuse planning and emphasizes on the decentralized wastewater treatment, especially in rural areas where high wastewater collection and treatment costs does not justify the installation of conventional wastewater treatment plants (WWTP) (Prasad and Kumar, 2012). Decentralized wastewater treatment systems involve the collection, treatment, disposal and reuse of wastewater from households, clusters of homes and isolated communities, at or near the point of generation (Li *et al*., 2009). Commonly, wastewater treatments must be able to reduce organic matter and nutrients concentration, but also promote *faecal* microorganisms' elimination (George *et al*., 2002).

Vermifiltration is a bio-oxidative process in which earthworms interact intensively with microorganisms within the decomposer community, increasing the stabilization of organic matter and greatly modifying its physical and biochemical properties (Liu *et al*., 2012), combining filtration processes with vermicomposting techniques.

Earthworm species and filter media types are crucial influencing factors for the removal efficiency of vermifiltration because they are considered as the main biological components of the process and can change directly or indirectly the main removal processes of contaminants over time (Sinha, 2010).

The design parameters of vermifilters (VFs) include stocking density of earthworms (Sinha *et al.*, 2008), filter media composition (Cardoso-Vigueros *et al*., 2013), hydraulic loading rate (HLR) (Kumar *et al*., 2015) and hydraulic retention time (HRT) (Arora *et al*., 2014a; Arora *et al*., 2016). Studies have been made with earthworm densities of 10 g L^{-1} (Arora *et al*., 2014b), 30 g L^{-1} (Arora *et al*., 2016) and intermediate values of 22.0 to 24.5 g L^{-1} (Tomar and Suthar, 2011) (Table 4.1). Typical HRT varies between 6 and 9 h and HLR between 2.0 and 3.0 m^3 m^{-2} day^{-1} (Xing *et al*., 2005).

Table 4.1 – Reported wastewater type, origin and operational parameters.

Parameter	Sinha *et al.* (2008)	Cardoso-Vigueros *et al.* (2014)	Arora *et al.* (2014b)	Arora *et al.* (2016)	Kumar *et al.* (2015)	Tomar and Suthar (2011)
Wastewater type	Municipal wastewater	Domestic wastewater	Synthetic wastewater	Synthetic wastewater	Synthetic wastewater	Urban wastewater
Wastewater origin	WWTP	Toilets	Locally produced	Locally produced	Locally produced	Wastewater stream
Filter packing material	Garden soil, gravel	Domestic organic wastes, vermicompost, volcanic stones, gravel	Vermicompost, sand and gravel	Vermicompost, riverbed gravel, gravel	Vermicompost, riverbed gravel	Stones, sawdust, dried leaves, soil mixed with stones and pebbles
Earthworms species	Mix of *Eisenia fetida*, *P. excavatus* and *Eudrillus euginae*	*Eisenia* spp.	*Eisenia fetida*	*Eisenia fetida*	*Eisenia fetida*	*Perionyx sansibaricus*
Stock density of earthworms (g L^{-1})	10	10	18	30	16.5	22-24.5
HRT (h)	1-2	0.18	-	6	-	-
HLR (m^3 m^{-2} day^{-1})	-	-	1.3	1.0	2.5	-

Vermifilter packing material is an important design parameter for maximizing the treatment efficiency (Arora *et al.*, 2014b). Filter medium materials should facilitate natural aeration (Cardoso-Vigueros *et al.*, 2013) and also serve as a dwelling habitat for earthworms to thrive and perform their function proficiently. Common filter packing materials include vermicompost (Arora *et al.*, 2016), wood chips, bark, peat, straw (Li *et al.*, 2008) and sawdust (Lourenço and Nunes, 2017) for organic packing, and gravel, quartz sand (Lourenço and Nunes, 2017), river bed gravel, mud balls, glass balls (Kumar *et al.*, 2015), ceramsite (Xing *et al.*, 2010) and coal for inert packing (Wang *et al.*, 2010b). Filter packings specific surface area and porosity of filter packing materials have also been reported to impact treatment performance (Toffey, 2008). Besides, specific surface area and porosity of filter packing can affect the treatment performance of VF (Kumar *et al.*, 2015).

In recent years, several studies regarding the removal of organic matter, nutrients and pathogens from domestic and urban wastewater using vermifiltration have been published (Arora and Kazmi, 2015; Tomar and Suthar, 2011). However, few have focused on the impact of different filter packings on vermifiltration performance (Adugna et al., 2019).

The present study focus on the evaluation of the performance of vermifiltration for the treatment of urban wastewater, studying sawdust and vermicompost as filter packing materials, considering a practical case study.

4.2. Material and methods

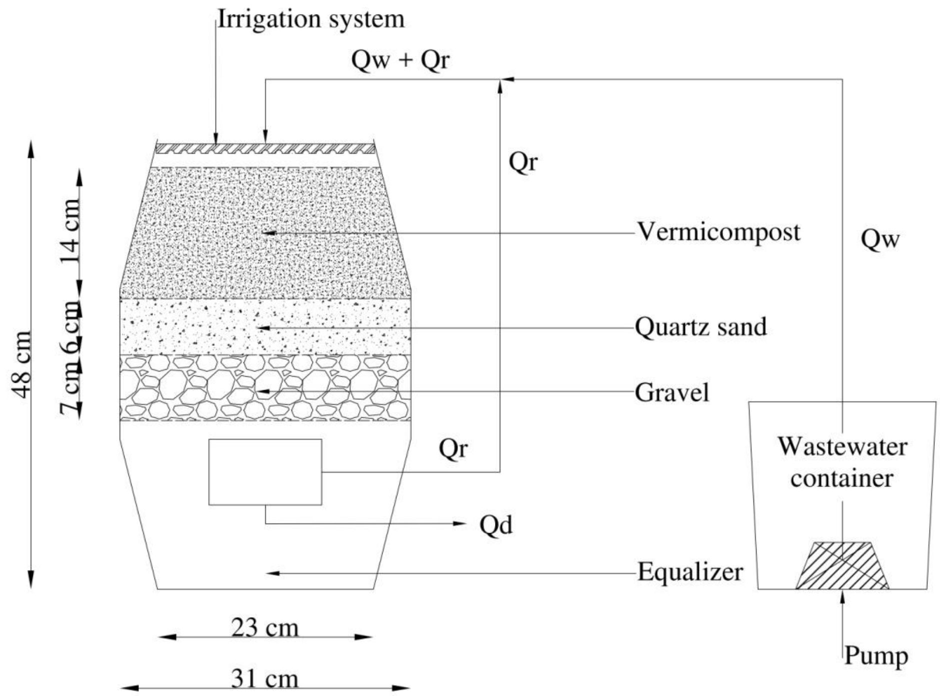
4.2.1. Raw wastewater

The wastewater used in the study came from the urban WWTP of Messines, Algarve, with a served population of 6,000 inhabitants which receives wastewater from a combined sewage collection system designed to transport both rain water and sewage together. All samples were collected on May 13th after the preliminary wastewater treatment. Wastewater used in the study was the same wastewater used in all experiments. No rain was registered during the days before wastewater collection. Wastewater physical-chemical and microbiologic characterization is shown in Table 4.4.

4.2.2. Reactor structure

Reactor modules were constructed in PVC containers with a total volume of 25 L (Figure 4.1) closely following the treatment scheme used in previous studies (Taylor *et al*., 2003). Experiments were made using vermicompost produced from municipal organic solid waste as the filtering material provided by a specialized company (FUTURAMB®). Vermicompost occupied the top 16.0 cm, underneath which was installed an inert filter constituted of 7.0 cm of gravel and 6.0 cm of quartz sand. Percolating water was collected in an equalizer located below the filtering materials. Experiments were made using vermicompost produced from municipal organic solid waste as the packing material, and sawdust produced in a local woodshop which was easily available and could be utilized without any prior treatment. Reactors were covered with a lid, leaving sufficient room and opening as to allow natural aeration. An irrigation system was attached on top of the vermifilter made from 0.5 cm diameter regular network of HDPE flexible plastic pipes. Pipes were perforated with 0.2 cm diameter holes separated by 2.0 cm, for wastewater irrigation, and were kept 3 cm above the filter surface to ensure optimal wastewater distribution, the creation of drop-overflow and thereby increase aerobic conditions. Gravel was separated from the equalizer by a stainless steel mesh (diameter=0.4 cm). Quartz sand was separated from gravel and from vermicompost or sawdust by a stainless steel mesh (diameter=80 μm). Physical-chemical characterization of vermicompost and sawdust are shown on Table

4.2. Parameters were determined by a commercial laboratory. The effluent from each VF was collected in the equalizer from where samples were taken. From here, recirculation was made with the help of a pump (Q_r) and mixed with raw wastewater as



(Q_w) to be feed to the top of the filters (Q_w+Q_r).

Figure 4.1 – Reactor unit design.

Table 4.2 – Characterization of the vermicompost and sawdust filter media (mean concentration ± standard deviation).

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Parameter	Vermicompost	Sawdust
Bulk density (kg m^{-3})	600±0.00	238.66±0.00
Porosity (%)	73.7±0.30	84.0±0.00
Particle size (mm)	<0.1-3.0	<0.1-6.0
*p*H (H_2O)	6.82±0.01	5.38±0.06
EC (μS cm^{-1})	2,530±2.00	99.0±1.20
Organic matter (%)	56.48±0.01	77.4±0.05
TOC (%)	32.76±0.04	45.0±0.05
TN (%)	3.64±0.02	0.50±0.05
C/N ratio	9.0±0.03	90.0±0.00
TP (mg kg^{-1})	3,769±0.4	<0.05
TK (mg kg^{-1})	7,150±0.08	0.11±0.01

4.2.3. Acclimation of the vermifilter

Moisture content was held constant after placing the filters to field capacity following procedures used by the company that provided the earthworms and the filter packing for acclimation of the earthworms. For this purpose filters were flushed with recirculating water for 30 days. After this time, VF was flushed and recirculated permanently for 45 days with wastewater collected from urban WWTP of Messines, Algarve, to allow the growth of heterotrophic microorganisms in the filter packing. Each filter was fed, by pumping raw wastewater from a PVC container. The flow was also adjusted to permit the optimum moisture conditions for the survival of the earthworms.

4.2.4. Experimental design and operational conditions

After the period for acclimation, four treatments were tested for packing material, identified as filter using vermicompost without earthworms (V), filter using vermicompost with the addition of earthworms (VE), filter using sawdust without earthworms (S), and filter using sawdust with the addiction of earthworms (SE). Influent wastewater flow, Q_w and recycling flow, Q_r, were adjusted to obtain a constant Q_{mix} equal to 0.04 m^3 day^{-1}, as this was the optimal flow for maintaining the ideal moisture of the filter. Hydraulic retention time (HRT) was fixed at 6 hours following previous experiments not shown here. Experiments were made for a period of 24 hours with permanent continuous wastewater recirculation. Samples for chemical analysis were taken at the onset of the experiment from the influent wastewater and at the end of the treatment period from the treated effluent. Organic loading rate (OLR) was measured as g BOD_5 m^{-2} day^{-1}. Recirculation ratio was related with Q_r and Q_w and was fixed at 0.72. These parameters were determined using the following equations.

$$HRT = V/Q_w \quad (4.1)$$

$$OLR = Q_w \times BOD_5 / A \quad (4.2)$$

$$HLR = Q_w/A \quad (4.3)$$

$$\text{Recirculation ratio} = Q_r/Q_w \quad (4.4)$$

Where V (m^3) is the volume of the reactor, Q_w (m^3 day^{-1}) the influent wastewater flow rate, BOD_5 (mg L^{-1}) the organic matter concentration in influent wastewater, A (m^2) the reactor's surface area, HLR is the hydraulic loading rate (m^3 m^{-2} day^{-1}), and Q_{mix} (m^3 day^{-1}) is the sum of Q_w and recirculating flow, Q_r (Table 4.3).

Table 4.3 – Hydraulic parameters used in the experiments.

Q_{mix} (m^3 day^{-1})	Q_r (m^3 day^{-1})	Q_w (m^3 day^{-1})	Q_r/Q_{mix}	HRT (h)	HLR (m^3 m^{-2} day^{-1})	OLR (g BOD_5 m^{-2} day^{-1})
0.13	0.09	0.04	0.72	6	0.89	177.63

Eisenia fetida (Bouché, 1972) is one of the most commonly used species for soil pollution and vermifiltration research (Taylor *et al.*, 2003). It has been shown to process organic solid wastes with high efficiency, be very proficuous, being capable to adapt to various environmental factors, including temperature and moisture levels (Edwards and Arancon, 2004). The earthworms were provided by a company specialized in vermicomposting (FUTURAMB®), and previously installed on plastic boxes with coffee grounds at adequate moisture content for 15 days. No signs of disease and stress in the individuals were found. A stocking density of 20 g L^{-1} was used, following previous unpublished studies made at FUTURAMB®. The individuals were placed on the top of the organic filter material and were allowed to install for an acclimation period of 15 days.

During experiments, wastewater was applied continuously for 24 hours. All filters were frequently monitored for foul odors, smooth percolation of wastewater through the vermicompost, and clogging. General earthworm behavior, including agility, movement, stress and health conditions was also monitored. After this period, 200 cm^3 of treated wastewater samples were collect from the equalizer and kept in the cold (4 °C) until analysis.

4.2.5. Sampling and chemical and microbiological analysis

For each treatment, samples were obtained at the beginning and at the end of the treatment. Samples of raw wastewater were taken from the feeding tank and samples of treated wastewater were taken from the equalizers (Figure 4.1). All samples were analyzed immediately after sampling for *p*H, EC, Five-Day Biochemical Oxygen

Demand (BOD_5), Chemical Oxygen Demand (COD), Total Suspended Solids (TSS), NH_4^+, NO_2^-, NO_3^-, Total Nitrogen (TN), PO_4^{3-}, Total Phosphorus (TP), Faecal Coliforms (FC) and helminth eggs. For the analysis, 5 L on each treatment were collected from the equalizer and three replicates were made for each parameter.

*p*H and EC were analyzed using a HANNA HI98129 meter with a precision and range ±0.01 and 0.00-14.00 for *p*H, ±2% and 0.0-3,999 to μS cm^{-1} for EC and the later converted automatically by the equipment do Total Dissolved Solids (TDS) in the range 0-2,000 mg L^{-1}. This later parameter was obtained from EC by a conversion factor of 0.5. BOD_5 was analyzed using an OxiTop®-C respirometric system with incubation at constant temperature for 5 days (APHA, 1998) with a precision and range of ±1%. tCOD was analyzed with a photometer (NOVA 60, Merck) with a precision and range of ±5.0 mg L^{-1} and 25-1,500 mg L^{-1} based on the permanganate method (APHA, 1998). Dissolved fractions were determined after filtrating through a Whatman® 40 μm cellulose filter paper as dissolved COD (sCOD). Particulate COD fraction, pCOD, was obtained as the difference between total, tCOD, and soluble, sCOD. TSS was determined by filtrating the sample through a Whatman® 40 μm cellulose filter paper, drying to a constant weight at 105 °C, and weighting (APHA, 1998). NH_4^+ was quantified by photometry using a HANNA HI733 meter with a precision and range of ±1.0 mg L^{-1} and 0.0-99.9 mg L^{-1} respectively. NO_2^- was analyzed with a HANNA HI708 photometer based on the ferrous sulphate method with a precision and amplitude range of ±3.0 mg L^{-1} and 0-150 mg L^{-1}; and NO_3^- was analyzed with a HANNA HI96786 photometer based on the cadmium reduction method with a precision and range of ±5.0 mg L^{-1} and 0-100 mg L^{-1}. TN analysis was performed through oxidative digestion of all nitrogenous compounds to nitrate based on the persulfate method using (APHA, 1998). PO_4^{3-} analysis was made using a HANNA HI717 photometer based on the heteropolymolybdenum blue method with a precision and range of ±1.0 mg L^{-1} and 0-30 mg L^{-1}. TP was obtained by oxidative digestion of organic matter followed by a colorimetric reaction based on the ascorbic acid method (APHA, 1998). FC were analyzed based on membrane filtration, subsequent culture on a chromogenic coliform agar medium with determination by the most probably number (MPN) per 100 mL^{-1} (ISO 9308, 2014) and *Ascaris lumbricoides* were analyzed as the number of target organisms in the sample (Number 100 mL^{-1}) (APHA, 1998). FC removal efficiency

(K_{FC}) was calculated using equation (4.5) as proposed by Arora *et al*. (2014a), where C_i and C_f are the wastewater FC initial and final FC, respectively.

$$K_{FC} = \mathrm{Log}_{10} (C_i / C_f) \tag{4.5}$$

4.2.6. Statistical analysis

One-way analysis of variance (ANOVA), followed by Tukey test at a significance level of α=0.05 was made to check for differences between treatments. T-test was also performed to compare means. The statistical package SPSS® 17.0 was used in the analysis.

4.3. Results and discussion

Earthworms showed good survival in the filter using vermicompost (VE) and in the filter using sawdust (SE) during the whole experiment, as individuals accommodated to the experimental conditions with no evidence of decrease in population numbers. Individuals meandered throughout all the volume of the organic filter packing, while not trying to escape, meaning that wastewater was not toxic and the environment conditions were suitable. In our study, during the first 15 hours, wastewater percolated smoothly into all reactors, but some clogging was observed in the control filter using vermicompost without earthworms (V) after that period, as indicated by an abnormal accumulation of wastewater on the surface of the filter bed. No clogging was reported in the remaining filters during the whole experiment: vermicompost with earthworms (VE), control filter with sawdust without earthworms (S), and filter with sawdust and earthworms (SE).

The ratio BOD_5/COD is one important way to assess the biodegradability of wastewater, as in a raw urban wastewater the BOD_5/COD ratio varies between 0.3 and 0.8 (Tchobanoglous *et al*., 2003). Besides, with a BOD_5/COD ratio of 0.5, wastewater is considered to be easily treatable by biological processes (Tchobanoglous *et al*., 2003). Also, the common BOD_5/COD ratio in a treated wastewater varies between 0.1 and 0.3 (Henze and Comeau, 2008). In a typical urban wastewater, BOD_5, COD and TSS have as average concentrations of 350, 750 and 400 mg L^{-1} respectively (Henze and Comeau, 2008) (Table 4.4). Comparing our results with the results from literature (Table 4.4), BOD_5 (210±10.0mg L^{-1}), COD (450±10.0 mg L^{-1}) and TSS (158±3.46 mg L^{-1}) were all lower than published ones. This could be justified by solids sedimentation in the PVC container during the study. The BOD_5/COD ratio found for the wastewater was 0.47 indicating good biodegradability (Tchobanoglous *et al*., 2003).

The content of the individual nutrients in wastewater should correspond to the bacteria needs, and there should be a balanced relationship between carbon, nitrogen and phosphorus, as this is crucial to the effectiveness of the biodegradation processes. The concentration of NH_4^+ (46.4 mg L^{-1} ± 0.26) obtained was similar to values referred by Henze and Comeau (2008) of 45 mg L^{-1}, supporting the argument that it could be mainly from domestic sources as urine or cleaning agents. Near 75% of the TN in a typical urban wastewater is NH_4^+ and the majority (70–90%) comes from urine, whilst the final 20% comes from cleaning agents, disinfectants and food wastes (Hughes *et al*., 2008). *Faecal* coliforms concentration ($5.7x10^8$±$3.98x10^1$ MPN 100 mL^{-1}, Table 4.4) was relatively high if compared with literature (George *et al*., 2002).

Table 4.4 – Characterization of influent wastewater and typical values from literature data.

Parameter	Value[1]	Typical values		
		Henze and Comeau (2008)	USEPA (2004b)	Tchobanoglous *et al*. (2003)[*]
*p*H	8.48±0.03	n.a.	n.a.	n.a.
BOD_5 (mg L^{-1})	210±10.0	350	221	300
COD (mg L^{-1})	450±10.0	750	580	650
TDS (mg L^{-1})	532±5.00	n.a.	n.a.	n.a.
Turbidity (NTU)	148.3±7.51	n.a.	n.a.	n.a.
TSS (mg L^{-1})	158±3.46	400	243	500
NH_4^+-N (mg L^{-1})	49.4±0.31	45	9	n.a.
NO_2^--N (mg L^{-1})	2.0±0.75	0.2	Σ<1	n.a.
NO_3^--N (mg L^{-1})[**]	0.2±0.12			n.a.
TN (mg L^{-1})	68.3±0.31	60	51	70
PO_4^{3-}-P (mg L^{-1})	16.3±0.75	10	n.a.	n.a.
TP (mg L^{-1})	5.7±0.12	15	9	15
BOD_5/tCOD	0.47±0.03	0.47	0.38	0.44
COD/NH_4^+-N	9.1±0.23	16.7	64.4	17.2
FC (MPN 100 mL^{-1})	$5.7x10^8$±$3.98x10^1$	$1.0x10^{12}$[**]	$1.0x10^7$	$2.2x10^6$
Helminth eggs (N.º L^{-1})[***]	8.00±6.24	13	n.a.	n.a.

[1] Mean concentration ± standard deviation.
n.a.: Not available.
[*] Average concentration.
[**] Converted by mass equation.
[***] As total coliforms.
[****] As *Ascaris lumbricoides* eggs.

Treatments showed a good efficiency for BOD_5, COD and TSS from wastewater (Tukey test, $p<0.05$, Table 4.5). BOD_5, COD and TSS values in all treatments met the EU standards (Directive 91/271/EEC, 21th May, 1991) for wastewater discharge, namely of 25 mg L^{-1}, or a minimum removal of 70-90% for BOD_5, 125 mg L^{-1} or a minimum removal of 75% for COD and 35 mg L^{-1} or a minimum removal of 90% for TSS. Removal efficiencies for BOD_5 were 91.27%±0.55 in VE, 96.19%±0.00 in V, 90.48% in SE and 92.06% in S as removal efficiencies for COD were 87.56%±0.45 in VE, 86.67%±0.89 in V, 79.70%±0.92 in SE and 77.63%±1.80 in S. As for TSS, removal efficiencies were 98.42±0.00 %±0.55 in VE, V, SE and S (Table 5). Dissolution of earthworm castings may have contributed to increase BOD_5 values in VE and SE (18.33±1.15 mg L^{-1} and 8.00±0.00 mg L^{-1}) compared to V and S (20.0±1.00 mg L^{-1} and 16.67±1.15 mg L^{-1}). Vermifiltration contributed to higher COD efficiencies in treated wastewater (56.0±2.0 mg L^{-1} in VE and 91.33±4.2 mg L^{-1} in SE (Table 4.5). COD removal efficiency was lower compared to BOD_5 (91.27% in VE and 90.48% in SE for BOD_5 and 87.56% in VE and 79.70% in SE for COD, Table 4.5), due to the fact that earthworms are mainly responsible for the removal of biodegradable substances. In comparison, Sinha *et al*. (2008) reported removal of TSS in the ranges of 90–92% and 90–95%, for COD and BOD_5, respectively. Xing *et al*. (2010) reported that the presence of earthworms was responsible for about 57 to 79% reduction in TSS in wastewater, which was lower than the values obtained here. The vermifilter system with sawdust was less efficient to reduce turbidity from wastewater (2.28 NTU±0.08 in SE and 1.17 NTU±0.14 in S), and earthworms in fact contributed to increase turbidity (3.94 NTU±0.16 in VE and 2.28 NTU±0.08 in SE) comparing to the systems without earthworms (V and S) (Table 4.6).

Table 4.5 – Parameters and efficiencies for the different treatments.

Parameter	**Raw wastewater**	**VE**	***η* (%)**	**V**	***η* (%)**
BOD_5 (mg L^{-1})	210±10.0	18.33^{bc}±1.15	91.27^{ab}±0.55	8.00^{a}±0.00	96.19^{c}±0.00
COD (mg L^{-1})	450±10.0	56.0^{a}±2.00	87.56^{b}±0.45	60.0^{a}±4.00	86.67^{b}±0.89
TSS (mg L^{-1})	532±5.00	2.5^{a}±0.00	98.42^{a}±0.00	2.5^{a}±0.00	98.42^{a}±0.00
NH_4^+ (mg L^{-1})	49.4±0.31	11.60^{c}±0.15	76.51^{b}±0.24	8.57^{b}±0.32	82.64^{c}±0.50
TN (mg L^{-1})	68.3±0.31	60.0^{d}±3.00	12.20^{a}±4.40	22.0^{c}±0.00	67.80^{b}±0.00
TP (mg L^{-1})	5.7±0.12	11.7^{b}±0.58	-105.88^{a}±10.18	11.7^{b}±0.58	-105.88^{a}±10.18
Parameter	**Raw wastewater**	**SE**	***η* (%)**	**S**	***η* (%)**
BOD_5 (mg L^{-1})	210±10.0	20.0^{c}±1.00	90.48^{a}±0.48	16.67^{b}±1.15	92.06^{b}±0.55
COD (mg L^{-1})	450±10.0	91.33^{b}±4.20	79.70^{a}±0.92	100.67^{b}±8.10	77.63^{a}±1.80
TSS (mg L^{-1})	532±5.00	2.5^{a}±0.00	98.42^{a}±0.00	$2,5^{a}$±0.00	98.42^{a}±0.00
NH_4^+ (mg L^{-1})	49.4±0.31	18.08^{d}±0.76	63.40^{a}±1.19	2.54^{a}±0.31	94.86^{d}±0.48
TN (mg L^{-1})	68.3±0.31	9.3^{b}±0.58	86.34^{c}±0.84	2.0^{a}±0.00	97.07^{d}±0.00
TP (mg L^{-1})	5.7±0.12	0.6^{a}±0.00	89.41^{b}±0.00	0.065^{a}±0.00	98.85^{b}±0.00

[1] Mean concentration ± standard deviation. Values followed by the same letter within each line are not significantly different (ANOVA; Tukey's test, α = 0.05).

Table 4.6 – BOD_5/COD and COD/NH_4^+-N, nutrient concentration[1], and *p*H, TDS and turbidity[1] for the different treatments.

Parameter	Raw wastewater	Experiment			
		VE	**V**	**SE**	**S**
BOD_5/COD	0.47±0.03	0.33^c±0.03	0.13^a±0.01	0.22^b±0.02	0.17^{ab}±0.02
COD/NH_4^+-N	9.1±0.23	4.8^a±0.22	7.0^b±0.55	5.1^a±0.40	39.7^c±1.21
NO_2^- (mg L^{-1})	2.0±0.75	3.2^b±0.68	3.5^b±0.92	0.6^a±0.60	0.7^a±0.17
NO_3^- (mg L^{-1})	0.2±0.12	4.9^c±0.29	1.5^b±0.21	0.0^a±0.00	0.0^a±0.00
PO_4^{3-} (mg L^{-1})	16.3±0.75	10.7^c±0.12	11.1^c±0.25	1.3^b±0.01	0.03^a±0.06
*p*H	8.48±0.03	7.85^a±0.04	8.37^c±0.02	8.22^b±0.00	8.46^d±0.01
TDS	532±5.00	476^c±3.51	418^b±2.89	423^b±1.00	363^a±1.00
Turbidity (NTU)	148.3±7.51	3.94^c±0.16	4.73^d±0.11	2.28^b±0.08	1.17^a±0.14

[1] Mean concentration ± standard deviation. Values followed by the same letter within each line are not significantly different (ANOVA; Tukey's test, $\alpha = 0.05$).

Earthworms significantly degrade the wastewater organics by enzymatic action in their gut, improving the degradation of several compounds which could not be decomposed by microorganisms (Sinha et al., 2010; Malek *et al.*, 2012). This can explain the higher COD efficiencies obtained in VE and SE, where microbial stimulation, biodegradation and enzymatic degradation of solid wastes by earthworms work simultaneously (Sinha *et al.*, 2010). In fact, vermifiltration is effective due to the biological, physical and chemical reactions occurred, including the adsorption of molecules and ions, oxidation–reduction of organic matter, and the synergetic effects of earthworms with microorganisms (Bouché and Soto, 2004).

The higher removal BOD_5 and COD efficiencies in VE compared to SE may be related with the higher C/N content in sawdust compared to that of vermicompost (45.0±0.05% to 32.76±0.04 %, Table 4.2) since more carbon contend (as carbonaceous BOD_5) from sawdust may have been released to the wastewater. Specific surface area and porosity of filter packing are one of the factors that affect the treatment performance in biologic filtration (Toffey, 2008). A filter packing with low granulometry improve biomass accumulation and attains higher treatment efficiency as compared to the performance of media with low specific surface area (Taylor *et al.*, 2003). Since vermicompost has lower granulometry compared to sawdust (Table 4.2), its higher specific surface may have created better conditions for microorganisms to survive and grow. This could explain the significantly higher BOD_5 removal efficiencies (Tukey test, $p<0.05$) obtained in VE (91.27%±0.55) and V (96.19%±0.00). Since organic solid particles are retained in the pores of the filter packing, high removal efficiencies for TSS are usually expected (Sinha *et al.*, 2008). In our experiments, there was no significant difference in TSS removal efficiency in systems with or without the presence of earthworms (Tukey test, $p>0.05$), which can indicate that the removal process is essentially physical.

In our study, when compared with raw wastewater (46.4±0.26 mg L^{-1}), NH_4^+ concentrations decreased in all experiments (11.60±0.15 mg L^{-1} in VE, 8.57±0.32 mg L^{-1} in V, 18.08±0.76 mg L^{-1} in SE and 2.54±0.31 mg L^{-1} in S, Table 5). Vermifiltration process contributed to decrease NH_4^+ removal efficiency (Tukey test, $p<0.05$) (76.51%±0.24 at VE and 63.40%±1.19 at SE) as V had an efficiency of 82.64%±0.50 and S had an efficiency of 94.86%±0.48 (Table 4.5). NH_4^+ is generated by organic nitrogen mineralization leading to ammonia emissions being the first inorganic nitrogen form produced during biological wastewater treatment (Henze and Comeau, 2008). Vermicasts increase nutrient content in soil (Edwards *et al*., 2011) since N cycling is directly influenced by earthworms. Kadam *et al*. (2009) concluded that NH_4^+, which is the dominant form of N present in domestic wastewater, was removed through rapid adsorption by the filter packing and subsequently converted into NO_3^- through nitrification. The increase in NH_4^+ on VE and SE compared with V and S may be due to the ion leachate from vermicastings during treatment. Besides, vermicompost packing may have contributed to increase NH_4^+ due to the fact the vermicompost is mainly constituted by earthworm castings and is rich in heterotrophic bacteria which increase organic nitrogen mineralization (Sinha *et al*, 2008). Also, the excess of ammonium in wastewater may contribute to earthworm' stress (Hughes *et al*., 2008). The former authors reported ammonium concentrations of 25 mg L^{-1} in treated effluent after vermifiltration and a LC50 of 1.49 mg L^{-1} and a 0% survival rate above 2.0 mg L^{-1}. In our study, the low toxicity of ammonium may be attributed to the rapid conversion of ammonium to nitrate.

All treatments contributed to decrease BOD_5/COD ratios (Tukey test, $p<0.05$) but in the presence of earthworms the BOD_5/COD ratios were higher (0.33±0.03 and 0.22±0.02 in VE and SE, and 0.13±0.01 and 0.17±0.02 in V and S, respectively (Table 6). The reason may be due to the release of dissolved organic compounds from the vermicastings. Comparing the two filter materials, no significant difference was found in BOD_5/COD ratio (Tukey test, $p>0.05$). Degradation of organic fractions of wastewater produces several acidic species of mineralized organic materials (CO_2, NH_4^+, NO_3^- and organic acids) which play an important role in shifting of *p*H of treated wastewater. This may justify the decrease in *p*H in all treatments. Besides, vermifiltration contributed to decrease *p*H from raw wastewater (7.85±0.04 at VE and 8.22 ± 0.00 at SE, Table 4.6). Edwards *et al*. (2011) and Arora *et al*. (2014b) reported the influence of earthworms in making *p*H converge to neutrality in soil, solid organic wastes treatment and vermifiltration. Hughes *et al*. (2008) has also found that vermicompost as filter packing has high buffering capacity for *p*H.

Carbon to nitrogen ratio in raw wastewater plays an important role in wastewater treatment and is measured by the COD/NH_4^+-N ratio change (Cardoso-Vigueros *et al*., 2013). Vermifiltration had a significant influence in COD/NH_4^+-N ratio (Tukey test, $p<0.05$). For TN removal rates by nitrification may be improved when carbon to nitrogen ratios in wastewater is in between 5:1 and 10:1 (Roy *et al*., 2010). The filter with vermicompost and earthworms showed the highest nitrification (Tukey test, $p<0.05$), as the lowest COD/NH_4^+-N, 4.8±0.22 mg L^{-1}, was obtained in VE (Table 4.6). Nitrification coupled with denitrification is pointed to be the major N removal process involved in many vermifiltration systems, while insufficient available organic C (as COD) is considered to be the reason for the inhibition of denitrification (Sinha *et al*., 2008). NO_2^- is an intermediate product of nitrification and its concentration in wastewater is usually negligible (Henze and Comeau, 2008). Comparing vermicompost and sawdust, the first contributed do increase NO_2^- (3.2±0.68 mg L^{-1} in VE and 3.5±0.92 mg L^{-1} in V) and also to increase NO_2^- concentration relatively to raw wastewater (0.2±0.12 mg L^{-1}). No statistically significant difference was obtained between treatments and NO_2^- concentration (Tukey test, $p>0.05$, Table 4.6). In nitrification, the adsorbed NH_4^+ is subsequently converted to NO_3^-, carried out by autotrophic bacteria which use molecular oxygen as an electron acceptor (Zhang *et al*. 2005). Nitrification step for NH_4^+ removal led to a substantial increase in NO_3^- concentration in VE and V as no NO_3^- was found in SE and S (Table 4.6). NO_3^- concentration increased in the treatment using vermicompost with VE registering 4.9±0.29 mg L^{-1} and V registering 1.5±0.21 mg L^{-1} (Table 4.6), comparing to raw wastewater (0.2±0.12 mg L^{-1}) (Table 4). The presence of earthworms contributed to increase NO_3^- concentration from 0.2±0.12 mg L^{-1} in raw wastewater to 4.9±0.29 mg L^{-1} in VE and 1.5±0.21 in SE ($p<0.05$, Table 4.6). Vermicompost is rich in nitrifying bacteria which help effluent mineralization (Sinha *et al*., 2008). This is also supported by Cardoso-Vigueros *et al*. (2013) who found a positive correlation between earthworm density and nitrifying bacteria, helped by abundant oxygen due to the burrowing action of earthworms. Also, earthworms excrete polysaccharides, proteins and other nitrogenous compounds as they mineralize nitrogen in wastewater (Sinha, 2010). The highest rates of mineralization occur in the vermicasts, which greatly enhances the availability of inorganic nutrients.

The presence of earthworms contributed to decrease TN removal efficiency (Tukey test, $p<0.05$) (12.40%±4.40 at VE and 86.34%±0.84 at SE) as V had an efficiency of 67.80%±0.00 and S had an efficiency of 97.07%±0.00 (Table 4.5). Besides, when comparing both filter packings, vermicompost (VE and V) contributed to reduce TN removal efficiency (Tukey test, $p<0.05$, Table 4.5).

In the current study PO_4^{3-} concentrations decreased in all treatments relatively to raw wastewater (Tukey test, $p<0.05$), while no statistically significant difference was obtained for PO_4^{3-} between VE and V (Tukey test, $p>0.05$, Table 4.6). The presence of earthworms did not improve TP removal since no statistically significant difference was obtained between treatments and TP (Tukey test, $p>0.05$, Table 4.5). Vermicompost contributed to increase TP concentration compared to raw wastewater (11.7±0.58 mg L^{-1} in VE and 11.7±0.58 mg L^{-1} in V) (Table 4.5). In contrary, sawdust contributed to reduce TP from raw wastewater with 0.6±0.00 mg L^{-1} in SE and 0.065±0.00 mg L^{-1} in S (Table 4.5). Due to this fact, TP removal efficiencies in treatments using vermicompost were negative (-105.88%±10.18 at VE and -105.88%±10.18 at V) (Table 4.5). TP removal in SE and S suggest that sawdust may have contributed to remove organic and PO_4^{3-} from wastewater due to absorption of inorganic constituents by different biological or non-biological components. Moreover, in the filters with sawdust it was possible to observe a statistically significant difference between SE and S (Tukey test, $p<0.05$, Table 4.6) with SE ending with higher PO_4^{3-} concentration (1.3±0.01 mg L^{-1}) than S (0.03±0.06 mg L^{-1}). PO_4^{3-} increase during vermifiltration is related with the enzymatic and microbial activity due to the presence of earthworms (Hait and Tare, 2011). An increase in TP concentration during vermifiltration has been reported by other authors (Cardoso-Vigueros *et al.*, 2013; Arora *et al.*, 2016; Kumar *et al.*, 2015). The vermicastings can increase the concentration of nutrients in vermifilter effluents more significantly, as indicated above, which can explain the negative removal efficiencies obtained for TP in our study.

Comparing to current EU standards, VE and V exceeded total nitrogen and total phosphorus emission limits of 15.0 mg L^{-1} and 2.0 mg L^{-1}, respectively. According with this regulation, these two parameters are especially important in sensitive water bodies and fundamental nutrients responsible for eutrophication processes. Nutrient increase is also supported by the fact that earthworms contributed to increase ion concentration in treated effluent since TDS was 476±5.31 mg L^{-1} in VF and 423±5.31 mg L^{-1} in SE, compared with 418±2.89 mg L^{-1} in V and 363±1.00 mg L^{-1} in S. This is also supported by the fact that, as expected, *p*H followed the mineralization process and oxidation of organic compounds (Table 4.6).

Removal of pathogens is one of the main objectives when treating wastewater for discharge in water bodies or reuse for irrigation. Faecal coliforms typical concentration in raw wastewater is usually between 10^6 and 10^8 MPN 100 mL^{-1} depending both of raw wastewater composition and treatment efficiency (George *et al*., 2002). *Ascaris lumbricoides* eggs are a good indicator of parasitological quality since 99.9% of removal must be achieved (WHO, 2006). All faecal coliforms (concentration, Log_{10} FC, K_{FC} and k_{FC}) and helminth eggs parameters during the study are given in Table 4.7. Log_{10} FC values were all between the values reported by WHO (2006) with 4.70±0.01 in VE, 4.78±0.03 in V, 4.72±0.02 in SE and 3.26±0.24 in S. No statistically significant difference was obtained for Log_{10} FC value between VE and V treatments (Tukey test, $p>0.05$). In filter with sawdust, vermifiltration did not contribute to decrease Log_{10} FC value (Tukey test, $p>0.05$). No statistically significant difference was obtained for K_{FC} and k_{FC} values between treatments (Tukey test, $p>0.05$). *Ascaris lumbricoides* eggs were all removed of 100% in all experiments (Table 4.7). Based on faecal coliforms concentration in raw wastewater and the maximum concentration permitted by WHO (2006) of 6-7 Log_{10} units for unrestricted irrigation, it is possible to predict that the minimal K_{FC} and k_{FC} values in the final effluent obtained from vermifiltration should be, respectively, 5.91 and 11.70.

Arora *et al*. (2014a) studied the removal of *E. coli* from urban wastewater using vermifiltration having obtained a reduction from a mean Log_{10} value of 4.48 MPN 100 mL^{-1} to 2.80 MPN 100 mL^{-1}. Using vermicompost as filter packing Arora *et al*. (2016) obtained an effluent wastewater with a mean Log_{10} value of 2.50 MPN 100 mL^{-1} starting from an influent wastewater of 5.48 MPN 100 mL^{-1}. In their studies, Kumar *et al.* (2015) using as filter packing, vermicompost and river bed material, vermicompost and wood coal, vermicompost and glass balls, and vermicompost and mud balls, reported a reduction of faecal coliforms of 3.4±0.67, 2.9±0.88, 2.6±0.45 and 2.6±1.05 Log_{10} MPN 100 mL^{-1}, respectively.

Guidelines for wastewater reuse in irrigation indicate a *p*H between 6.0 and 9.0, a BOD_5 concentration ≤10 mg L^{-1} (for food crops consumed uncooked) or ≤30 mg L^{-1} (for non-food crops and food crops consumed after processing), a TSS concentration between ≤30 mg L^{-1} (for processed food crops) and, for faecal coliforms and helminth eggs, a maximal MPN of 10^3 100 mL^{-1} (or 3.0 Log_{10}) and 1 unit L^{-1}, for agricultural irrigation (USEPA, 2004). For pathogens only, WHO (2006) indicates a maximum MPN of 10^3 100 mL^{-1} (for unrestricted use), a maximum MPN of 10^4 100 mL^{-1} for restricted use and ≤1 No. L^{-1} for helminth eggs. In all experiments, *p*H and TSS attain with these limits. For BOD_5, all treatments attained the limit concentration for non-food crops and crops consumed after processing (≤30 mg L^{-1}), but only vermicompost without earthworms attained the limit concentration for food crops consumed uncooked (≤10 mg L^{-1}) (Table 4.6). Besides, none of the experiments reduced faecal coliforms to less than 10^3 MPN 100 mL^{-1} or a Log_{10} value less than 3.0 (Table 4.7). All treatments removed helminth eggs with an efficiency of 100%. This could be explained due to the destruction of the three layers of protective shells that constituted helminth eggs. Nevertheless, all values related with faecal coliforms and helminth eggs were in accordance with the proposed by WHO (2006) for primary and secondary wastewater treatment technologies.

Table 4.7 – Faecal coliforms in treated wastewater.

Parameter	**VE**	**V**	**SE**	**S**
FC	$5.07x10^{4b}$±$1.53x10^3$	$6.03x10^{4c}$±$4.04x10^3$	$5.3x10^{4b}$±$3.00x10^3$	$2.0x10^{3a}$±$1.00x10^3$
Log_{10} FC value	4.70^b±0.01	4.78^b±0.03	4.72^b±0.02	3.26^a±0.24
K_{FC}	3.693^a±0.01	3.618^a±0.03	3.674^a±0.03	5.139^b±0.24
k_{FC}	7.484^a±0.03	7.320^a±0.06	7.445^a±0.05	10.412^b±0.49
Helminth eggs removal efficiency η (%)	100.00^a±0.00	100.00^a±0.00	100.00^a±0.00	100.00^a±0.00

[1] Mean concentration ± standard deviation. Values followed by the same letter within each line are not significantly different (ANOVA; Tukey's test, α = 0.05).

The results using single stage vermifiltration were not completely positive since the efficiencies obtained for some of the parameters were still short to attain the EU regulation for discharges in sensitive water bodies (in case TN and TP) and USEPA and WHO guidelines for irrigation (in case faecal coliforms). The efficiencies were nonetheless higher than those found in similar conditions as, e.g., in Arora *et al*. (2016), with 85.5% for BOD_5, 77.8% for COD and 82.2% for TSS. As for NH_4^+, high efficiencies with VF may be only attainable with vertical stage VF since Yang *et al*. (2015) reported an increase in NH_4^+ removal with the depth of filter packing. Faecal coliform removal efficiencies here obtained do not meet guidelines as a maximal MPN of 10^3 100 mL^{-1} for irrigation through VF has shown to not attain such level of use.

Table 4.8 resumes the parameters removal efficiencies obtained in literature. Comparing this data with the values obtained in this study, the best removal efficiency for BOD_5 (96.19% working with filter with vermicompost without earthworms) was just lower than efficiencies obtained by Sinha *et al*. (2008) – 98%, and Cardoso-Vigueros *et al*. (2013) (99%). For COD, the best removal efficiency (87.56% working with filter with vermicompost and earthworms) was just lower than the obtained by Cardoso-Vigueros *et al*. (2013) – 92%, and for TSS, in all treatments, removal efficiency (98.42%) was higher when compared with current literature. When analyzing the NH_4^+ removal, the value obtained in filter with vermicompost and earthworms – 76.51%, was lower than the efficiencies obtained by Cardoso-Vigueros *et al*. (2013) – 98%, and Arora *et al*. (2016) – 90%. In what concerns TP, Cardoso-Vigueros *et al*., (2013), Arora *et al*. (2016) and Kumar *et al*. (2015), all registed increases in TP concentration in treated effluent, which is in line with our results. The latter two authors obtained in their studies a treated effluent with less than a Log_{10} of 3.0 of faecal coliforms, which clearly surpasses the results obtained in our study.

Table 4.8 – Treatment efficiencies obtained in literature using VF.

Parameter	**Sinha *et al*. (2008)**	**Cardoso-Vigueros *et al*. (2013)**	**Arora *et al*. (2016)**	**Arora *et al*. (2014a)**	**Kumar *et al*. (2015)**
BOD_5	98%	99%	86%	76%	81%
COD	45%	92%	78%	67%	72%
TSS	90%	97%	82%	-	75%
NH_4^+	-	98%	90%	-	76%
TN	-	78%	-	-	-
TP	-	[a]	[a]	-	[a]
Faecal coliforms*	-	-	2.82	2.70	3.40

[a] Authors reported an increase in TP final concentration.
*As Log_{10} FC value.

4.4. Conclusions

Vermicompost and sawdust showed high treatment efficiency for BOD_5, COD and TSS. Even so, the values using single stage vermifiltration were not completely positive given that the efficiencies obtained for TP and TN were still above EU guideline values for discharge in sensitive water bodies, and faecal coliforms values were above the WHO guideline values for irrigation.

Earthworms contributed to reduce treatment efficiencies for BOD_5, NH_4^+ and TN, and to increase treatment efficiency for COD. In vermicompost, earthworms contributed to increase NO_3^- concentration. Comparing with raw wastewater, vermicompost contributed to increase TP. No treatment eliminated faecal coliforms down to guidelines values for wastewater irrigation but helminth eggs were completely eliminated in all experiments.

In order to attain EU guideline values for discharge in sensitive water bodies and WHO guideline values for irrigation, alternative treatment technologies are needed, namely, sequential vermifiltration systems or vermifilters followed by wetlands, working as hybrid systems suited for small communities.

Vermifilters must be capable of operating with variable input load rates and climatic conditions as these variables cannot be controlled at individual homesites. A research program to quantify the performance and sustainability of the system at various input load rates and temperatures should be, therefore, developed.

Chapter 5

Life cycle assessment to compare nature-based solutions for wastewater treatment in small communities

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5.1. Introduction

Urban wastewater treatment for small rural communities, where technical and financial resources are usually limited, can pose a problem, as solutions adopted in larger urban areas are not applicable. A number of small-scale treatment systems have been developed, which are adapted to the needs of these areas (Martin et al., 2006). These WWTPs have also good applicability in developing countries as they require less investment and have less technically demanding maintenance operations common in large centralized facilities (Muga and Mihelcic, 2008). Given the need for alternative solutions, priority has been given to technologies which have minimum energy requirements, simple operational procedures, and sufficient level of inertia when faced large fluctuations in the flow and water quality (Salas, 2004). Several types of decentralized wastewater treatment technologies for rural areas include: i) active sludge system (AS) (Ahansazan *et al*., 2014), anaerobic sludge reactor (e.g. septic tank) (Tilley et al., 2014), stabilization ponds (Mara, 2006a), constructed wetlands (Vymazal, 2011), and infiltration systems (Li et al., 2015).

Vermifiltration (VF) is an example of the later, recognized as a low-cost and sustainable technology to treat wastewater (Jiang *et al*., 2016, Kadam *et al*., 2009), sewage sludge (Zhao *et al*., 2010) and fecal matter (Furlong *et al*., 2014, 2015, 2016). On top of the above-mentioned advantages over conventional wastewater treatment solutions, VF has been shown to have higher efficiencies when flows are small (Li *et al*., 2008). It is a bio-oxidative process combining filtration with vermicomposting processes (Pathania *et al*., 2013) in vermifilters (VFs) (Lourenço and Nunes, 2017a). Filter packing in VFs is constituted by an organic packing and an inert packing (Lourenço and Nunes, 2017a). It usually includes sawdust and vermicompost (Lourenço and Nunes, 2017b), peat and wood flour (Li *et al*., 2009), wood chips, (Li *et al*., 2008) for organic packing; and gravel, quartz sand (Lourenço and Nunes, 2017b), river bed gravel, mud balls, glass balls (Kumar *et al*., 2014), ceramsite and coal, for inert packing (Wang *et al*., 2010b). Many of these packing materials are wastes from other activities, which are recycled into secondary raw materials, in line with circular economy principles.

VF has found application in households, small communities, and to treat mixtures of urban and industrial wastewater (Sinha *et al*., 2014). It is also accepted that VF can be more cost-effective when compared with conventional WWTP (Liu *et al*., 2012) since it does not require mechanical equipment other than pumps (Sinha *et al*., 2008). Several VF are already working in France (1,000 PE) (RECYCLAQUA, 2019), Spain, Chile (Fundación Chile, 2019), Brazil (Madrid, 2016), and P. R. China (Nie et al., 2013).

The present paper makes a comparative LCIA study for three small WWTPs, where vermifiltration is compared against conventional treatment technologies. The study uses the international standards for LCA (ISO 14040, 2006). The life cycle impact assessment (LCIA) provides a mechanism for systematically evaluating the inputs and outputs linked to a product or process and can aid in guiding process or product improvement efforts. It also provides information about product choices, maintenance and end-of-product-life strategies, having gained global acceptance as regulatory methodology (e.g., by the European Commission, the United States Environment Agency, and the United Nations Environment Programme). A LCIA quantifies the environmental impacts for a given product based upon the established, study boundaries focusing on the entire life cycle of a product, from raw material acquisition to final disposition. It has been used to explore the sustainability of WWTPs (Ortiz et al., 2007) and, using life cycle inventory analysis (LCIA), to compare between alternative unit treatment processes (Wu *et al*., 2010), and alternative integrated wastewater management solutions (Emerson *et al*., 2005; Palme *et al*., 2005).

The key elements of the life cycle inventory (LCI) methodology include the study boundaries, resources (raw materials and energy), emissions (atmospheric, waterborne, and solid waste), and disposal practices. It consists of detailed tracking of all the flows in and out of the product system, including raw resources or materials, energy, water, and greenhouse gases (GHG), and substances. Characterization in LCA is the conversion of LCI results to common units within each impact category, so that results can be aggregated into category indicator results. In fact, once the different LCI results are assigned to the different impact categories, one should define the characterization factors. These factors define the relative contribution of the different LCI results to the impact category (Van den Bossche et al. 2006).

Most LCIA studies focus on conventional WWTPs, analyzing the system's end-of-life by-products, waste, and wastewater discharge (Rego, 2012; Renou, 2006; Li et al., 2009; Foley et al., 2010). Little attention has been given so far to nature-based decentralized WWTPs, which commonly serve population sizes below 2,000 person-equivalent (PE) or process influent flow rates below 200 m^3 day^{-1} (Lens et al., 2001).

5.2. Case-studies

The present study focuses on systems used for wastewater treatment in rural areas with communities up to 120 PE. Three real case-studies were chosen: i) a slow rate filtration plant (SRI); ii) a constructed wetland (CW); and iii) an self-supporting activated sludge prefabricated steel unit (AS). The data from each WWTP necessary for the inventory and analysis of the current WWTPs were retrieved from published studies (Machado et al., 2007; Nogueira et al., 2009). To ease interpretation, the case-studies were named according to their original treatment systems. The systems were evaluated first by considering their present process diagram (base solution), and after an hypothetical replacement of the secondary treatment by vermifiltration (VF) (alternative solution).

SRI includes a pre-treatment followed by the slow infiltration system with *Populus euroamericana* and *Eucaliptus colmadulensis* as the used biomass, which occupies an area of 2,000 m^2. It has a design capacity of 40 PE in winter and 120 in summer, with a flow rate of 5.0 m^3 day^{-1} in winter and 15.0 m^3 day^{-1} in summer. The biomass produced is harvested each 5 years, shredded, and sold to the paper pulp industry. The irrigation system is formed by a polyethylene piping network. It is an experimental infrastructure located at Carrión de los Céspedes (Spain). CW is an experimental infrastructure located also in Carrión de los Céspedes (Spain). It includes an Imhoff tank which is used as primary treatment and two vertical-flow wetlands in series with 317.0 m^2 each, followed by a horizontal-flow wetland with 277.0 m^2. Grown biomass is cut yearly and transported to a landfill. It has a design capacity of 120 PE and processes a flow rate of 15.0 m^3 day^{-1}. AS is a full-scale municipal infrastructure located in Vila Verde (Braga, Portugal). It has a design capacity of 500 PE and treats a flow rate of 60.0 m^3 day^{-1}. It is constituted by an activated sludge tank with two surface aerators, working each 11 h day^{-1}, a primary clarifier and a secondary clarifier. It is assumed that all sludge produced in CW and AS is deposited in landfill for biogas production.

VFs are packed media filters using earthworms where wastewater is loaded intermittently into the upper surface and allowed to percolate through the system

(USEPA, 2002). Sizing is made according to the influent flow rate, hydraulic retention time (HRT), hydraulic loading rate (HLR), organic loading rate (OLR), packing depth, and abundance of earthworms (Lourenço and Nunes, 2017a; Singh *et al*., 2017; Samal *et al*., 2017).

The data necessary to compute the materials and resources necessary to build the VF was retrieved from literature, based on sizing optimization parameters (Lourenço and Nunes, 2017a). The VF is made of pre-cast concrete and is constituted by an organic packing (sawdust) (0.3 m depth) and an inert packing (sand with 0.3 m depth and aggregate with 0.3 depth). The bottom of the VF includes a clarifier for wastewater recirculation (Figure 3.1).

5.3. Method

5.3.1. Life cycle inventory

A full comparative life cycle inventory was carried out for the three WWTPs for the base solution and the alternative. A material balance was made for each individual process. In the analysis all the inputs required, and all the outputs generated were identified and quantified. The computations were made in OpenLCA® Nexus (version 1.7) (openLCA.org, 2019), using Ecoinvent database, v. 3.5 (ECOINVENT, 2019). CML Baseline v.4.4 (January 2015) was used to perform the quantification of the impacts, and EU25+3 (2000) for the normalization.

Though VFs are suited for primary or secondary treatment (Sinha *et al*., 2008; Li *et al*., 2009; Hill and Baldwin, 2012; Lourenço 2017a), in order to keep the original primary treatments of the case-studies, VF was sized for secondary treatment only. SRI and AS included a septic tank, and CW an Imhoff tank, as primary treatments, which were maintained when studying the replacement of the original secondary treatment by VF. So, the LCIA is used to assess the impacts of using vermifiltration as secondary treatment, replacing wetlands and the activated sludge tank (Figure 5.1).

The sizing of the VF followed optimized operational parameters obtained before (Lourenço and Nunes, 2017b), namely: (hydraulic retention time (HRT) (6h), hydraulic loading rate (HLR) (0.89 m^3 m^{-2} day^{-1}), organic loading rate (OLR) (0.8 kg BOD m^{-3} day^{-1}), abundance of earthworms (20 g L^{-1}), recirculation flow rate (Q_r/Q_{mix}) (0.7) and the packing materials, which includes sawdust (0.3 m), quartz sand (0.3 m) and gravel (0.3 m), and wet to dry time ratio (W:D) (1:3) (Wang et al., 2014). The growth of earthworm's biomass was 0.12 g L^{-1} packing day^{-1} (Zhao et al., 2010). The size of the population of earthworms needed to be maintained constant throughout the exploration period, leading to the need to subtract some individuals. These are assumed to be released back in nature. Since both SRI-VF and CW-VF receive the same wastewater inflow (15 m^3 day^{-1}) (Table 5.1), and in order to keep the same W:D ratio in the VF (Wang et al., 2014) we assumed the same pumping work time and thus, the same electricity consumption.

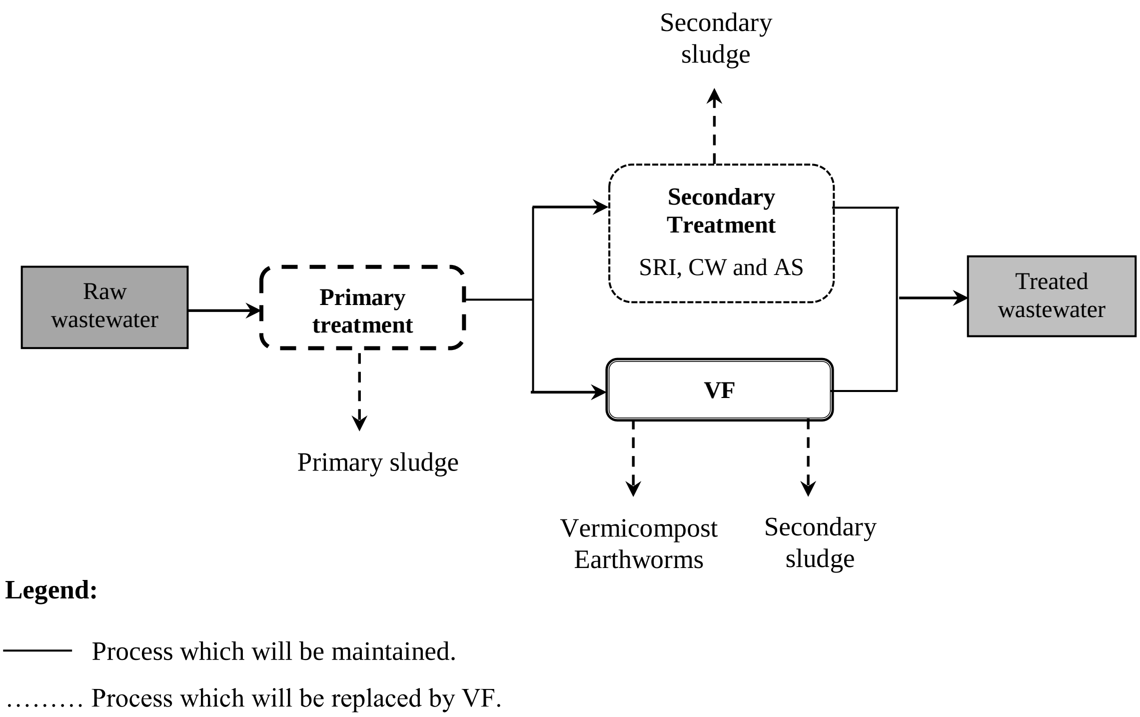


Figure 5.1 – Flowchart for SRI, CW and AS original WWTPs with VF alternative.

Since the lifetime of a typical WWTP is between 25 to 50 years (Piao et al., 2016) the expected lifetime for the mechanical equipment was set at 15 years, and at 30 years for construction works.

The principles and requirements for LCI follow the established in the ISO 14000 series standards, namely for goal and scope (ISO 14040, 2006), and inventory analysis (ISO 14044, 2006). According to the standards, the functional unit (FU) may be one cubic meter (Piao et al., 2016), or one person equivalent (PE) (Tillman *et al*. (1998), Lundin *et al*. (2000), Kärrman and Jonsson, (2001)). In the present study, the functional unit is one PE.

The system boundary was established at the WWTP fence, with a gate to gate approach, which includes the analysis from reception of the ready to use raw materials and influent watewater to the outflow of treated wastewater and wastes. It is a common procedure when unit process are specifically studied. It was assumed that the sludges from the primary treatment are deposited in landfill and those from the secondary treatment are sold as fertilizer.

Construction, operation, and dismantling phases were included, following common procedures (Zhang *et al*., 2010; Foley *et al*., 2010; Bravo and Ferrer, 2011; Liu *et al*., 2013; Li *et al*., 2013; Buyukkamaci, 2013). The LCI included detailed tracking of all inputs and outputs throughout the facilities' life cycle (construction, operation, dismantling), including assessment of raw resources or materials, water, energy and fuel, solid wastes, air and water emissions. The analysis involved several individual unit processes, being the inputs and outputs assigned to each of them. All the energy requirements for the processes identified in the LCI were first quantified in terms of fuel and electricity units. The fuel used to transport raw materials to each process is included as a part of the LCI energy requirements. Emissions are categorized as atmospheric emissions, water pollutants, and solid wastes. Atmospheric emissions (as greenhouse emissions) include carbon dioxide, methane and nitrogen oxide, and carbon dioxide equivalent, all reported as kg PE^{-1}. Water pollutants are reported as kg of pollutant per volume of wastewater per PE, which includes biochemical oxygen demand (BOD), chemical oxygen demand (COD), and total suspended solids (TSS). The volume of wastewater produced per PE was considered equal to 90 L (Paixão, 2004).

- Construction

The input flows considered in the construction phase included i) fossil fuel (kg); ii) construction materials (kg) and iii) transport of materials (km). All values are reported as per functional unit. Design data for the different treatment schemes is shown in Table 5.1.

Fossil fuel consumption was associated to i) excavator (150-153 HP), wheel loader (260-285 HP), truck (413-496 HP), bulldozer (354 HP), and a diesel powered electricity generator (80 HP) (DSS, 2019); and ii) estimated travel distances.

The time of use of on-site and electric machinery was determined considering the unit operation area required to build the infrastructures, considering they were made from precast concrete. The latter was supplied by a local company 25 km away from the WWTPs. The sewerage system was built using PVC pipes.

Fuel consumption included diesel from fuel-powered electricity generators and transport vehicles. Fuel consumptions for the generator were based on the average values indicated by Klanfar (2016). For road vehicles, heavy-duty transport vehicles (16-32 ton) in Portugal, Spain, Germany, and Luxembourg were assumed to be compliant with EURO 0-4 mix, 22 t total weight, 17.3 t max payload. Travelled distances are shown in Table 5.1.

- Operation

During operation, four main activities were considered i) wastewater treatment; ii) sludge management; iii) packing renewal, in the case of vermifiltration; and iv) substitution of equipment and parts. The wastewater treatment included the total volume of influent and effluent wastewater (m^3), therefore, assuming no losses in the process. The following flows were included: i) electricity for pumping (MJ), ii) fossil fuel consumption (kg).

The origin of the treatment sludges the physic-chemical primary treatment and the biological secondary treatment. The following production values were assumed: i) 0.11 L PE^{-1} day^{-1} for primary sludge (Paixão, 2005) with 80% moisture; and ii) 0.070 kg suspended solids, from secondary sludge produced from vermifiltration (Xing et al., 2011). The amount of nutrients in sludge was defined in 0.45% for nitrogen and 0.30% for phosphorus per dry wet matter (Lourenço, 2014). In larger WWTPs, polyaluminum chloride and aluminum sulfate are usually used as the inorganic flocculants in sludge thickening and dewatering (Piao et al., 2016), but due to the small size of the studied WWTP, they are not used. For VF, the secondary sludge is a mixture of earthworm castings and excess sludge (Xing et al., 2011). Due to the low production of secondary sludge in VF, it was assumed that all secondary sludge is recirculated.

The size of the population of earthworms in the VF increases for the first 200 days of operation, stabilizing after that at a relatively constant size for the remaining 30 years of project. The production of earthworm castings, the precursor of vermicompost in the VF, followed a kinetic production of 11.9 mg g^{-1} earthworm biomass day^{-1} (Zhao et al., 2010).

The electricity for the operation phase was obtained from the grid, using the electricity mix of the country, AC, 1-60 kV. During this phase, the only relevant electrical equipment were a wastewater pump with 1.3 HP which worked for 6 h day^{-1}, and a recirculation pump with 0.07 HP which worked for 2 h day^{-1} in order to keep the adequate HRT (Lourenço and Nunes, 2017a) and an adequate wet to dry ratio in the VF (Wang et al., 2014) described previously.

The removal of the vermicompost from the uppermost layer of the VF was made every 3 months and assumed to be made using appropriate manual tools and was assumed to used onsite to fertilize the green areas.

For sawdust, one single trip was assumed every year for each WWTP. The transport of primary sludge, secondary sludge and greases was assumed to be made at the same time, on a yearly basis in SRI-VF and CW-VF and, due to the amounts of primary sludge produced, twice a year on AS-VF. Therefore, the two flows were considered together. Transport of materials and wastes in and out of the facilities was assumed to be made by a 7.5 ton lorry, 3.3 t max payload, EURO 0-4 mix. The distance from the WWTPs to the landfill was 29.0 km (from Carrión de los Céspedes to Seville, Spain) (SRI-VF and CW-VF), and 14.4 km (from Vila Verde to Ferreiros – Braga, Portugal) (AS-VF). Other detailed information is shown in Table 5.1.

Table 5.1 – Detailed construction data and operation data for 30 years for the WWTPs (per PE).

Parameter	Reference solution			Alternative solution		
	SRI*)	CW*)	AS*)	SRI-VF**)	CW-VF**)	AS-VF**)
Construction						
Unit operation area (m^2)a)	2,000.0	594.0	95.0	12.5	12.5	50.0
Total plant area (m^2)b)	2,014.8	620.7	139.4	27.3	39.2	94.4
Flow rate (m^3 day^{-1})	15.0	15.0	60.0	15.0	15.0	60.0
Population served (No.)	120.0	120.0	500.0	120.0	120.0	500.0
Wet to dry time ratio (W:D)b)	-	-	-	1:3	1:3	1:3
Organic load (kg BOD PE^{-1} $year^{-1}$)	25.0	25.0	15.0	25.0	25.0	15.0
Earthwork volume (m^3)c)	-	-	-	0.270	0.270	0.110
Total distance travelled (km PE^{-1})d)	137.7	137.7	26.0	137.7	137.7	26.0
Operation						
BOD load (kg)e)	740.0	740.0	453.0	740.0	740.0	453.0
COD load (kg)	1,272	1,272	1,005	1,272	1,272	1,005
Pollutants removal (kg BOD)	666.4	629.4	397.2	654.6	654.6	400.5
Pollutants removal (kg COD)	1,081.1	1,017.5	828.5	960.6	960.6	758.8
Primary sludge production (kg)f)	-	1,080.0	1,080.0	1,080.0	1,080.0	1,080.0
Secondary sludge production (kg)g)	-	-	-	67-413	67-413	53-326
Total distance travelled (km)h)	0.483	0.483	0.115	0.525	0.525	0.151

a)Exclusively for secondary treatment; b) Sum of primary and secondary treatment areas; b) 6 h wet time/18 hour dry time (Wang et al., 2014); c)Data not available for the reference situation; d)Transportation includes the way back, excluding the inputs which were not used for each particular WWTP; e) The organic load discharged to the receiver; f) Based on a daily production of 0.11 L $PE^{-1}day^{-1}$ (Paixão, 2005) and a bulk density of 900 kg m^{-3} (Lourenço, 2014). According to Machado et al. (2007) SRI did not produce primary sludge; g) Production in the reference solution was not reported. We opted for recirculation of this material in VFs. Production was based on a production of 0.07 to 0.43 kg suspended solids kg^{-1} COD removed (Xing et al., 2011); h) Distance to landfill, including the way back, plus the distance travelled for sawdust transport. The reference solution does not include the sawdust transport, which was 0.042 km PE^{-1} in SRI-VF and in CW-VF, and 0.036 km PE^{-1} in AS-VF.

- Dismantling

The flows in the dismantling phase included i) fossil fuels consumption for the on-site machinery, electricity generator, concrete crushing machine and transportation (kg), and ii) construction and general wastes (kg).

Fossil fuel consumption was related with excavator (150-153 HP), wheel loader (260-285 HP), truck (413-496 HP), and bulldozer (354 HP) and stone crushing machine (50 HP) with a mean fuel consumption of 32 L $hour^{-1}$ (Klanfar, 2016). A pneumatic hammer with 3.35 HP was fed by a diesel-powered electricity generator with 80 HP and a fuel consumption of 18 L $hour^{-1}$ at full load (DSS, 2019). All the fuel consumptions were based on the same average values described above.

About 50% of the construction and demolition wastes were assumed to be crushed and recycled, following international experience (Trevor et al., 2019). Thus, detailed dismantling data does not include as wastes the materials that are recovered for reuse or recycling, which includes 50% of the total of concrete, gravel, sand, steel, and all the metal parts of the pumps. The remaining general wastes, which include steel mesh, PVC and HDPE pipes and PP, were considered to be 50% recycled. Transport of wastes out of the facilities was assumed to be made in the same conditions described in construction phase.

5.3.2. Life cycle impact assessment

The impact assessment step was made using the CML Baseline (v. 4.4, January 2005) characterization method, since it is one of the few which considers organic matter and nutrients as emissions. Normalization was made using EU25+3 (2000). No allocation was made. The selection of impacts was based on the works made by Corominas et al. (2013) and Jeppsson and Hellström (2002): Abiotic deplection (AD) (kg Sb eq.), Acidification (AC) (kg SO_2 eq.), Eutrophication (EUT) (kg PO_4^{3-} eq.); Fresh water aquatic ecotoxicity (FWT) (kg 1,4-DB eq.); Global warming potential (GWP) (kg CO_2 eq.); MAEC (Marine aquatic ecotoxicity) (kg 1,4-DB eq.); Ozone layer deplection (OLD) (kg CFC-11 eq.); Human toxicity (HT) (g 1,4-DB eq.); Photochemical oxidation (PO) (kg C_2H_4 eq.); and Terrestrial ecotoxicity (TE) (1,4-DB eq.).

5.4. Results and discussion

5.4.1. Life cycle inventory

Table 5.2 shows the most important inputs for the three WWTP scenarios under study. The area of land occupied by the alternative VF solution is much lower than in the base solution, decreasing from 16.7 to 0.230 m^2 PE^{-1} in SRI-VF; and from 4.95 to 0.330 m^2 PE^{-1} in CW-VF. Given that the size of the activated sludge tank is similar to the vermifilter, the occupied land is equal, 0.190 m^2 PE^{-1}. Despite having the same served population (120 PE), SRI-VF occupies less land area than CW-VF due to different primary treatment systems, septic tank in SRI-VF and Imhoff tank in CW-VF. Imhoff tanks are used by small communities and their underground construction minimizes land use (Tilley et al., 2014). The estimated values are close to those obtained elsewhere for VF-based solutions. In France (Combaillaux), the typical area of land for a VF was 0.25 m^2 PE^{-1} (with no pre-treatment included) (RECYCLAGUA, 2019). In rural areas of China, Nie et al (2013) found that the typical area of land for VF varied from 0.060 m^2 PE^{-1} to 0.21 m^2 PE^{-1} (with no pre-treatment included). In India, typical land area was reported to vary between 0.5 m^2 PE^{-1} and 0.6 m^2 PE^{-1} (Sinha et al., 2014; Soto and Tohá, 1998).

The amount of materials used during construction phase of the alternative were, by decreasing order, concrete (515.2 kg PE^{-1} in CW-VF), sand (46.9 kg PE^{-1} both in SRI-VF and CW-VF), gravel (44.7 kg PE^{-1} both in SRI-VF and CW-VF), and steel (18.2 kg PE^{-1} in CW-VF). These are somewhat higher than those found elsewhere for similar infrastructures (Sapkota, 2016), namely for cement (8.98 - 21.1 kg PE^{-1}), sand (18.27 - 27.46 kg PE^{-1}), aggregate (28.58 - 73.71 kg PE^{-1}), steel (1.23 - 3.13 kg PE^{-1}), and PVC (0.006 - 0.31 kg PE^{-1}). Concrete use was always higher in the alternative solution due to the need to build the vermifilter (Table 5.3). Gravel represents one of the most significant inputs during construction of CW, resulting in being one the materials with the highest variations between the base and alternative solutions. In the opposite direction go sand and iron, both used in the construction of the vermifilter (Table 5.3).

Steel, stainless steel, and pumps came from suppliers far away from the construction sites, justifying the large transport distances (stainless steel, 5.50 km PE^{-1} for both SRI/SRI-VF and CW/CW-VF, and 1.20 km PE^{-1} for AS/AS-VF; pumps, 6.33 km PE^{-1} for both SRI/SRI-VF and CW/CW-VF, and 1.40 km PE^{-1} for AS/AS-VF).

Fossil fuel consumption is proportional to the travelled distances, and the number of hours of electricity generator use in the construction and demolition phases. Steel and stainless steel transportation showed the highest fossil fuel consumption since these materials have the longest distance travelled between each manufacturer and the construction site (Table 5.2). In terms of fuel consumption, no significant alteration between the reference solution and the alternative was found due to the negligible contribution of the transport of sawdust for filling of the vermifilters.

The inventory during operation for the three WWTP is reported in Table 5.3. Discussion is henceforward referred to a period of 30 years. Both SRI and CW treat the same total inflow of 1369 m^3 PE^{-1} while AS treats 1314 m^3 PE^{-1}. Water consumption in the reference solution was of 142.7 kg PE^{-1} in SRI and CW to 1629 kg PE^{-1} in AS-VF and, in the alternative of 142.8 kg PE^{-1} both in SRI-VF and CW-VF to 965.5 kg PE^{-1} in

AS-VF. The amount of sawdust used during operation was 74.3 kg PE^{-1} for both SRI-VF and CW-VF, and 71.3 kg PE^{-1} for AS-VF. No sawdust or earthworms were used or vermicompost produced in the reference solution since they they are specific to the filter packing of VF.

Emissions to water courses were computed for the chemical oxygen demand (COD), total nitrogen and total phosphorus. The accumulated COD emissions during operation were of 311.4 kg PE^{-1} in SRI-VF and in CW-VF, and of 245.9 kg PE^{-1} in AS-VF (Table 5.3), therefore between 22% and 63% higher than in the reference solution (190.8 kg PE^{-1} in SRI, 254.4 kg PE^{-1} in CW, and 176.2 kg PE^{-1} in AS). They are nonetheless about ten times lower than those reported for treatment plants of similar size (Hospido et al., 2008), so the emission of nutrients is already well optimized in the studied base and alternative solutions.

Electricity consumption in the reference solution varied between 100 MJ PE^{-1} in SRI and CW, and 1156 MJ PE^{-1} in AS. In the alternative solution the values were different only in AS-VF by about 50% less. Electricity consumption is a key element in the overall environmental performance of a WWTP. In fact, energy use has been already identified as one of the major sustainable development indicators for wastewater treatment systems (Palme et al., 2005). Energy consumption in the reference solution and in the alternative are in line with values reported by other authors. Nie et al. (2013), in a study with tower VF to attend a served population of 230 PE, reported electricity consumptions in the range of 200 MJ PE^{-1} to 750 MJ PE^{-1}. Tillman et al. (1998) found, for a period of 30 years, for two small WWTPs in Sweden, electricity consumptions in the range of 228 MJ PE^{-1} to 576 MJ PE^{-1}. Hospido et al. (2008) studying thirteen small WWTPs in Galicia (Spain) found mean electricity consumptions in the range between 102 MJ PE^{-1} and 427.8 MJ PE^{-1}. Magar (2016) refers electricity consumptions during operation of 234 MJ PE^{-1} and 1944 MJ PE^{-1} when managing three small WWTPs in Norway and De Feo et al. (2016) refer electricity consumptions of 4320 MJ PE^{-1}, and Weiss et al. (2008) of 4.5×10^5 MJ PE^{-1}. Treating wastewater using VF, Laws (2003) refer electricity consumption in the range of 5.0×10^5 MJ PE^{-1} to 3.7×10^6 MJ PE^{-1}, due to the use of UV for disinfection. In fact, an increase of 30% in the total use of electricity can be expected when UV disinfection is used (Nie el al., 2013). Electricity demand in WWTPs is also influenced by the inflow since higher inflows show generally lower consumption per PE (Trapote et al. 2014), due to economies of scale in larger infrastructures, leading to larger but more efficient equipment, better performing automation, and better-trained staff operators CUAS (2015).

The total production of sludges during the operation period was the same on all WWTPs, equal to 1100 kg PE^{-1}, which is similar to values referred by Laws (2003) for his VF study in Chile, of near 1056 kg PE^{-1} (after correcting his moisture content to 80%). Magar (2016) refers sludge productions between 5.98 kg PE^{-1} and 50 kg PE^{-1}, Hospido (2008) indicates productions in the range of 84.6 kg PE^{-1} to 717.3 kg PE^{-1}; and Tillman et al. (1998) in the range of 96.3 kg PE^{-1} to 768 kg PE^{-1}. The prediction of vermicompost production was 16.3 kg PE^{-1} in SRI-VF and in CW-VF, and 15.6 kg PE^{-1} in AS-VF (Table 5.3).

During the dismantling phase, the main difference between the base and alternative solutions resides in the need to dismantle the vermifilter, which resulted in an added amount of construction waste of around 300 kg PE^{-1}. Other unspecified waste was also created, but in small amounts (Table 5.3). During this phase, the heavy duty equipment and generator were responsible for the consumption of a substantially larger amount of fuel in the alternative solution than in the reference (about 900 times for SRI-VF; 540 times for AS-VF; and 90 times for CW).

Table 5.2 – Estimated travel distances for transportation of inputs used in the construction of the WWTPs (per PE).

Input[a)]	**Distance to WWTP (km)**		
	SRI-VF	**CW-VF**	**AS-VF**
Pre-cast concrete	0.42	0.42	0.10
Gravel	9.7	9.7	0.01
Quartz sand	10.5	10.5	1.20
Steel (inc. stainless steel)	36.3	36.3	8.00
Geotextile fiber (PP)	10.6	10.6	4.20
Pipes (PVC)	19.2	19.2	0.500
Pipes (HDPE)	11.5	11.5	0.400
Sawdust	0.040	0.040	0.020
Earthworms[b)]	-	-	-
Pumps	39.2	39.2	8.70
Total	137.9	137.9	23.8

[a)]From each manufacturer/supplier, including the way back; [b)] Assumed to be picked in nature near the WWTPs.

Table 5.3 – LCI results for the construction, operation and dismantling phases of the three WWTPs (per PE).

Flow	Unit	Reference solution			Alternative solution		
		SRI	CW	AS	SRI-VF	CW-VF	AS-VF
Construction phase							
Concrete[a)]	kg	175.2	110.8	-	487.1	515.2	354.4
Gravel	kg	-	$1.08x10^{4}$	-	44.7	44.7	42.9
Sand[b)]	kg	9.54	0.170	0.270	46.9	46.9	45.0
Steel (inc. stainless steel)	kg	-	-	24.28	4.1	18.2	3.0
Iron	kg	169.5	0.3400	7.100	206.0	661.7	625.2
Polypropylene	kg	0.121	0.643	-	$8.07x10^{-3}$	$8.07x10^{-3}$	$7.75x10^{-3}$
PVC	kg	0.034	$3.17x10^{-3}$	-	0.137	0.137	0.033
HDPE	kg	4.57	$1.47\ x10^{-2}$	-	$7.92x10^{-3}$	$7.92x10^{-3}$	$1.90x10^{-3}$
Sawdust	kg	-	-	-	7.46	7.46	7.16
Earthworms	kg	-	-	-	7.5	7.5	7.2
Fossil fuel (on-site machinery)	kg	0.386	0.698	0.278	0.386	0.698	0.278
Fossil fuel (electricity generator)	kg	0.498	0.902	0.360	0.498	0.902	0.360
Total fossil fuel	kg	0.884	1.60	0.638	0.884	1.60	0.638
Operation							
Raw wastewater	m^3	1369.0	1369.0	1314.0	1369.0	1369.0	1314.0
Emissions to water (COD)	kg	190.8	254.4	176.2	311.4	311.4	245.9
Emissions to water (total nitrogen)[a)]	kg	39.8	76.3	63.2	77.4	77.4	55.0
Emissions to water (total phosphorus)[a)]	kg	29.3	60.7	10.5	36.6	36.6	7.50
Total nitrogen (in primary sludge)[b)]	kg	-	4.88	4.88	4.88	4.88	4.88
Total phosphorus (in primary sludge)[b)]	kg	-	3.25	3.25	3.25	3.25	3.25
Sawdust consumption[c)]	kg	-	-	-	74.3	74.3	71.3
Production of earthworms[d)]	kg	-	-	-	0.75	0.75	0.72
Production of vermicompost	kg	-	-	-	16.3	16.3	15.6
Dismantling							
Construction waste[a)]	kg	$2.43x10^{-5*}$	$1.77x10^{-4*}$	-*	291.0	312.0	222.0
General waste[b)]	kg	0.179*	1.309*	2.517*	0.119	0.119	0.114
Fossil fuel (on-site machinery)	kg	0.386	0.698	0.278	0.386	0.698	0.278
Fossil fuel (electricity generator)	kg	0.125	0.225	0.090	0.125	0.225	0.090
Fossil fuel (crushing machinery)	kg	-	-	-	0.664	1.200	0.479
Total fossil fuel	kg	0.510	0.924	0.368	1.170	2.130	0.848

[a)]As pre-cast concrete; [b)] As quartz sand; [c)] In the reference solution, the distance suppression of the sawdust transportation was negligible. [a)] Total nitrogen in raw wastewater was established as 1.75 of the NH_4^+ emissions and total phosphorus as 1.67 of the PO_4^{3-} emissions (Henze and Comeau, 2008); [b)] According with the values from Lourenço (2014); [c)] The application of sawdust during the packing renew; [d)] Biomass grow; [e)] The electricity required to pump the wastewater from primary treatment and for effluent recirculation. [a)] 50% of the construction waste was assumed to be recycled by private companies (50% of concrete + gravel + steel + sand)); [b)] Stainless steel + PP + PVC + HDPE. In the reference situation, values from construction and general waste were the ones reported by Machado et al. (2007).

5.4.2. Life cycle inventory assessment

The construction, operation and dismantling of the WWTPs lead to positive net emissions of most selected LCIA indicators (Table 5.4). The exceptions were for a minor reincorporation of abiotic resources in AS and all alternative solutions during construction, maybe due to the way how recycling of materials was incorporated into CML’s characterization factors. Due to the way how LCIA results are presented, the positive values may result in negative environmental impacts; while the opposite may happen for the negative values of the indicators. We will refer throughout the text to the signal of impacts having these relationships in mind.

Considering the reference solution, the treatment showing worst impacts in most categories throughout the life cycle is CW, followed by AS and SRI (Sums in Table 5.4). The only exception was OLD, where AS has marginally higher emissions of CFC-11 eq. made during the production of the stainless steel activated sludge prefabricated unit. In CW the construction phase contributes most to its life cycle impacts due to the earthworks involved, namely in building access roads, clearing, constructing basins and dikes, installing piping and valving, planting, seeding, liming, fertilizing, and mulching dikes and disturbed areas. In the remaining systems, operation is the phase where impacts are highest (see Figure 5.2 to 5.4). Dismantling has the lowest impact in our simulations due to the assumption of recycling 50% of demolition wastes.

In the alternative solution, the replacement of the constructed wetland or the activated sludge by the vermifilter seems to bring important environmental benefits, as reflected in an improvement in most impact categories, in particular in the construction phase. The substitution of slow rate filtration by the vermifilter results in the improvement of indicators AC and EUT, but in the deterioration of the remaining indicators. Thus, the vermifilter would be a better environmental solution than CW (Figure 5.3) and AS (Figure 5.4) in the studied WWTPs.

A more detailed analysis shows that during the operation phase, the vermifilter has lower impacts than the remaining solutions in most impact categories. The exceptions are in AD, with the indicator worsening between one and two orders of magnitude; and MAEC and TE, with only marginal deterioration of the indicators' values.

Several studies report the high relative weight of operation phase impacts of WWTPs (Emerson et al., 2005; De Feo and Iuliano, 2016), being usually electricity and sludge managment the most important flows during operation (De Feo and Iuliano, 2016). Several authors even assume that the contribution of construction and dismantling is negligible if compared with operation, especially during a relatively long lifetime (Larsen et al, 2008; Corominas et al., 2013). Our results, on the contrary, show that for very small WWTPs the impacts of the construction phase do not dilute enough throughout the lifecycle and number of served inhabitants, unlike the economies of scale of larger facilities.

Dismantling was the phase which contributed less for the impact categories on both solutions, being in some categories negligible or even zero.

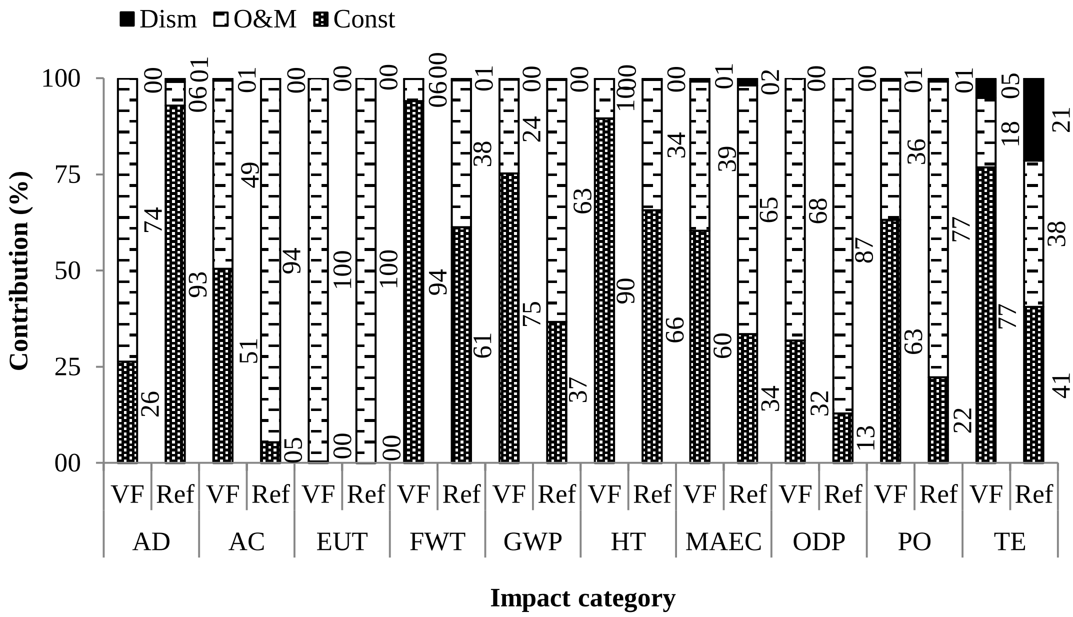
Table 5.4 – Results per impact category by each one of the three phases for the two scenarios (per PE). Negative values indicate removal of the indicator substance, therefore a positive impact.

Phase	Impact category	Reference unit	Impact result					
			Reference solution			Alternative solution		
			SRI	CW	AS	SRI-VF	CW-VF	AS-VF
Const	AD	kg Sb eq.	$1.47x10^{-5}$	$2.20x10^{-4}$	$-8.26x10^{-4}$	$-9.93x10^{-5}$	$-5.76x10^{-4}$	$-7.22x10^{-5}$
O&M			$9.77x10^{-7}$	$1.99x10^{-6}$	$1.23x10^{-5}$	$2.76x10^{-4}$	$2.76x10^{-4}$	$2.82x10^{-4}$
Dism			$1.49x10^{-7}$	$2.20x10^{-7}$	$1.98x10^{-7}$	$1.14x10^{-7}$	$1.83x10^{-7}$	$2.98x10^{-7}$
Sum			$1.58x10^{-5}$	$2.22x10^{-4}$	$-8.14x10^{-4}$	$1.77x10^{-4}$	$-3.00x10^{-4}$	$2.10x10^{-4}$
Const	AC	kg SO_2 eq.	0.0451	8.21	0.165	0.196	0.280	0.113
O&M			0.790	0.321	6.19	0.190	0.190	0.953
Dism			$1.75x10^{-3}$	$3.17x10^{-3}$	$1.26x10^{-3}$	$2.23x10^{-3}$	$3.10x10^{-3}$	$4.90x10^{-3}$
Sum			0.837	8.53	6.36	0.388	0.473	1.07
Const	EUT	kg PO_4^{3-} eq.	$7.7x10^{-3}$	1.83	0.0248	0.037	0.048	0.0197
O&M			13.1	24.3	20.7	8.92	8.92	7.49
Dism			$1.84x10^{-4}$	$3.33x10^{-4}$	$1.33x10^{-4}$	$3.90x10^{-4}$	$4.93x10^{-4}$	$7.62x10^{-4}$
Sum			13.1	26.1	20.7	8.96	8.97	7.51
Const	FWT	kg 1,4-DB eq.	0.433	40.4	0.9977	1.75	1.81	0.459
O&M			0.270	3.19	2.15	0.106	0.106	0.964
Dism			$3.80x10^{-3}$	$6.88x10^{-3}$	$2.74x10^{-3}$	$2.42x10^{-3}$	$4.14x10^{-3}$	$6.82x10^{-3}$
Sum			0.707	43.6	3.15	1.86	1.92	1.43
Const	GWP	kg CO_2 eq.	23.0	$1.78x10^{3}$	46.3	81.7	108	52.7
O&M			39.5	150	218	26.4	26.4	130
Dism			0.234	0.425	0.169	0.396	0.520	0.811
Sum			62.7	$1.93x10^{3}$	264	108	135	183
Const	HT	kg 1,4-DB eq.	6.63	349	3.47	22.5	24.5	11.4
O&M			3.43	2.88	33.1	2.57	2.57	16.8
Dism			0.038	0.068	0.027	0.041	0.059	0.094
Sum			10.1	352	36.6	25.1	27.1	28.3
Const	MAEC	kg 1,4-DB eq.	213	$2.47x10^{4}$	463	770	102	469
O&M			411	$1.11x10^{3}$	$4.67x10^{3}$	495	495	$5.93x10^{3}$
Dism			11.0	20.0	7.95	9.78	15.0	24.1
Sum			635	$2.58x10^{4}$	$5.14x10^{3}$	$1.28x10^{3}$	612	$6.42x10^{3}$
Const	OLD	kg CFC-11 eq.	$4.32x10^{-7}$	$9.96x10^{-6}$	$1.11x10^{-6}$	$1.38x10^{-6}$	$2.08x10^{-6}$	$9.96x10^{-7}$
O&M			$2.93x10^{-6}$	$2.4x10^{-6}$	$3.39x10^{-5}$	$2.4x10^{-6}$	$2.94x10^{-6}$	$7.33x10^{-7}$
Dism			$6.04x10^{-10}$	$1.10x10^{-9}$	$4.36x10^{-10}$	$8.40x10^{-10}$	$1.15x10^{-9}$	$1.80x10^{-9}$
Sum			$3.36x10^{-6}$	$1.24x10^{-5}$	$3.50x10^{-5}$	$3.78x10^{-6}$	$5.02x10^{-6}$	$1.73x10^{-6}$
Const	PO	kg C_2H_4 eq.	$3.89x10^{-3}$	0.594	0.019	0.017	0.027	0.010
O&M			0.013	0.047	0.084	$9.79x10^{-3}$	$9.97x10^{-3}$	0.051
Dism			$1.42x10^{-4}$	$2.58x10^{-4}$	$1.03x10^{-4}$	$1.69x10^{-4}$	$2.40x10^{-4}$	$3.80x10^{-4}$
Sum			0.017	0.641	0.103	0.027	0.037	0.061
Const	TE	kg 1,4-DB eq.	0.020	0.527	0.060	0.091	0.125	0.064
O&M			0.019	0.021	0.220	0.021	0.021	0.235
Dism			0.011	0.019	$7.64x10^{-3}$	$5.93x10^{-3}$	0.011	0.018
Sum			0.050	0.567	0.288	0.118	0.157	0.317

Legend:

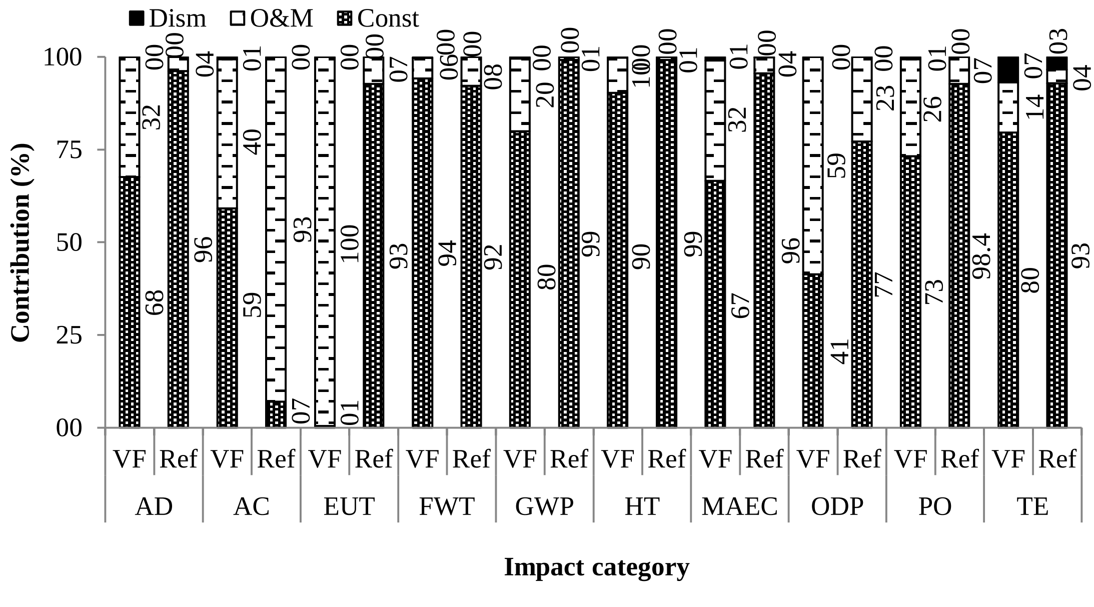
Const.: Construction phase. O&M: Operation phase (values for 30 years). Dism.: Dismantling phase; Sum: Sum of all three phases.

[a] Values or the original WWTP with VF. [b] Values of the original WWTP with the original secondary treatment.



(AD: Abiotic deplection; AC: Acidification; EUT: Eutrophication; FWT: Fresh water aquatic ecotoxicity; GWP: Global warming potential; HT: Human toxicity; MAEC: Marine aquatic ecotoxicity; OLD: Ozone layer deplection; PO: Photochemical oxidation; TE: Terrestrial ecotoxicity). The 100% correspond to the sum of the magnitude of the impacts, regardless of their signal.

Figure 5.2 – Relative weight of the phases to total impacts for SRI (Ref) and SRI-VF (VF).



(AD: Abiotic deplection; AC: Acidification; EUT: Eutrophication; FWT: Fresh water aquatic ecotoxicity; GWP: Global warming potential; HT: Human toxicity; MAEC: Marine aquatic ecotoxicity; OLD: Ozone layer deplection; PO: Photochemical oxidation; TE: Terrestrial ecotoxicity). The 100% correspond to the sum of the magnitude of the impacts, regardless of their signal.

Figure 5.3 – Relative weight of the phases to total impacts for CW (Ref) and CW-VF (VF).

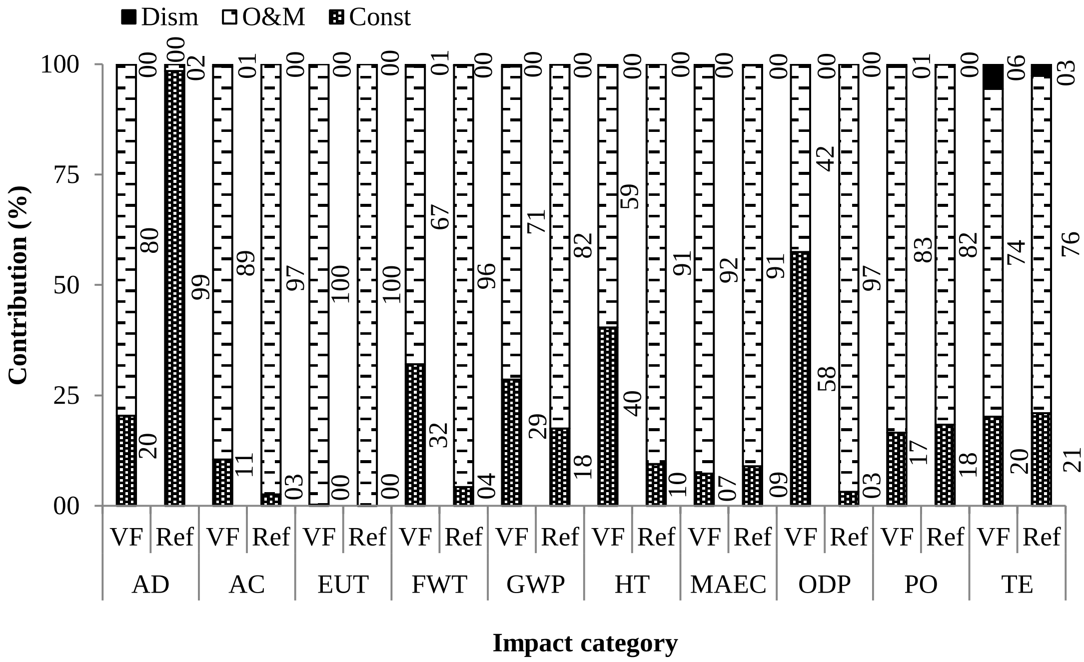


Figure 5 – Relative weight of the phases to total impacts for AS (Ref) and AS-VF (VF).

(AD: Abiotic deplection; AC: Acidification; EUT: Eutrophication; FWT: Fresh water aquatic ecotoxicity; GWP: Global warming potential; HT: Human toxicity; MAEC: Marine aquatic ecotoxicity; OLD: Ozone layer deplection; PO: Photochemical oxidation; TE: Terrestrial ecotoxicity). The 100% correspond to the sum of the magnitude of the impacts, regardless of their signal.

5.5. Conclusions

This study made a full comparative life cycle assessment for nature-based solutions for wastewater treatment in rural areas where a slow rate filtration plant, a constructed wetland, and an activated sludge (designated reference solution), were all evaluated as were, and after substitution of the secondary treatment by vermifiltration (alternative solution). Detailed lifecycle inventory was obtained for each solution, which showed that more material resources are used during construction than in any other phase. Given the small served population, the material intensity (per PE) was higher than that found in other larger facilities. On the contrary, electricity was the resource more used during operation, which was an expectable result. They are more aligned with consumption rates found elsewhere, indicating that the operation of the facilities follows standardized procedures, therefore with little optimization freedom.

The replacement of a constructed wetland by vermifiltration as secondary treatment seems to bring important environmental benefits in most impact categories, in particular in the construction phase. The replacement in facilities with slow rate filtration seems to result in the improvement of some impact indicators, such as AC and EUT, but in the deterioration of others. In conclusion, the vermifilter would be a better environmental solution than CW and AS in the studied WWTPs.

Our results show that the impacts during the construction phase outweigh those of the other phases when the number of served inhabitants is small, due to lack of economies of scale. The use of vermifiltration as alternative secondary treatment, in particular to constructed wetlands and activated sludge may help reduce the impacts of this phase. However, life cycle assessment provides but another measure of efficiency of alternative technologies, which complements socio-economic, environmental and technological constraints, studied in a complementary paper (Lourenço and Nunes, 2020).

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Final remarks

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Wastewater treatment technologies vary from conventional centralized systems, typically used on urban areas, to decentralized systems, more common in sparse dwellings and small communities of rural areas. Comparing with centralized wastewater treatment, decentralized wastewater treatment, is being progressively considered as more sustainable solution. Most do not require energy or expensive or sophisticated operation, being easy to adapt to different geographic contexts.

A general lack of consistent regulatory control over most dry rudimentary systems and primary treatment systems, may compromise water quality and human health. In the future, a mix of new policies and accurate accounting of the location, performance and degree of failure of such systems should be performed. However, forcing users and communities to face the capital, operational or repairing costs may be challenging.

Since many of the discussed technologies may be important sources of contamination with nutrients, pathogens and toxic chemicals, new opportunities are still open, which include the conversion of dry rudimentary systems into dry controlled systems. A thorough review of technologies was made in Chapter 2 comparing dry and wet solutions currently available, discussing their operational requirements, applicability, effluent output quality, efficiencies, environmental impacts, costs, challenges, as well as their advantages and implementation difficulties.

The optimization the vermifiltration process was made in Chapter 3, including the parameters hydraulic retention time, hydraulic loading rate, recirculation ratio, organic loading rate, earthworm abundance, and reactor type, for the organic matter removal from urban wastewater using a small-scale vermifiltration process, comprising two types of reactor modules – a single stage vermifilter and a four-stage vermifilter.

System performance was assessed by the removal efficiencies of BOD_5, tCOD, sCOD, pCOD, TSS and NH_4^+. In the earthworm study, four abundances were evaluated: 10 g L^{-1} (W10), 20 g L^{-1} (W20), 30 g L^{-1} (W30) and 40 g L^{-1} (W40). In the four-stage vermifilter the earthworm abundance evaluated was 20 g L^{-1}.

W20 proved to be the optimal treatment condition with efficiencies for BOD_5, tCOD, pCOD, TSS and NH_4^+ of 97.5%, 74.3%, 91.1%, 98.2% and 88.1%, for a pCOD/tCOD ratio of 0.20. The four-stage sequential VF promoted a decrease of BOD_5 (98.5%), tCOD (74.3%), pCOD (86.7%), TSS (96.6%), and NH_4^+ (99.1%).

Results indicate that sequential vermifilters can significantly improve treatment efficiencies when compared to single stage vermifilters.

Chapter 4 studied the possibility of improving treatment efficiency by altering the packing materials, namely vermicompost and sawdust in a single stage vermifilter (VF) for urban wastewater treatment. After an acclimation period of 45 days, urban wastewater from a combined sewage collection system was applied continuously for 24 hours. Earthworm stock density was of 20 g L^{-1}, HRT of 6 hours, HLR of 0.89 m^3 m^{-2} day^{-1}, and OLR of 177.63 g BOD_5 m^{-2} day^{-1}. System performance was assessed by the removal efficiencies of BOD_5, COD, TSS, NH_4^+, TN and TP, and faecal coliforms and helminth eggs elimination. Vermicompost (VE) and sawdust (SE) were tested as filter packing, using an earthworm abundance of 20 g L^{-1}.

Treatment efficiencies were 91.3% for BOD_5, 87.6% for COD, 98.4% for TSS and 76.5% for NH_4^+ in VE, and 90.5% for BOD_5, 79.7% for COD, 98.4% for TSS and 63.4% for NH_4^+ in SE. Earthworms contributed to reduce NH_4^+ and TN removal, and to increase NO_3^- concentration. No treatment was able to eliminate faecal coliforms down to guidelines values for wastewater irrigation but helminth eggs were completely eliminated.

Single stage vermifiltration system using both filter packings is inconsistent and cannot meet EU guideline values for discharge in sensitive water bodies and WHO guidelines for irrigation with treated wastewater.

To attain international guideline values for discharge in sensitive water bodies and WHO guideline values for irrigation, alternative treatment technologies are needed, namely, sequential vermifiltration systems or vermifilters followed by wetlands, working as hybrid systems suited for small communities. The former alternative was studied in Chapter 2.

In Chapter 5 vermifiltration is compared as alternative secondary treatment to small rate infiltration, a constructed wetland, and an activated sludge plant using life cycle assessment.

It was shown that material resources were more used during construction than in any other phase. Given the small served population, the material intensity was higher than that found in other larger facilities. Electricity was more used during operation, being more aligned with consumption rates found elsewhere.

Our results show that the impacts during the construction phase far outweigh those of operation and dismantling when the number of served inhabitants is small, due to lack of economies of scale, in particular for WWTP using constructed wetlands and activated sludge. The substitution of these secondary treatments by an alternative vermifiltration system was here shown to contribute to reduce the environmental impacts of the facilities.

During the making of this thesis several research questions were left unanswered, namely: i) the impact of climatic variables in the treatments' dynamic and efficiencies, affecting performance and resilience of the vermifiltration at various input load rates and temperatures; ii) which new filter packing can be used in vermifiltration, in particular using wastes from other local activities, contributing to the circular economy; and iii) the technical and legal applicability of the vermicompost to soil, as fertilizer, should be included in the boundaries of future life cycle assessment.

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References

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Abdel-Raouf, N., Al-Homaidan, A. A., Ibraheem, I. B. M. (2012). Microalgae and wastewater treatment. Saudi *J. Biol. Sci.*, 19(3), pp. 257-275.

Abdullahi, I., Nasiru, I., Saminu, A., Sagir, L., Charity, E. (2014). Design of waste stabilization pond for sewage treatment at Nigerian defence academy staff quarters, permanent site Mando Kaduna. *IJEAS*, (2), pp. 9-15.

Adugna, A. T., Andrianisa, H. A., Konate, Y., Maiga, A. H. (2019). Fate of filter materials and microbial communities during vermifiltration process. *J. Environ. Manage.*, 242, pp. 98-105.

Agnew, J., Leonard, J. (2003). The physical properties of compost. *Compost Sci. Util.*, 11(3), pp. 238-264.

Ahansazan, B., Afrashteh, H., Ahansazan, N., Ahansazan, Z. (2014). Activated sludge process overview. *Int. J. Environ. Sci. Dev.*, 5(1), pp. 1-5.

Aira, M., Domínguez, J., Monroy, J., Velando, A. (2007). Stress promotes changes in resource allocation to growth and reproduction in a simultaneous hermaphrodite with indeterminate growth. *Biol. J. Linnean. Soc*. 91, 593-600.

Alexiou, G. E., Mara, D. D. (2003). Anaerobic waste stabilization ponds: A low-cost contribution to a sustainable wastewater reuse cycle. *Appl. Biochem. Biotechnol.* 109(1-3), pp. 241-252.

Anand, C. K., Apul, D. S. (2014). Composting toilets as a sustainable alternative to urban sanitation – A review. *Waste Manage.*, 34(2), pp. 329-343.

Andreo-Martínez, P., García-Martínez, N., Almela, L. (2016). Domestic wastewater depuration using a horizontal subsurface flow constructed wetland and theoretical surface optimization: A case study under dry Mediterranean climate. *Water*, 8(10), pp. 434.

Anusha, V., Sundar, K. M. S. (2015). Application of vermifiltration on domestic wastewater treatment. *Int. J. Innov. Res. Sci. Eng. Technol*. 4(8), pp. 7301-7304.

APHA (1998). Standard Methods for the examination of water and wastewater. American Public Health Association, American Water Works Association, Water Environmental Federation, twenty ed. Washington.

Appelhof, M., Fenton, M. F. (2003). Worms Eat My Garbage: How to Set Up & Maintain a Worm Composting System, second ed. Chelsea Green Publishing Company, Hartford.

Arora, S., Rajpal, A., Kazmi, A. A. (2016). Antimicrobial Activity of Bacterial Community for Removal of Pathogens during Vermifiltration. *J. Environ. Eng*. pp. 142(5).

Arora, Kazmi, A. A. (2015). The effect of seasonal temperature on pathogen removal efficacy of vermifilter for wastewater treatment. *Water Res*. 74, pp. 88-99.

Arora, S., Rajpal, A., Kumar, T., Bhargava, R., Kazmi, A. A. (2014a). A comparative study for pathogen removal using different filter media during vermifiltration. *Water Sci. Technol*. 70, pp. 996-1003.

Arora, S., Rajpal, A., Kumar, T., Bhargava, R., Kazmi, A. A. (2014b). Pathogen removal during wastewater treatment by vermifiltration. *Environ. Technol*. 35, pp. 2493-2499.

Azim, K., Soudi, B., Boukhari, S., Perissol, C., Roussos, S., Thami Alami, I. (2017). Composting parameters and compost quality: a literature review. *Org. Agr.*, 8, pp. 141-158.

Bansal, S., Kapoor, K. K. (2000). Vermicomposting of crop residues and cattle dung with *Eisenia foetid*a. *Bioresour. Technol*. 73, pp. 95-98.

Barbagallo, S., Cirelli, G. L., Marzo, A., Milani, M., Toscano A. (2011). Hydraulic behavior and removal efficiencies of two H-SSF constructed wetlands for wastewater reuse with different operational life. *Water Sci. Technol*., 64(5), pp. 1032-1039.

Berger, W. (2011). Technology review of composting toilets - Basic overview of composting toilets (with or without urine diversion). Deutsche Gesellschaft für Internationale Zusammenarbeit (GIZ) GmbH: Eschborn, Germany, p. 21.

Berger, W. (2009). Appendix 1 – Technology review of composting toilets. List of manufacturers and commercially available composting toilets. Deutsche Gesellschaft für Internationale Zusammenarbeit (GIZ) GmbH: Eschborn, Germany, p. 9.

Berland, J. M., Cooper, P. F. (2001). Extensive wastewater treatment processes adapted to small and medium-sized communities (500 to 5000 population equivalents). Office of official publications of the European Union, Luxemburg, Luxembourg, p. 44.

Bhise, H. S., Anaokar, G. S. (2015). Design and suitability of modular vermifilter for domestic sewage treatment. *Int. J. Emerg. Eng. Res. Techn*., 44(4), pp. 44-51.

Binet, F., Fayolle, L., Pussard, M. (1998). Significance of earthworms in stimulating soil microbial activity. *Biol. Fertil. Soils* 27, pp. 79-84.

Binet, F, Trehen, P. (1992). Experimental microcosm study of the role of *Lumbricus terrestris* (Oligochaeta: Lumbricidae) on nitrogen dynamics in cultivated soils. *Soil Biol. Biochem*. 24, pp. 1501-1506.

Bouché, M. B., Soto, P. (2004). An industrial use of soil animals for environment: the treatment of organically polluted water by lumbrifiltration. Proceedings of the XIVth International Colloquium on Soil Zoology and Ecology. University of Rouen, Mont Saint Aignan, pp. 1-13.

Bouché, M. B. (1972). Lombriciens de France, Ecologie et systématique, first ed. INRA, Paris.

Borchardt, M. A., Bertz, P. D., Spencer, S. K., Battigelli, D. A. (2003). Incidence of enteric viruses in groundwater from household wells in Wisconsin. *Appl. Environ. Microbiol*. 69, pp. 1172-1180.

Boyle, W. C. (1995). Nitrogen removal from domestic wastewater in unsewered areas. In: Presentation – 8th northwest on-site wastewater treatment short course and equipment exhibition, Washington, D.C., United States of America.

Bravo, L., Ferrer, I. (2011). Life cycle assessment of an intensive sewage treatment plant in Barcelona (Spain) with focus on energy aspects. *Water Sci. Technol.*, 64, pp. 440-447.

Buckley, C., Foxon, K., Brouckaert, C., Rodda, N., Nwaneri, C., Balboni, E., Couderc, A., Magagna, D. (2008). Scientific support for the design and operation of ventilated improved pit latrines (VIPs) and the efficacy of pit latrine additives. Report to the Water Research Commission. Pollution Research Group – School of Chemical Engineering. University of KwaZulu-Natal, Durban, South Africa.

Buyukkamaci, N. (2013). Life cycle assessment applications in wastewater treatment. *J. Pollut. Eff. Cont.* 1(2), pp. 10-11.

Cagle, W. A., Johnson, L. A. (1994). On-site intermittent sand filter systems, a regulatory/scientific approach to their study in Placer County, California. In: Proc. 7th International symposium on individual and small community sewage systems, Atlanta, United States of America, pp. 283-291.

Campos, C., Guerrero, A., Cárdenas, M. (2002). Removal of bacterial and viral fecal indicator organisms in a waste stabilization pond system in Choconta, Cundinamarca (Colombia). *Wat. Sci. Tech.*, 45, pp. 61-66.

Capodaglio, A. G., Callegari, A., Cecconet, D., Molognoni, D. (2017). Sustainability of decentralized wastewater treatment technologies. *Water Pract. Tech.*, 12(2), pp. 363-477.

Cardoso-Vigueros, L., Ramírez-Camperos, E., Garzón-Zúñiga, M. (2013). Evaluation of a pilot vermifilter for the treatment of wastewater. *Ing. Agr. Biosis*., 5(2), pp. 33-44.

CAWST (2011). Introduction to low-cost sanitation – Latrine construction. A CAWST construction manual. Centre for affordable water and sanitation technology, Calgary, Canada. p. 66.

Chaggu (2004). Sustainable environmental protection using modified pit-latrines. Ph.D Thesis. Wageningen University, The Netherlands, p. 149.

Chalatsi, M., Gratziou, M. (2014). Efficiency and design parameters of waste stabilization ponds in north-east Greece. *Desalin. Water Treat.*, 56(5), pp. 1240-1247.

Chen, Z., Hu, S., Hu, C., Huang, L., Liu, H., Vymazal, J. (2016). Preliminary investigation on the effect of earthworm and vegetation for sludge treatment in sludge treatment reed beds system. *Environ. Sci. Pollut. Res. Int.*, 23, pp. 11957-11963.

Chien, B. T., Phi, D. T., Chung, B. C. (2001). Biological study on retention time of microorganisms in faecal material in urine-diverting eco-san latrines in Vietnam, In: Proceedings of the 1st International Conference on ecological sanitation, 5-8 Nov., Nanning, China.

Collado, R., Díez, R. (2010). Wastewater treatment facilities for isolated buildings. In: Proceedings of the 37th IAHS World Congress on housing science, 26-29 Oct., Santander, Spain.

Collins, K. E., Cronin, A. A., Rueedi, J., Pedley, S., Joyce, E., Humble, P. J., Tellam, J. H. (2006). Fate and transport of bacteriophage in UK aquifers as surrogates for pathogenic viruses. *Eng. Geology*, 85, pp. 33-38.

Cooper, P. F. (2005). The performance of vertical flow constructed wetland systems with special reference to the significance of oxygen transfer and hydraulic loading rates. *Water Sci. Technol.*, 51(9), pp. 81-90.

Corley, M., Rodgers, M., Mulqueen, J., Clifford, E. (2006). The performance of fibrous peat biofilters in treating domestic strength wastewater. *J. Environ. Sci. Health*, 41, pp. 811-824.

Corominas, L. Foley, G., Guest, J. S., Hospido, A., Larsen, H. F., Morera, S., Shaw, A. (2013). Life Cycle Assessment Applied to Wastewater Treatment: State of the Art. *Water Res.* 47, pp. 5480-5492.

Correia, G. M. P., Galvão, A., Freire, J. (2016). Assessment of clogging in constructed wetlands. In: Proceedings of the 17th ENASB, 14-16 Sep., Guimarães, Portugal.

Couwenberg, J. (2009). Emission factors for managed peat soils (organic soils, histosols). An analysis of IPCC default values. John Couwenberg, Greifswald Universit. Wageningen, The Netherlands. p. 16.

Crites, R. W., Ogden, M. (1998). Cost of constructed wetlands systems. In: Natural systems & water reuse, WEFTEC, 98' Water Environmental Federation, 71st Annual Conference, Orlando, United Stated of America, pp. 293-302.

Crites, R., Tchobanoglous, G. (1998). Small and decentralized wastewater management systems. Boston: WCB McGraw-Hill, Inc., pp. 703-760.

Darby, J., Tchobanoglous, G., Asri, M. N., Maciolek, D. (1996). Shallow intermittent sand filtration: Performance Evaluation. *The Small Flows Journal*, 2(1), pp. 3-14.

Darimani, H., Ito, R., Maiga, Y., Sou, M., Funamizu, N., Maiga, A. H. (2015). Effect of Post-treatment Conditions on the Inactivation of Helminth eggs (*Ascaris suum*) after the composting process. *Environ. Technol.*, 37, pp. 1-31.

Dave, J. I., Klein, R. N. (2008). Progress in Waste Management Research, first ed. Nova Publishers, New York.

Davis, L., Edwards, R., Garber, L., Isaacs, B. (2000). *A handbook of constructed wetlands – volume 1 (general considerations).* United States Environmental Protection Agency, Philadelphia, United States of America. p. 53.

De Feo, G., Ferrara, C., Iuliano, G. (2016). Comparative life cycle assessment (LCA) of two on-site small-scale activated sludge total oxidation systems in plastic and vibrated reinforced concrete.

Diaz-Valbuena, L. R., Leverenz, H. L., Cappa, C. D., Tchobanoglous, G., Horwath, W.R., Darby, J.L. (2011). Methane, carbon dioxide, and nitrous oxide emissions from septic tank systems. *Env. Sci. Technol.*, 45(7), pp. 2741-2747.

Domínguez, J. (2004). State of the art and new perspectives on vermicomposting research. In: Edwards, C.A., ed., Earthworm Ecology. Boca Raton: CRC Press, pp. 401-424.

Domínguez, J., Edwards, C. A. (2004). Vermicomposting organic wastes: A review. In: Soil Zoology for Sustainable Development in the 21st century, Shakir, S.H. and Mikhaïl, W.Z.A., ed., El Cairo, pp. 369-396.

DSS – Diesel Service Supply (2019). [online] Available at https://www.dieselserviceandsupply.com [Accessed 6 Apr. 2019].

Duarte, A. A. L. S., Canais-Seco, T., Peres, J. A., Bentes, I., Pinto, J. (2010). *Performance evaluation of Portuguese constructed wetlands for municipal wastewater treatment*. Advances in waste management, p. 6.

Duclos, L., Fillit, F., Soto, P., Bruneau, D. (2013). Lombrifiltration method and device for sanitising wastewater. Google Patents.

Eaton, A. D., Clesceri, L. S., Rice, E. W, Grennberg, A. E. (2005). Standard Methods for the Examination of Water and Wastewater, 21st ed. American Public Health Association, American Water Works Association and Water Environmental Federation, Washington.

ECOINVENT (2019). [online] Available at: http://www.ecoinvent.ch/ [Accessed 31 Oct. 2018].

Edwards, C. A., Arancon, N. Q., Sherman, R. L. (2011). Vermiculture Technology: Earthworms, Organic Wastes, and Environmental Management, first ed. CRC Press, Boca Raton, Florida.

Edwards, C. A., Arancon, N. Q. (2004). The use of earthworms in the breakdown of organic wastes to produce vermicomposts and animal feed protein, in Earthworm Ecology (Eds.), C.R.C. Press, Boca Raton, FL, London, New York, Washington, pp. 345-438.

EEA (2013). Durostat database. (http://ec.europa.eu/eurostat/472data/database. Last access: 20 October 2016).

Epstein, E. (1997). *The science of composting*. Lancaster: Technomic Publishing Company, Inc., p. 504.

Fang, C. X., Zheng, Z., Luo, X. Z., Guo, F. H. (2010). Effect of hydraulic load on domestic wastewater treatment and removal mechanism of phosphorus in earthworm ecofilter, Fresen. Environ. Bull. 19, pp. 1099-1108.

Flores, A. (2010). *Towards sustainable sanitation: evaluating the sustainability of resource-oriented sanitation*. Ph.D. Thesis. University of Cambridge, Cambridge, United Kingdom.

Foley, J., Haas, D., Hartley, K., Lant, P. (2010). Comprehensive life cycle inventories of alternative wastewater treatment systems. *Water Res*. 44, pp. 1654-1666.

Fujita, M., Suzuki, J., Sato, D., Kuwahara, Y., Yokoki, H., Kayanne, H. (2013). Anthropogenic impacts on water quality of the lagoonal coast of Fongafale Islet, Funafuti Atoll, Tuvalu. Sustain. Sci. 8(3), 381–390.

Fundación Chile (2019). [online] Available at: https://fch.cl/en/ [Accessed 14 May 2019].

Furlong, C., Enrique-Hernández, C. (2017). Optimization of fecal sludge processing via vermifiltration. In: 40th WEDC Internacional Conference, Loughborough, United Kingdom.

Furlong, C., Gibson, W. T., Oak, A., Thakar, G., Kodgire, M., Patankar, R. (2016). Technical and user evaluation of a novel worm-based, on-site sanitation system in rural India. *Waterlines*, 35(2), pp. 148-162.

Furlong, C., Gibson, W. T., Oak, A., Savant, S., Patankar, R. (2015). Fecal sludge treatment by vermifiltration: a proof of concept. Loughborough, UK: WEDC 38th International Conference.

Furlong, C., Templeton, M. R., Gibson, W. T. (2014). Processing of human feces by wet vermifiltration for improved on-site sanitation. *J. Water Sanit. Hyg. Dev.*, pp. 231-239.

Gajalakshmi, S., Ramasamy, E. V., Abbasi, S. A. (2001). Potential of two epigeic and two anecic earthworm species in vermicomposting of water hyacinth. *Bioresour. Technol*. 76, 177-181.

Galvão, A., Matos, J. (2004). Sustentabilidade de pequenos sistemas de tratamento de águas residuais. In: 7.º Congresso da Água, Associação Portuguesa dos Recursos Hídricos, Lisboa, Portugal. p.12.

Garg, P., Gupta, A., Satya, S. (2006). Vermicomposting of different types of waste using *Eisenia foetida*: a comparative study. *Bioresour. Technol*., 97(3), pp. 391-395.

George, I., Crop, P., Servais, P. (2002). Fecal coliform removal in wastewater treatment plants studied by plate counts and enzymatic methods. *Water Res*. 36, 2607–2617.

Ghermandi, A., Bixio, D., Thoeye, C. (2007). The role of free water surface constructed wetlands as polishing step in municipal wastewater reclamation and reuse. *Sci. Total Environ.*, 380, pp. 247-258.

Gikas, G. D., Tsihrintzis, V. A. (2014). Stabilization pond systems for wastewater treatment: Facility costs and environmental footprint assessment. *Global Nest. Journal,* 16(2), pp. 375-385.

Gkika, D., Gikas, G. D., Tsihrintzis, V. A. (2014). Construction and operation costs of constructed wetlands treating wastewater. *Water Sci. Technol*., 70(5), pp. 803-810.

Gill, L. W., Ring, P., Higgins, N. M. P., Johnston, P. M. (2014). Accumulation of heavy metals in a constructed wetland treating road runoff. *Ecol. Eng.*, 70, pp. 133-139.

Glaz, P., Bartosiewicz, M., Laurion, I., Reichwaldt, E. S., Maranger, R., Ghadouani, A. (2016). Greenhouse gas emissions from waste stabilization ponds in Western Australia and Quebec (Canada). *Water Res.*, 101, pp. 64-74.

Graham, J. P., Polizzotto, M. L. (2013). Pit latrines and their impact on groundwater quality - A systematic review. *Environ Health Perspect*, 121(5), pp. 521-530.

Greenway, M. (2005). The role of constructed wetlands in secondary effluent treatment and water reuse in subtropical and arid Australia. *Ecol. Eng.*, 25(5), pp. 501-509.

Guerreiro, P. Q. (2015). Low-cost on-site sanitation systems. Instituto Superior Técnico, Lisboa, Portugal, [online] p. 13. Available at: http://www.proquest.com/productsservices/ProQuestResearchLibrary.html [Accessed 8 Apr. 2015].

Gunes, K., Tuncsiper, B., Masi, F., Ayaz, S., Leszczynska, D., Hecan, N. F., Ahmad, H. (2011). Construction and maintenance cost analyzing of constructed wetland systems. *Water Practice Technol.*, 6(3), p. 6.

Gunes, K., Ayaz, S. C. (1998). Wastewater treatment by peat filtration. *Fresen. Environ. Bull.*, 7(9A-10A), pp. 777-782.

Gupta, H., Petla, S. H. (2015). A critical review on applications of vermifiltration for treatment of wastewater, *IJENAS*, 6, pp. 8-12.

Gustafson, D. M., Anderson, J. L., Christopherson, S. H. (2001). Single pass sand filters. University of Minnesota Extension Service, St. Paul, United States of America. p. 4.

Hait, S., Tare, V. (2011). Vermistabilization of primary sewage sludge. *Bioresour. Technol*. 102, pp. 2812-2820.

Hansch, S. (2003). What does it take to put together a latrine project? A practical guide for the NGO. American University, Washington, D.C., United States of America.

Harada, H., Matsui, S., Phi, D. T., Shimizu, Y., Matsuda, T., Utsumi, H. (2006). Keys for successful introduction of ecosan toilets: experiences from an ecosan project in Vietnam. Osaka University knowledge achieve, Osaka, Japan. p. 11.

Harvey, P., Baghri, S., Reed, B. (2002). Emergency Sanitation: Assessment and programme design. Water, Engineering and Development Centre. Loughborough University, Loughborough, United Kingdom.

Headley, T. R. (2006). Suitability of peat filters for on-site wastewater treatment in the Gisborne region. HAM2006-172. National Institute of Water and Atmospheric Research, Ltd., Hamilton, New Zealand, p. 22.

Hellström, D., Jönsson, L. (2006). Evaluation of small on-site wastewater treatment systems, *J. Environ. Qual.*, 17(6), pp. 728-739.

Henze, M., Comeau, Y. (2008). Wastewater Characterization. *Biol. Wastewater Treat. Princ. Model. Des*. pp. 33-52.

Hernández-Paniagua, I., Ramirez-Vargas, R., Ramos Gómez, M., Dendooven, L., González, F., Thalasso, F. (2014). Greenhouse gas emissions from stabilization ponds in subtropical climate. *Environ. Technol.*, 35. pp. 727-34.

Heubeck, S., Craggs, R. J. (2010). Biogas recovery from a temperate climate covered anaerobic pond. *Water Sci. Technol.*, 61, pp. 1019-1026.

Hill, G. B., Baldwin, S. A. (2012). Vermicomposting toilets, an alternative to latrine style microbial composting toilets, prove far superior in mass reduction, pathogen destruction, compost quality, and operational cost. *Waste Manag.*, 32, pp. 1811-1820.

Hinds, T., Brown, R., Burns, Jr. E. (2012). Reduction of fecal coliform levels in created wetlands at the Olentangy river wetland research park. In: Coliforms in experimental wetlands. Shawnee State University, Portsmouth, United Kingdom, pp. 69-72.

Hospido, A., Moreira, M. T., Feijoo, G. (2008). A comparison of municipal wastewater treatment plants for big centers of population in Galicia (Spain). *Int. J. Life Cycle Assess.* 13(1), pp. 57-64.

Howard, W., Ripley, V. F. (2016). *Design and construction.* [online] Vermicomposting toilets - A simple, proven, low tech and open source approach for ecofriendly human waste disposal. Available at: http://www.vermicompostingtoilets.net [Accessed 22 Oct. 2017].

Howarth, R. W., Sharpley, A. W., Walker, D. (2002). Sources of nutrient pollution to coastal waters in the United States: implications for achieving coastal water quality goals. *Estuaries*, 25, pp. 656-676.

Hughes, R. J., Nair, J., Ho, G. (2008). The toxicity of ammonia/ammonium to the vermifiltration wastewater treatment process. *Water Sci. Technol*. 58, pp. 1215-1220.

Hughes, R., Ho, G., Mathew, K. (2006). Conventional small and decentralized wastewater systems. Municipal wastewater management in developing countries: principles and engineering. pp. 134-167.

Inamori, R., Wang, Y., Yamamoto, T., Zhang, J., Kong, H., Xu, K., Inamori, Y. (2008). Seasonal effect on N2O formation in nitrification in constructed wetlands. Chemosphere, 73 (7), pp. 1071-1077.

Indiamart (2018). Indiamart Official Website. [online] Available at: https://www.indiamart.com/environmental-sanitation-institute/services.html [Accessed 4 Apr. 2018].

IPCC (2006). 2006 IPCC guidelines for national greenhouse gas inventories. Chapter 6: Wastewater treatment and discharge. Geneva, Switzerland, p. 28.

IRC (2012). The cost of sustaining sanitation services for 20 years can be 5-20 times the cost of building a latrine. [online] WASHCost Infosheet 2. Available at: https://www.ircwash.org [Accessed 5 Mar. 2018].

ISO 14040 (2016). *Environmental management - Life cycle assessment - Principles and framework: International Standard 14040*. International Standards Organisation, Geneva, pp. 20.

ISO 14044 (2006). *Environmental management - Life cycle assessment – Principles and framework*. Available at: https://www.iso.org/obp/ui/#iso:std:iso:14040:ed-2:v1:en [Accessed 19 August 2016].

Jasinski, S. M. (2017). Peat. USGS Mineral Resources Program. United States Geological Survey, Reston, United States of America, p. 6.

Jenkins, J. (2005). *The Humanure Handbook - A guide to composting human manure.* 2nd ed. Grove City: Joseph Jenkins, Inc., p. 308.

Jeppsson, U., Hellström, D. (2002). Systems analysis for environmental assessment of urban water and wastewater systems. *Water science and technology* 46. pp. 121-129.

Jiang, L., Liu, Y., Hu, X., Zeng, G., Wang, H., Zhou, L., Tan, X., Huang, B., Liu, S., Liu, S. (2016). The use of microbial-earthworm ecofilters for wastewater treatment with special attention to influencing factors in performance: a review, *Biores. Technol*. 200, pp. 999-1007.

Johansson, A. E., Klemedtsson, A. K., Klemedtsson, L., Svensson, B. H. (2003). Nitrous oxide exchanges with the atmosphere of a constructed wetland treating wastewater - parameters, and implications for emission factors. *Tellus B*, 55(3), pp. 737-750.

Kaczala, F. (2006). A review of dry toilet systems. Technical report 116. University of Kalmar – Department of Technology, Kalmar, Sweden, p. 62.

Kadam, A. M., Nemade, P. D., Oza, G. H., Shankar, H. S. (2009). Treatment of municipal wastewater using laterite-based constructed soil filter. *Ecol. Eng*. 35, pp. 1051-1061.

Kadlec, R. H., Wallace, S. D. (2008). *Treatment wetlands*. 2nd ed. Boca Raton: CRC Press, p. 366.

Klanfar, M., Korman, T., Kujundžić, T. (2016). Fuel consumption and engine load factors of equipment in quarrying of crushed stone. *Tehnicki Vjesnik*. 23, pp. 163-169.

Kärrman, E, Jönsson, H. (2001). Normalising impacts in an environmental systems analysis of wastewater systems. *Water Sci. Technol.*, 43(5), pp. 203-300.

Kaya, D, Dilek, F. B., Gokcay, C. F. (2007). Reuse of lagoon effluents in agriculture by post-treatment in a step feed dual treatment process. *J. Desalination*, 215, 29-36.

Kayombo, K., Mbwette, T. S. A., Katima, J. H. Y., Ladegaard, N., Jorgensen, S. E. (2005). Waste stabilization ponds and constructed wetlands design manual. Osaka: IWA Publising, p. 59.

Keim, E. K. (2015). Inactivation of pathogens by a novel composting toilet: bench-scale and field-scale studies. MSc Thesis. University of Washington, Washington, D.C.

Kõiv, M., Vohla, C., Mõtlep, R., Liira, M., Kirsimäe, K., Mander, Ü. (2009). The performance of peat-filled subsurface flow filters treating landfill leachate and municipal wastewater. *Ecol. Eng.*, 35(2), pp. 204-212.

Kumar, C., Ghosh, A. K. (2019). Fabrication of a vermifiltration unit for wastewater recycling and performance of vermifiltered water (vermiaqua) on onion (*Allium cepa*). International Journal of Recycling of Organic Waste in Agriculture.

Kumar, T., Bhargava, R., Prasad, K. S. H., Pruthi, V. (2015). Evaluation of vermifiltration process using natural ingredients for effective wastewater treatment. *Ecol. Eng.*, 75, pp. 370-377.

Kumar, T., Rajpal, A., Bhargava, R., Prasad, K. S. H. (2014). Performance evaluation of vermifilter at different hydraulic loading rate using river-bed material. *Ecol. Eng.* 62, pp. 77-82.

Kuroda, K., Murakami, M., Oguma, K., Muramatsu, Y., Takada, H., Takizawa, S. (2012). Assessment of groundwater pollution in Tokyo using PPCPs as sewage markers. *Environ. Sci. Technol*. 46(3), pp. 1455-1464.

Lachnicht, S. L., Hendrix, P. F. (2001). Interaction of earthworm Diplocardia mississippiensis (Megascolecidae) with microbial and nutrient dynamics in subtropical Spodosol. *Soil Biol. Biochem*. 33, pp. 1411-1417.

Lakshmi, C., Ranjitha, J., Vijayalakshmi S. (2014). Waste water treatment using vermifiltration Technique at institutional level. *JASR*. 4(1), pp. 581-590.

Lamichhane, K. M. (2007). On-site sanitation : a viable alternative to modern wastewater treatment plants. *Wat. Sci. Tech.* 55(1-2), pp. 433-440.

Larsen, H. F., Hauschild, M. (2008). Review of existing LCA studies on wastewater treatment technologies. In Proceedings of the SETAC-Europe Annual Meeting, 18th Warsaw, SETAC Europe, Brussels, Belgium, 25-29 May.

Laws, J. E. A. (2003) Evaluación ambiental del sistema Tohá em la remoción de Salmonella en aguas servidas domésticas. Dissertação de mestrado em gestão e planejamento ambiental. Universidade do Chile. Santiago, Chile, pp. 79.

Lens, P., Zeeman, G. Lettinga, G. (2001). Decentralized sanitation and reuse, IWA Publishing, New York, United States of America, pp. 650.

Lesikar, B., Persyn, R. (2000). Leaching chambers. L-5342. Texas Agricultural Extension Service. The Texas A&M University System, College Station, United States of America, p. 2.

Leverenz, H. L., Tchobanoglous, G., Darby, J. L. (2011). Evaluation of greenhouse gas emissions from septic systems. IWA Publishing DEC1R09. Water Environment Research Foundation. University of California, Davies, United States of America, p. 134.

Li, Y., Li, H., Xu, X., Gong, X., Zhou, Y. (2015). Application of subsurface wastewater infiltration system to on-site treatment of domestic sewage under high hydraulic loading rate. *Water Sci. Eng.*, 8(1), pp. 49-54.

Li, Y., Luo X., Huang, X., Wang, D., Zhang, W. (2013). Life cycle assessment of a municipal wastewater treatment plant: A case study in Suzhou, China. *J. Clean Prod.*, 57, pp. 221-227.

Li, Y. S., Xiao, Y. Q., Qiu, J. P., Dai, Y. Q., Robin, P. (2009). Continuous village sewage treatment by vermifiltration and activated sludge process. *Wat. Sci. Tech.*, 60(11), pp. 3001-3010.

Li, Y. S., Robin, P., Cluzeau, D., Bouché, M., Qiu, J. P., Laplanche, A., Hassouna, M., Morand, P., Dappelo, C., Callarec, J. (2008). Vermifiltration as a stage in reuse of swine wastewater: Monitoring methodology on an experimental farm. *Ecol. Eng*. 32, pp. 301-309.

Licata, M., Tuttolomondo, T., Leto, C., La Bella, S., Virga, G. (2017). The use of constructed wetlands for the treatment and reuse of urban wastewater for the irrigation of two warm-season turfgrass species under Mediterranean climatic conditions. *Water Sci. Technol*., 76(2), pp. 459-470.

Liu, B., Wei, Q., Zhang, B., Bi, J. (2013). Life cycle GHG emissions of sewage sludge treatment and disposal options in Tai Lake Watershed, China. *Sci. Total Environ.*, 447, pp. 361-369.

Liu, J., Lu, Z., Yang, J., Xing, M., Yu, F., Guo, M. (2012). Effect of earthworms on the performance and microbial communities of excess sludge treatment process in vermifilter. *Bioresour. Technol*., 117, pp. 214-221.

Lourenço, N., Nunes, L. M. (2020). Review of dry and wet decentralized sanitation technologies for rural areas: Applicability, challenges and opportunities. *Environ. Manage.* 65, pp. 642-664.

Lourenço, N., Nunes, L. M. (2017a). Optimization of a vermifiltration process for treating urban wastewater. *Ecol. Eng.*, 100, pp. 138-146.

Lourenço, N., Nunes, L. M. (2017b). Is filter packing important in a small-scale vermifiltration process of urban wastewater? *Int. J. Env. Sci. Techn*., 14(11), pp. 2411-2422.

Lourenço, N. (2014). Manual de Vermicompostagem e Vermicultura para a Agricultura Orgânica. Lisbon, Portugal: Engebook.

Lundin, M., Bergtsson, M., Molander, S. (2000). Life cycle assessment of wastewater systems: influence of system boundaries and scale on calculated environmental loads. *Environ. Sci. Technol.,* 34, pp. 180-186.

Luth, L., Robin, P., Germain, P., Lecomte, M., Landrain, B., Li, Y., Cluzeau, D. (2011). Earthworm effects on gaseous emissions during vermifiltration of pig fresh slurry. *Bior. Tech.*, 102(4), pp. 3679-3686.

Machado, A. P., Urbano, L., Brito, A. G., Janknecht, P., Salas, J. J., Nogueira, R. (2007). Life cycle assessment of wastewater treatment options for small and decentralized communities. *Water Sci Tech.*, 56(3), pp. 15-22.

Maddison, M., Mauring, T., Remm, K., Lesta, M., Mander, Ü. (2009). Dynamics of *Typha latifolia* L. populations in treatment wetlands in Estonia. *Ecol. Eng.*, 35(2), pp. 258-264.

Madrid, F. J. (2016). Aplicação da vermifiltração no tratamento de esgoto sanitário. Dissertação de Mestrado. Universidade Estadual de Campinas, Campinas, Brasil. pp. 118.

Magar, K. T. K. K. (2016). Comparative environmental performance of small scale wastewater treatment systems in Norway – life cycle analysis. MSc Thesis, Norwegian University of Life Science, Norway, pp. 153.

Maier, R. M. (1999). Biochemical Cycling, Chapter 14. In: Maier RM, Pepper IL, Gerba CP (eds). Environmental Microbiology, Academic Press, pp. 319-346.

Malek, T. E. U. A., Ismali, S. A., Ibrahim, M. H. (2012). Vermifiltration of palm oil effluent (POME). UMT 11th International Annual Symposium on Sustainability Science and Management, Terengganu.

Mander, Ü., Dotro, G., Ebie, Y., Towprayoon, S., Chiemchaisri, C., Nogueira, S. F., Mitsch, W. J. (2014). Greenhouse gas emission in constructed wetlands for wastewater treatment: A review. *Ecol. Eng.*, 66, pp. 19-35.

Mander, Ü., Jenssen, P. D. (2003). Constructed wetlands for wastewater treatment in cold climates. Southampton: WIT Press, p. 325.

Mano, A. P., Franco, M., Barros, L., Antunes, R., Houghton, S. (2003). *Tratamento de águas residuais de pequenos aglomerados - Aspectos de concepção.* Tecnologia da Água, pp. 22-26.

Manyuchi, M. M., Kadzungura, L., Boka, S. (2013). Vermifiltration of sewage wastewater for potential use in irrigation purposes using Eisenia fetida earthworms. World Academy of Science, Engineering and Technology, 78, pp. 538-542.

Mara, D. (2006). Constructed wetlands and waste stabilization ponds for small rural communities in the United Kingdom: A comparison of land area requirements, performance, and costs. *Env. Technol.*, 27, pp. 753-757.

Masi, F., Giovannelli, L., Cortés, V., Innocenti, A., Pucci, B. (2006). Reed bed treatment systems as an affordable and reliable opportunity for solving sanitary emergencies in developing countries with a tropical climate. In: Proc. 10th International conference on wetland systems for water pollution control, Lisbon, Portugal, pp. 1124.

Massoud, M. A., Tarhini, A., Nasr, J. A. (2009). Decentralized approaches to wastewater treatment and management: Applicability in developing countries. *J. Environ. Manage.*, 90, pp. 652-659.

Mburu, N., Tebitendwa, S., van Bruggen, J. J. A., Rousseau, D. P. L., Lens, P. N. L. (2013). Performance comparison and economics analysis of waste stabilization ponds and horizontal subsurface flow constructed wetlands treating domestic wastewater: A case study of the Juja sewage treatment works. J. Environ. Manage., 128C, pp. 220-225.

Meireles, M. F. (2011). Otimização da Estação de Tratamento de Águas Residuais de Crestuma. Tese de mestrado em Engenharia Química. ISEP.

Menter, U. (2016). *New sanitation techniques in the development cooperation: An economical reflection.* Stoughton: Books on Demand, p. 176.

Morgan, P. (2007). Toilets that make compost: Low-cost, sanitary toilets that produce valuable compost for crops in an African context. Stockholm Environment Institute, Stockholm, Sweden, p. 114.

Mozaheb, S. A., Ghaneian, M. T., Ghanizadeh, G. H., Fallahzadeh, M. (2010). Evaluation of the stabilization ponds performance for municipal wastewater treatment in Yazd, Iran, Middle-East. *J. Sci. Res.*, 6(1) pp. 76-82.

Muga, H. E., Mihelcic, J. R. (2008). Sustainability of wastewater treatment technologies. *J. Environ. Manage.*, 88(3), pp. 437-447.

Municipality of Anchorage (2017). Health and human services [online] Available at: http://www.muni.org/departments/health/Pages/default.aspx [Accessed 17 Nov. 2017].

Mustin, M. (1999). *Le compost: Gestion de le matière organique*. Paris: Ed. François Dubusc, pp. 954.

Naddafi, K, Hassanvand, M. S., Dehghanifard, E., Faezi R. D., Mostofi, S., Kasaee, N., Nabizadeh, R., Heidari, M. (2009). Performance evaluation of wastewater stabilization ponds in Arak-Iran. *Iran. J. Environ. Health*, 6(1), pp. 41-46.

Nasr, A., Mikhaeil, B. (2013). Treatment of domestic wastewater using conventional and baffled septic tanks. *Environ. Technol.*, 34(16) pp. 2337-2343.

Nelson, K. L. (2005). Small and Decentralized Systems for Wastewater Treatment and Reuse, in: Water Conservation, Reuse, and Recycling: Proceedings of an Iranian-American Workshop. National Academies Press, Washington, D. C., U.S.A., pp. 54-66.

NESC (2004). Maintaining your septic system - A guide for homeowners. Pipeline: Small community wastewater issues explained to the public, 15(4), p. 8.

Nguyen, A. V., Pham, N. T., Nguyen, T. H., Morel, A., Tonderski, K. (2006). Improved septic tank, a promising decentralized wastewater treatment alternative in Vietnam. In: International Conference of the National Onsite Wastewater Recycling Association and International Water Association "Water for all Life", 10-14 Mar., 2007, Baltimore, United States of America.

Nie, E., Wang, D., Yang, M., Luo, X., Fang, C., Yang, X., Su, D., Zhou, L., Zheng, Z. (2015). Tower bio-vermifilter system for rural wastewater treatment: bench-scale, pilot-scale, and engineering applications. Int. J. Environ. Sci. Technol., 12, pp. 1053-1064.

Nilsson, E., Sha, L., Qian, W., Leedo, M. (2012). Constructed wetlands wastewater treatment. VVAN01. Decentralized Water and Wastewater Treatment. [online] Available at: http://www.vateknik.lth.se/vvan01/Arkiv/Report_Constructed_wetlands_Group_D.pdf. [Accessed 13 Jan. 2018].

Nogueira, R., Brito, A. G., Machado, A. P., Salas, J. J., Vera, L., Martel, G. (2009). Economic and environmental assessment of small and decentralized wastewater treatment systems. 10th International Conference on wetland systems for water pollution control. September 23-29, 2006 Lisbon, Portugal, pp. 47-57.

OECD (2019). [online] Available at: https://www.oecd.org [Accessed 1 Nov. 2020].

Oliveira, J. M. S. C. (2007). Diagnóstico e optimização do tratamento de águas residuais em leitos de macrófitas. MSc Thesis. Universidade do Porto, Porto, Portugal.

openLCA (2019). [online] Available at: http://www.openlca.org [Accessed 16 Apr. 2019].

Ortiz, M., Raluy, R., Serra, L. (2007). Life cycle assessment of water treatment technologies: wastewater and water-reuse in a small town. *Desalination* 204, pp. 121-131

Orumieh, H. R., Mazaheri, R. (2015). Efficiency of Stabilization Ponds under different climate conditions in *Iran. J. Fundam. Appl. Sci.*, 5(1), pp. 794-805.

Osler, G. H, Sommerkorn, M. (2007). Toward a complete soil C and N cycle: incorporating the soil fauna. Ecology. 88: pp. 1611-1621.

Otis, R. J. (2001). Boundary Design: A Strategy for Subsurface Wastewater Infiltration System Design and Rehabilitation. On-Site Wastewater Treatment Procedures: Proceedings of the Ninth National Symposium on Individual and Small Community Sewage Systems. ASAE. St. Joseph MI. pp. 245-260.

Oxfam (2008). Septic tank guidelines. Oxfam technical briefing notes. [online] Available at: http://policy-practice.oxfam.org.uk/publications/septic-tank-guidelines-126711. [Accessed 4 Aug. 2017].

Paixão, M. A. (2004). Águas e esgotos em urbanizações e instalações prediais. Orion edn., Lisbon, pp. 508.

Palme, U., Lundin, M., Tillman, A.M., Molander, S. (2005). Sustainable development indicators for wastewater systems – researchers and indicator users in a co-operative case study. *Resour. Conserv. Recy.*, 43, pp. 293-311.

Patterson, R. A. (2004). Effective treatment of domestic effluent with a peat biofilter - A case study at Tingha. In: 10th National Symposium on individual and small community sewage systems proceedings. American Society of Agricultural Engineers, St. Joseph, United States of America, pp. 526-536.

Patterson, R. A. (1999). Peat treatment of septic tank effluent. In: On-site ’99 conference: Making on-site wastewater systems work. University of New England, Armindale, Australia, pp. 273-281.

Peasey, A. (2000). *Health aspects of dry sanitation with waste reuse*. WELL: London, p. 49.

Pedescoll, A., Uggetti, E., Llorens, E., Granés, F., García, D., García, J. (2009). Practical method based on saturated hydraulic conductivity used to assess clogging in subsurface flow constructed wetlands. *Ecol. Eng.*, 35(8), pp. 1216-1224.

Peña, M. V., Mara, D. (2004). Waste stabilization ponds. IRC International Water and Sanitation Centre. The Hague, The Netherlands, p. 43.

Phi, D. T., Chung, B. C., Hanh, L. T. H., Harada, H. (2004). Report on results of *Ascaris* suum tests to evaluate pathogen die-off in fecal material inside the ecosan toilets built in Dan Phuong-Lam Ha-Lam Dong-Vietnam. Report to Ministry of Health, Hanoi, Vietnam, and JICA, Tokyo, Japan.

Piao, W., Kim, Y., Kim, H., Kim, M., Kim, C. (2016). Life cycle assessment and economic efficiency analysis of integrated management of wastewater treatment plants. *J. Clean Prod*., 113, pp. 325-337.

Piluk, R. J., Peters, E. C. (1994). Small recirculating sand filters for individual homes. In: Proc. 7th International Symposium on individual and small community sewage systems. American Society of Agricultural Engineers, St. Joseph, United States of America.

Platzer, M., Cáceres, V., Fong, N., Haberl, R. (2002). Investigations and experiences with subsurface flow constructed wetlands in Nicaragua, Central America. In: Proc. 8th International Conference on wetland systems for water pollution control. International Water Association, Arusha, Tanzania, pp. 350-365.

Prasad, K. S. H., Kumar, T. (2012). Decentralized waste water treatment by vermifiltration using river bed material. 7th International Conference on Sustainable Built Environment, 14-16 December 2012, Kandi, Siri Lanka.

Puigagut, J., Villaseñor, J., Salas, J.J., Becares, E., García, J. (2007). Subsurface-flow constructed wetlands in Spain for the sanitation of small communities: A comparative study. *Ecol. Eng*., 30. pp. 312-319.

Pundsack, J., Axler, R., Hicks, R., Henneck J, Nordman D, McCarthy B. (2001). Seasonal pathogen removal by alternative on-site wastewater treatment systems. *Water Environ Res*., 73, pp. 204-212.

Pundsack, J. P. (2000). Pathogen removal by alternative on-site systems for treatment of single home wastewater flows. MSc Thesis, University of Minnesota, Minneapolis, United States of America.

Quiroga, F. J. T. (2018). Waste stabilization ponds for wastewater treatment, anaerobic pond. Iowa State University, Ames, United States of America, p.11.

RECYCLAGUA (2019). Lombri-Station – épuation des eaux usées par les lombriciens. Available at: http://www.recyclaqua.agropolis.fr/dossier_presse/lom06.html [Accessed 12 March 2019].

Redlinger, T., Graham, J., Corella-Barud, V., Avitia, R. (2001). Survival of fecal coliforms in dry-composting toilets. J. of Applied and Env. *Microbiology*, 67(9), pp. 4036-4040.

Reed, B., Smith, M., Shaw, R. (2014). Septic tank and aqua privy design. Guide 30. WEDC - Developing knowledge and capacity in water and sanitation. Loughborough University, Leicestershire, United Kingdom, p. 20.

Reid, M. C., Guan, K., Wagner, F., Mauzerall, D. L. (2014). Global methane emissions from pit latrines. *Environ. Sci. Technol.*, 48(15), pp. 8727-8734.

Reinecke, A. J., Viljoen, S. A. (1990). The influence of worm density on growth and cocoon production of the compost worm *Eisenia fetida* (Oligochaeta). *Rev. Ecol. Biol. Sol*. 27 (2), pp. 221-230.

Rego, R. R. (2012). Análise de desempenho de estações de tratamento de águas residuais recorrendo a um modelo de metabolismo. Dissertação de Mestrado. Instituto Superior Técnico, Lisboa, Portugal, pp. 106.

Reinoso, R., Torres, L. A., Bécares, E. (2008). Efficiency of natural systems for removal of bacteria and pathogenic parasites from wastewater. Sci. Total Environ., 395(2-3), pp. 80-86.

Renou, S. (2006). Life cycle analysis applied to wastewater treatment systems. Ph.D thesis. Institut National Polytechnique de Lorraine, Nancy, France, pp. 259.

Robbins, D. M., Ligon, G. C. (2014). How to design wastewater conditions for local conditions in developing countries. The International Water Association Publishing, London, United Kingdom, p. 156.

Robertson, W. D. (2010). Rates of nitrate removal in woodchip media of varying age. *Ecol. Eng.*, 36, pp. 1581-1587.

Rodrigues, J., Galvão, A., Matos, J., Heath, P. (2004). Sustainable Sewage Solutions for Small Agglomerations. *Water Sci. Technol.*, 52, 12, pp. 25-32.

Rose, C., Parker, A., Jefferson, B., Cartmell, E. (2015). The characterization of feces and urine: A review of the literature to inform advanced treatment Technology. *Crit. Rev. Environ. Sci. Technol.*, 45, pp. 1827-1879.

Roy, D., Hassan, K., Boopathy, R. (2010). Effect of carbon to nitrogen (C:N) ratio on nitrogen removal from shrimp production waste water using sequencing batch reactor. *J. Ind. Microbiol. Biotechnol*. 37, pp. 1105-1110.

Roy, C., Dube, J. P. (1994). A recirculating gravel filter for cold climates. In: Proc. 7th International Symposium on individual and small community sewage systems. American Society of Agricultural Engineers, St. Joseph, United Stated of America.

Rousseau, D. P. L., Vanrolleghem, P. A., De Pauw, N. (2004). Model-based design of horizontal subsurface flow constructed wetlands: a review. *Wat. Res*., 38, pp. 1484-1493.

Salas, J. J. (2005). Ampliación de la Planta Experimental de Carrión de los Céspedes (Sevilla), *Tecno Ambiente*, 150 (XV), pp. 34-36.

Salas, J. J. (2004). Tecnologías de depuración, Situación actual y perspectivas, IV Congrés Ibéric de gestió i planificació de l'aigua, 8-12 Diciembre 2004, Tortosa.

Samal, K., Dash, R. R., Bhunia, P. (2017). Treatment of wastewater by vermifiltration integrated with macrophyte filter: A review. *J. Environ. Chem. Eng.*, 5(3), pp. 2274-2289.

Sapkota, N. (2016). Environmental performance evaluation of decentralized wastewater treatment systems using life cycle analysis. MSc Thesis, Norwegian University of Life Sciences, Norway, pp. 94.

Sasse, L. (1998). DEWATS – Decentralized wastewater treatment in developing countries. Bremen Overseas Research and Development Association. Delhi, India. p. 161.

Schönning, C., Stenström, T. A. (2004). Guidelines for the safe use of urine and faeces in ecological sanitation. Report 2004-1. EcoSanRes, Sweden, p. 44.

Scott, E. (2002). Dry sanitation solutions. *Rural Remote Health*, 1(2), pp. 23-25.

Scholz, M., Lee, B. (2005): Constructed wetlands: a review. *Int. J. Environ. Sci. Technol.*, 62(4), pp. 421-447.

Seco, M. T. D. H. C. (2008). Avaliação de leitos de macrófitas no tratamento de águas residuais domésticas em Portugal: sistemas de fluxo superficial horizontal. MSc Thesis. Universidade de Trás-os-Montes e Alto Douro, Bragança, Portugal.

Siegrist, R. L. (1987). Soil Clogging During Subsurface Wastewater Infiltration as Affected by Effluent Composition and Loading Rate. *J. Environ. Qual*., 16(2), pp. 181-187.

Singh, R., Bhunia, P., Dash, R. R. (2017). A mechanistic review on vermifiltration of wastewater: Design, operation and performance. *J. Environ. Manage.*, 197, pp. 656-672

Sinha, R. K., Patel, U., Soni, B. K., Li, Z. (2014). Wastes and wastewaters, remediation of contaminated soils and mitigation of global warming: A review. *IJEWM*, 1(1), pp. 11-25.

Sinha, R. K., Patel, U., Soni, B. K., Li., Z. (2014). Earthworms for safe and useful management of solid wastes and wastewaters, remediation of contaminated soils and restoration of soil fertility, promotion of organic farming and mitigation of global warming: A review. *Int. J. Environ. Waste Manag.*, 1(1), pp. 11-25.

Sinha, R. K., Herat, S., Bharambe, G., Brahambhatt, A. (2010). Vermistabilization of sewage sludge (biosolids) by earthworms: converting a potential hazard destined for landfill disposal into a pathogen-free, nutritive and safe biofertilizer for farms. *Waste Manag. Res.*, 28, pp. 872-881.

Sinha, R. K. (2010). Vermiculture Technology: Reviving the Dreams of Sir Charles Darwin for Scientific Use of Earthworms in Sustainable Development Programs. Technol. Invest. 1, pp. 155-172.

Sinha, R. K., Bharambe, G., Chaudhari, U. (2008). Sewage treatment by vermifiltration with synchronous treatment of sludge by earthworms: a low-cost sustainable technology over conventional systems with potential for decentralization. *Environmentalist*, 28(4), pp. 409-420.

Somlai-Haase, C., Knappe, J., Gill, L. (2017). Carbon dioxide emissions from a septic tank soakaway in a northern maritime climate. *Sci. Total Environ.*, 586, pp. 485-491.

Song, Z. W., Zheng, Z. P., Li, J., Sun, X. F., Han, X. Y., Wang, W., Xu, M. (2006). Seasonal and annual performance of a full-scale constructed wetland system for sewage treatment in China. *Ecol. Eng*., 26, pp. 272-282.

Soto, M. A., Tohá, J. (1998). Ecological Wastewater Treatment; Advanced Wastewater Treatment, Recycling and Reuse. AWT 98 Conference, 14-16 September 1998, Milano, Italy.

Søvik, A. K., Augustin, J., Heikkinen, K., Huttunen, J. T., Necki, J. M., Karjalainen, S. M., Kløve, B., Liikanen, A., Mander, Ü., Puustinen, M., Teiter, S., Wachniew, P. (2006). Emission of the greenhouse gases nitrous oxide and methane from constructed wetlands in Europe. *J. Environ. Qual*., 35(6), pp. 2360-2373.

Stein, O., Hook, P. (2005). Temperature, plants, and oxygen: how does season affect constructed wetland performance? *J. Environ. Sci. Health. A Tox. Hazard. Subst. Environ. Eng.*, 40, pp. 1331-1342.

Stenström, T. A., Seidu, R., Ekane, N., Zurbrügg, C. (2011). Microbial exposure and health assessments in sanitation technologies and systems (EcoSanRes Programme). Stockholm Environment Institute, Stockholm, Sweden, p. 165.

Stewart, G. C., Louis, A. S. (2010). Nitrate removal and hydraulic performance of organic carbon for use in denitrification beds. *Ecol. Eng.*, 36, pp. 1588-1595.

Strande, L. (2014). Chapter 1: The global situation. In: Strande, L., Ronteltap, M., Brdjanovic, D., eds., Fecal sludge management: Systems approach for implementation and operation. IWA Publishing: London.

Study.com. (2020). Environmental Sustainability: Definition and Application. [online] Available at: https://study.com/academy/lesson/environmental-sustainability-definition-and-application.html#:~:text=Environmental%20sustainability%20is%20defined%20as,for%20long- term%20environmental%20quality.

Sundaravadivel, M., Vigneswaran, S. (2001). Constructed wetlands for wastewater treatment. *Crit. Rev. Environ. Sci. Technol.*, 31, pp. 351-409.

Tadesse, I., Green, F. B., Puhakka, J. A. (2004). Seasonal and diurnal variations of temperature, pH and dissolved oxygen in advanced integrated wastewater pond system treating tannery effluent. *J. Water Research.* 38(3), pp. 645-654.

Taylor, P., Tsihrintzis, V. A., Akratos, C. S., Gikas, G. D., Karamouzis, D., Angelakis, A. N. (2010). Performance and cost comparison of a FWS and a VSF constructed wetland system. *Environ. Technol.*, 28, pp. 37-41.

Taylor, M., Clarke, W., Greenfield, P. (2003). The treatment of domestic wastewater using small-scale vermicompost filter beds. *Ecol. Eng.* 21, pp. 197-203.

Tchobanoglous, G., Crites, R. (2003). Wastewater Engineering: Treatment, disposal and reuse. 4th ed. Metcalf & Eddy, Inc. New York: McGraw-Hill, Inc.

Tchobanoglous, G., Burton, F. L., Stensel, H. D. (2003). Wastewater Engineering: Treatment and Reuse, fourth ed. McGraw-Hill, New York.

Tilley, E., Ulrich, L., Luethi, C., Reymond, P., Zurbruegg, C. (2014). Compendium of sanitation systems and technologies. 2nd ed. Zurich: The International Water Association, p. 180.

Tillman, A. M., Svingby, M., Lundoström, H. (1998). Life cycle assessment of municipal wastewater systems. *Int. J. Life Cycle Assess*, 3, pp. 145-157.

Tomar, P., Suthar, S. (2011). Urban wastewater treatment using vermi-biofiltration system. *Desalination*, 282, pp. 95-103.

Toffey, W. E. (2008). Biofiltration Basics: Biofilter Design, Operational Overview, pp. 54-58.

Torondel, B. (2010). Sanitation ventures literature review : on-site sanitation waste characteristics. London School of Hygiene and Tropical Medicine, London, United Kingdom, p. 30.

Trapote, A., Albadalejo, A, Simón, P. (2014) Energy consumption in an urban wastewater treatment plant: the case of Murcia Region (Spain). *Civil Eng. Environ. Syst.*, 31(4), pp. 304-310.

Trevor, L., Vallero, D. (2019). Waste. Academic Press, Elsevier Inc. Second edn., pp. 803.

Truhlar, A. M., Rahm, B. G., Brooks, R. A., Nadeau, S. A., Makarsky, E. T., Walter, M. T. (2016). Greenhouse gas emissions from septic systems in New York state. *J. Environ. Qual.*, 45(4), pp. 1153-1160.

Tunçiper, B. (2009). Nitrogen removal in a combined vertical and horizontal subsurface-flow constructed wetland system. *Desalination*, 247, pp. 466-475.

Tunira, B., Saxena, K. G. (2010). Role of earthworms in soil fertility Maintenance through the Production of Biogenic Structures. Applied and Environmental Soil Science, vol. 2010, Article ID 816073, 7 pages.

Tyagi, V. K., Kazmi, A. A., Chopra, A. K. (2016). Removal of fecal indicators and pathogens in a waste stabilization pond system treating municipal in India. Water Environ. Res., 80(11), pp. 2111-2117.

Ulrich, L., Salian, P., Saul, C., Justrich, S., Luthi, C. (2016). Assessing the costs of on-site sanitation facilities (Study report). Eawag - Swiss Federal Institute of Aquatic Science and Technology, Dübendorf, Switzerland, p. 27.

UNICEF (2016). Latrine technology manual. [online] New York.: The United Nations International Children's Emergency Fund. p. 109. Available at: https://www.unicef.org/ghana/Latrine_technology_option_manual_final__a4_size.pdf.

UNITED NATIONS (2020). Transforming our world: The 2030 agenda for sustainable development. [online] New York.: Division for Sustainable Development Goals. p. 41. Available at: https://sustainabledevelopment.un.org/content/documents/21252030%20Agenda%20for%20Sustainable%20Development%20web.pdf.

United Nations Environment Programme. https://www.unenvironment.org/about-un-environment/sustainability"Sustainability." Accessed June 27, 2020.

USEPA (2018). Inventory of U.S. greenhouse gas emissions and sinks: 1990-2016. EPA 430-R-18-003. United States Environmental Protection Agency, Washington, D.C., United States of America, p. 655.

USEPA (2004a). Constructed treatment wetlands. Office of Water. EPA 843-F-03-013. United States Environmental Protection Agency, Washington, D.C., United States of America, p. 2.

USEPA (2004b). Onsite Wastewater Treatment Systems Manual. Office of Water and Office of Research and Development, Washington.

USEPA (2002). Onsite wastewater treatment systems manual. EPA/625/R-00/008. United States Environmental Protection Agency, Washington, D.C., United States of America, p. 367.

USEPA (2000). Manual constructed wetlands treatment of municipal wastewaters. EPA/625/R-99/010. United States Environmental Protection Agency, Washington, D.C., United States of America, p. 166.

USEPA (1999a). Wastewater characteristics fact sheet - Intermittent sand filters. EPA 832-F-99-067. United States Environmental Protection Agency, Washington, D.C., United States of America, p. 7.

USEPA (1999b). Decentralized systems technology fact sheet - Recirculating sand filters. EPA 832-F-99-079. United States Environmental Protection Agency, Washington, D.C., United States of America, p. 7.

USEPA (1995). Industrial waste treatment, a field study training program, volume 2, second edition. Prepared by California State University, Sacramento and California Water Pollution Control Association for the USEPA.

Van den Bossche, P., Vergels, F., Van Mierlo, J., Matheys, J., Van Autenboer, W. (2006). SUBAT: An assessment of sustainable battery technology *J. Power Sources*, 162 (2), pp. 913-919.

Vergeles, Y., Vystavna, Y., Ishchenko, A., Rybalka, I., Marchand, L., Stolberg, F. (2015). Assessment of treatment efficiency of constructed wetlands in East Ukraine. Ecol. Eng., 83, pp. 159-168.

Vohla, C., Kõiv, M., Bavor, H.J., Chazarenc, F., Mander, Ü. (2011). Filter materials for phosphorus removal from wastewater in treatment wetlands – a review. *Ecol. Eng.*, 37(1), pp. 70-89.

von Muench, E., Rieck, C., Hoffman, H. (2011). Technology review of urine diversion dehydration toilets (UDDTs) - Overview of design, operation, management and costs.

Deutsche Gesellschaft für Internationale Zusammenarbeit (GIZ) GmbHpp: Boon, Germany, p. 61.

Vymazal, J. (2011). Constructed wetlands for wastewater treatment: five decades of experience. *Env. Sci. Technol.*, 45(1), pp. 65-69.

Vymazal, J. (2009). Nitrogen removal in constructed wetlands with horizontal sub-surface flow - can we determine the key process? In: J. Vymazal, ed., Nutrient cycling and retention in natural and constructed wetlands. Backhuys Publishers: Leiden, p. 17.

Vymazal, J., Kröpfelova, L. (2008). Wastewater treatment in constructed wetlands with horizontal sub-surface flow. Springer, Dordrecht, The Netherlands, p. 566.

Vymazal, J. (2007). Removal of nutrients in various types of constructed wetlands. *Sci. Total Environ.*, 380, pp. 48-65.

Vymazal, J. (2005). Removal of Enteric Bacteria in Constructed Treatment Wetlands with Emergent Macrophytes: A Review. *J. Environ. Sci. Heal. Part A*, 40, pp. 1355-1367.

Wallace, S., Schmidt, M., Larson, E. (2011). Long-term hydrocarbon removal using treatment wetlands. SPE-145797-MS. Society of Petroleum Engineers. SPE Annual Technical Conference and Exhibition, 30 Oct. Denver, United States of America.

Wang, C. (2015). Comparative study of packing materials of filtration reactors for on-site wastewater treatment. PhD Thesis. University of Limoges, Belgium.

Wang, D., Nie, E., Luo, X., Yang, X., Liu, Q., Zheng, Z. (2015). Study of nitrogen removal performance in pilot-scale multi-stage vermi-biofilter: operating conditions impacts and nitrogen speciation transformation. *Environ. Earth Sci*. 74(5), pp. 3815-3824.

Wang, L., Guo, Z., Che, Y., Yang, F., Chao, J., Gao, Y., Zhang, Y. (2014). The effect of vermifiltration height and wet:dry time ratio on nutrient removal performance and biological features, and their influence on nutrient removal efficiencies. Ecol. Eng., 71, pp. 165-172.

Wang, L. M., Luo, X. Z., Zhang, Y. M., Lian, J. J., Gao, Y. X., Zheng Z. (2013). Effect of earthworm loads on organic matter and nutrient removal efficiencies in synthetic domestic wastewater, and on bacterial community structure and diversity in vermifiltration. Water Sci. Technol., 68(1), pp. 43-49.

Wang, L., Guo, F., Zheng, Z., Luo, X., Zhang, J. (2011). Enhancement of rural domestic sewage treatment performance, and assessment of microbial community diversity and structure using tower vermifiltration. *Bioresour. Technol.* 102, pp. 9462-9470.

Wang, D., Zhang, Z., Li, X., Zheng, W., Yang, Q., Ding, Y., Zeng, T., Cao, J., Yue, X., Shen, T., Zeng, G., Deng, J. (2010a). A full-scale treatment of freeway toll-gate domestic sewage using ecology filter integrated constructed rapid infiltration. *Eco. Eng.*, 36, pp. 827-831.

Wang, S., Yang, J., Lou, S. J. (2010b). Wastewater treatment performance of a vermifilter enhancement by a converter slag–coal cinder filter. *Ecol. Eng*. 36, pp. 489-494.

Wang, J. Q., Sun, F. Y., Wang, B. W., Xiao, J., Xue, J. R., Zhang, Y. H., Pan, S. C. (1999). Reduction of microorganisms in dry sanitation due to different adsorbents under low-temperature conditions. In: Urban stability through integrated water-related management. The 9th Stockholm water symposium, Stockholm, Sweden, pp. 396-398.

Warężak, T., Włodarczyk-Makuła, M., Sadecka, S. (2016). Accumulation of PAHs in plants from vertical flow-constructed wetland. *Desalin. Water Treat.*, 57(3), pp. 1273-1285.

Weiss, P., Eveborn, D., Kärrman, E., Gustafsson, J. P. (2008). Environmental systems analysis of four on-site wastewater treatment options. *Resour. Conserv. and Recy*. 52(10), pp. 1153-1161.

Whalen, J. K., Parmelee, R. W., Subler, S. (2000). Quantification of nitrogen excretion rates for three lumbricid earthworms using 15N. *Biol. Fert. Soils*. 32, pp. pp. 347-352.

WHO (2017a). Sanitation [online] Geneva. World Health Organization, pp. 103-128. Available at: http://www.who.int/water_sanitation_health/hygiene/om/linkingchap8.pdf. 29 November 2017. [Accessed 29 Nov. 2017].

WHO (2017b). Aquaprivies – Fact Sheet 3.8 [online] Geneva. World Health Organization. pp. 59-68. http://www.who.int/water_sanitation_health/sanitation-waste/fs3_8.pdf. [Accessed 3 Sep. 2017].

WHO/UNICEF (2017). Progress on drinking water, sanitation and hygiene - Update and SDG baselines. World Health Organization, Geneva, Switzerland, p. 116.

WHO (2006). Guidelines for the safe use of wastewater, excreta, and greywater (Vol. 1 - Policy and regulatory aspects). World Health Organization, Geneva, Switzerland, p. 114.

WHO (1992). A guide to the development of on-site sanitation [online]. Geneva. World Health Organization, pp. 59-68. Available at: http://www.who.int/water_sanitation_health/hygiene/envsan/onsitesan.pdf. [Accessed 5 Oct. 2017].

Willems, J. J. G. M., Marinissen, J. C. Y., Blair, J. (1996). Effects of earthworms on nitrogen mineralization. *Biol. Fert. Soils*. 23, pp. 57-63.

Wiesman, J. (2016). Intermittent sand filter systems - Recommended standards and guidance for performance, application, design, and operation & maintenance. Washington State Department of Health, Washington, D.C., United States of America, p. 36.

Winblad, U., Simpson-Hébert, M. (2004). *Ecological sanitation - Revised and enlarged edition*. Stockholm: Stockholm Environment Institute, p. 147.

Winblad, U. (2002). Ecological sanitation pilot project in Palestine - a project appraisal. Swedish International Development Cooperating Agency. Department for Natural Resources and the Environment, Stockholm, Sweden, p. 30.

Winkler, M. (1997). Optimal nutrient ratios for wastewater treatment Nutrients in activated sludge. Bioserve GmbH, Mainz.

Wu, S., Carvalho, P. N., Müller, J. A., Manoj, V., Dong, R. (2016). Sanitation in constructed wetlands: A review on the removal of human pathogens and faecal indicators. Sci. Total Environ., 541, pp. 8-22.

Wu, G., Zheng, D., Xing, L. (2014). Nitritation and N_2O emission in a denitrification and nitrification two-sludge system treating high ammonium containing wastewater. *Water*. 6(10), pp. 2978-2992.

Wüst, P. K, Horn, M. A., Drake, H. L. (2009). In situ hydrogen and nitrous oxide as indicators of concomitant fermentation and denitrification in the alimentary canal of the earthworm *Lumbricus terrestris*. *Appl. Environ. Microbiol*. 75, pp. 1852-1859.

Xing, M., Jian, Y., Wang, Y., Liu, J., Yu, F. (2011). A comparative study of synchronous treatment of sewage and sludge by two vermifiltrations using an epigenic earthworm *Eisenia fetida*. *J. Hazard. Mater*. 185(2-3), pp. 881-888.

Xing, M., Li, X., Yang, J. (2010). Treatment performance of small-scale vermifilter for domestic wastewater and its relationship to earthworm growth, reproduction and enzymatic activity. *African J. Biotechnol*. 9, pp. 7513-7520.

Xing, M., Yang, J., Lu, Z. (2005). Microorganism-earthworm integrated biological treatment process – a sewage treatment option for rural settlements. In: ICID 21st European Regional Conference, Frankfurt, Germany

Xu, D., Li, Y., Howard, A., Guan, Y. (2013). Effect of earthworm *Eisenia fetida* and wetland plants on nitrification and denitrification potentials in vertical flow constructed wetland. *Chemosphere*, 92, pp. 201–206.

Yadav, K., Vinod, T., Ahammed, M. (2011). Vermicomposting of source-separated human feces by *Eisenia fetida*: Effect of stocking density on feed consumption rate, growth characteristics, and vermicompost production. *Waste Manag.*, 31, pp. 1162-1168.

Yadav, K. D., Tare, V., Ahammed, M. M. (2010). Vermicomposting of source-separated human feces for nutrient recycling. *Waste Manag.*, 30, pp. 50-56.

Yang, F., Wang, L., Wang, G., Du, P., Zhang, Y. (2015). Organic Matter and Nitrogen Distribution, and Functional Groups of Filter at Earthworm Packing Bed in Vermifiltration. *Pol. J. Environ. Stud*. 24 (1), pp. 375-380.

Yi, Q., Kim, Y., Tateda, M. (2009). Evaluation of nitrogen reduction in water hyacinth ponds integrated with waste stabilization ponds. *Desalination*, 249(2), pp. 528-34.

Zhao, L., Wang, Y., Yang, J., Xing, M., Li, X., Yi, D., Deng, D. (2010). Earthworm-microorganism interactions: a strategy to stabilize domestic wastewater sludge. *Water Res*. 44(8), pp. 2572-2582.

Zhang, Q. H., Wang, X. C., Xiong, J. Q., Chen R., Cao, B. (2010). Application of life cycle assessment for an evaluation of wastewater treatment and reuse project. Case study of Xi'an. China. *Biores. Technol.*, 101, pp. 1421-1425.

Zhang, D., Gersberg, R. M., Keat, T. S. (2009). Constructed wetlands in China. *Ecol. Eng*., 35(10), pp. 1367-1378.

Zhang, D. (2008). Reuse-oriented decentralized wastewater treatment based on ecological sanitation in fast growing agglomerations. Ph.D Thesis. TU Dortmund University, Dortmund.

Zhang, J., Huang, X., Liu, C., Shi, H., Hu, H. (2005). Nitrogen removal enhanced by intermittent operation in a subsurface wastewater infiltration system. *Ecol. Eng.* 25, pp. 419-428.

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Appendices

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APPENDIX A

The pictures show the setup of the vermifiltration system described in Chapter 3.

Figure A.1 – Reactors used in the earthworm abundance study.

Figure A.2 – Detail of the reactor during the acclimation of the earthworms.

Figure A.3 – Setup of the sequential four-reactors system.

Figure A.4 – Detail showing the raw wastewater used in the studies.

Figure A.5 – Detail of the reactor used in treatment W10.