Development of an Open-Air Laboratory to Test Nitrate Removal in Bioswales

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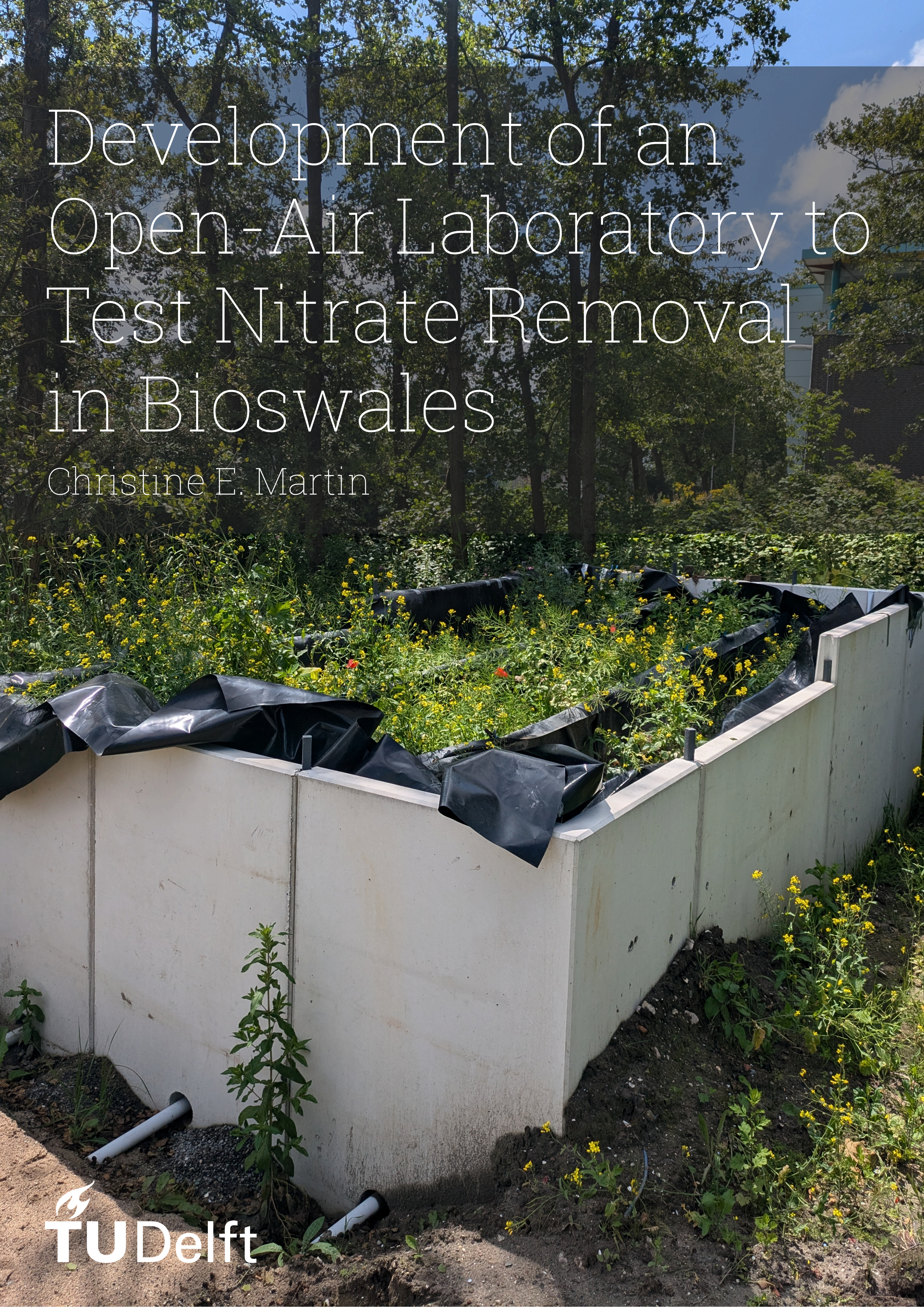

TU Delft

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by

Christine E. Martin

to obtain the degree of Master of Science
at the Delft University of Technology,
to be defended publicly on Thursday May 8, 2025 at 13:00.

Project duration:	July 13, 2024 – May 8, 2025	
Thesis committee:	Prof. dr. T. A. Bogaard,	TU Delft, supervisor
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TU Delft

Acknowledgement

Completing this master's thesis has been a journey with a lot of ups and downs, but I am incredibly grateful for the experience and for the many people who helped me along the way.

First, I would like to extend my sincere appreciation to my daily supervisor, Dr. Job van der Werf. Thank you, Job, for the constant enthusiasm and advice in completing this project, and for always making time to answer my ”quick questions”. I'm grateful to have had you as my supervisor and I've learned a great deal from you.

I would also like to express my sincere gratitude to my committee chair, Prof. Thom Bogaard, for giving me the opportunity to work on this project and get my hands dirty. Thank you for always having an open door and time for discussion, and for encouraging me to push through the (several) moments when my fieldwork felt more like disaster mitigation.

Thank you Jean-Paul, for all of your work and help at FPH. Building MeSUDa would not have been possible without you, and my days in the field would certainly have been much longer.

I would also like to thank the TU Delft WaterLab, particularly Bokure and Armand, for making time to help me with field equipment and water quality testing.

To Andie, Ine, and Noor: I can't thank you enough for your support and friendship, from both near and far, and for making Delft finally feel like home. To Sophie, I'm grateful to have had you by my side for countless hours spent building MeSUDa and even more so that we became friends through the experience. Bjorn, Karen, and Mannes, thank you for seeing me through these last months in the Afstudeerhok with a lot of laughter. To all of my other friends from the Afstudeerhok and beyond, you all made it a fantastic place to come back to after days at FPH.

Thank you Omi, for giving me some much-needed perspective and another home on this side of the ocean. Thank you Opa, for the field jacket that kept me warm and dry on some very long days.

And finally, to my family, who has supported me throughout my entire education: thank you for believing in me and for keeping my head up from thousands of miles away.

Christine E. Martin
Delft, May 2025

Abstract

The implementation of Sustainable Urban Drainage Systems (SUDS) is increasingly adopted to address stormwater management challenges in urban environments. Among these, bioswales – vegetated channels with bioretention media – are designed to mitigate urban flooding while improving runoff quality through physical and chemical pollutant removal. Despite their widespread application in the Netherlands, especially as climate adaptation measures, limited research has assessed their water treatment performance, particularly with respect to nitrate removal. Existing studies are either insufficiently monitored at full scale or confined to controlled lab settings that fail to represent field heterogeneity. Additionally, municipal bioswale monitoring remains limited due to the high resource requirements of standard sensing equipment.

This study introduced MeSUDa (Managed experimental Sustainable Urban Drainage area), an open-air lab developed to support research in bioswale development and monitoring. The setup was equipped with both low-cost and industry standard sensors, evaluated using a set of performance indicators to assess their suitability for municipal implementation. Tracer testing was conducted to characterise MeSUDa's hydrological behaviour, and a 1D advection-dispersion model was developed to investigate nitrate removal via woodchip-enhanced denitrification.

Results indicated that low-cost sensors may be appropriate for short-term monitoring but lack the reliability required for long-term municipal use. Hydrological testing revealed excessively slow infiltration and extended residence times, reducing MeSUDa's representativeness of actual bioswale conditions; however, design modifications could address these limitations. The nitrate removal model projected up to 42% removal, with further improvements possible by optimising woodchip mass and placement.

Overall, MeSUDa proved to be a flexible and adaptable research platform for bioswale development and monitoring. The study provides a foundation for continued investigation into enhanced denitrification and offers design and monitoring recommendations for future research.

Contents

Nomenclature

Abbreviations

Abbreviation	Definition
ADE	Advection Dispersion Equation
BTC	Breakthrough Curve
DO	Dissolved Oxygen
DOC	Dissolved Organic Carbon
EC	Electroconductivity
FPH	Flood Proof Holland
IC	**I**on Chromatography
K	Unsaturated Hydraulic Conductivity
LDPE	Low-Density Polyethylene
LPD	Live Pole Drain
MeSUDa	**M**anaged **e**xperimental **S**ustainable **U**rban **D**rainage **a**rea
MRT	Mean Residence Time
NSE	Nash-Sutcliffe Efficiency
OAL	Open-Air Lab
PVC	Polyvinyl Chloride
RMSE	Root Mean Square Error
SD	Standard Deviation
SM	Soil Moisture
T	Temperature
TB	Tipping Bucket
TDS	Total Dissolved Solids
TN	Total Nitrogen
TOC	Total Organic Carbon
TSS	Total Suspended Solids

1 Introduction

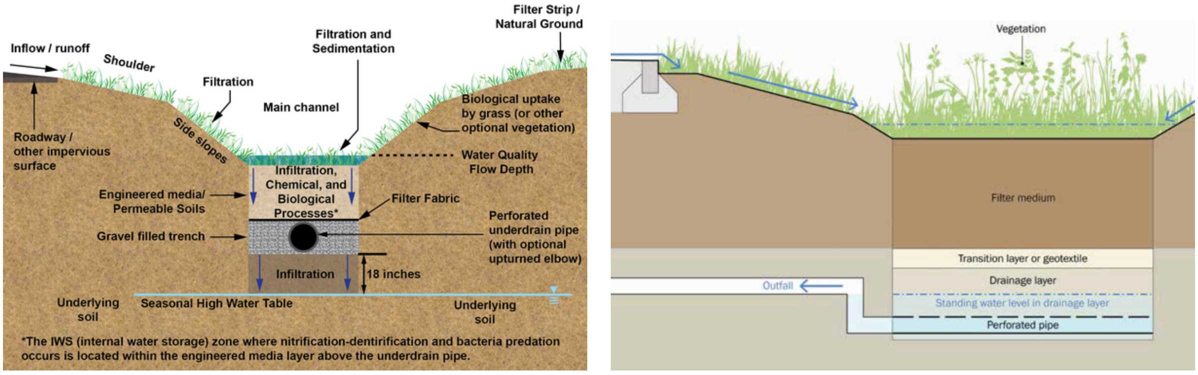
1.1. Problem description

The global acceleration of urban development has become increasingly relevant in the field of water management. Urbanisation is a key factor in reducing land permeability, thereby preventing the natural infiltration of rainwater. Coupled with intense rainfall events driven by climate change, these changes to the urban landscape have significant consequences for both water quantity and quality. Primarily, this combination of increased land impermeability and extreme rainfall is responsible for the intensification of urban runoff and subsequent flooding. In addition to the obvious challenges in water quantity management, the increase in runoff and flooding contributes to the dissemination of pollutants, threatening not only urban water quality but the health of receiving surface waters as well [1], [2], [3]. Traditional stormwater infrastructure tends to prioritise the immediate removal of excess water, facilitating the rapid transport of urban pollutants with little or no opportunity to treat polluted water [1], [3]. Concerns regarding stormwater pollution are longstanding, with research dating back over 50 years [2]. In a 2002 National Water Quality Inventory, the United States Environmental Protection Agency reported that urban runoff in the United States was the "leading source of water quality impairments to surveyed estuaries and the third-largest source of impairments to surveyed lakes" [4], [5]. While the challenge of polluted urban runoff has persisted, the recent shift towards nature-based solution (NbS) implementation in urban areas presents a valuable opportunity to change the approach to runoff management, enabling water quantity controls while also addressing water quality.

1.2. SUDS and bioswales

One subset of NbS is sustainable urban drainage systems (SUDS), which are employed in an effort to restore natural hydrological processes (e.g., infiltration or evapotranspiration) while still maintaining the runoff volume control of traditional stormwater infrastructure. SUDS are also intended to mitigate pollution stemming from urban runoff. These systems are designed to capture, infiltrate, store, slowly transport, and treat runoff at the source, as natural drainage processes would. SUDS are particularly desirable not only as a cost-effective flood prevention strategy, but also because of the myriad of other environmental, social, and economic services they provide [1], [6]. Examples of SUDS include green roofs, wetlands, permeable pavements, and swales; the latter – specifically bioswales – was the principal focus of the present research.

A bioswale (Figure 1.1a) is a shallow, vegetated channel with a bioretention element at the base, intended to facilitate the infiltration and treatment of runoff. The bioretention component typically consists of engineered fill media above a drainage layer made up of gravel and a perforated underdrain. Water treatment in a bioswale occurs via infiltration through the engineered fill, which often includes a mixture of sand and organic matter intended to enable the chemical (e.g., transformation) and physical (e.g., sorption) removal of pollutants such as heavy metals, phosphorous, or nitrogen. The top of the bioswale is generally composed of a permeable soil mix and vegetation [7], [8]. A bioswale can also include an anaerobic zone characterised by a permanent water table in the gravel layer that potentially extends into the engineered media layer (Figure 1.1b). An anaerobic zone can be created by an upturned drainage pipe and is required for the treatment of some pollutants, including nitrogen [6].



(a) Cross-section of a typical bioswale [7].

(b) Cross-section of a bioswale with anaerobic zone [6].

Figure 1.1: Bioswale cross-sections.

Swales (and therefore bioswales) are particularly well-suited for urban applications because they can be designed and implemented to best support local requirements. They are also considered cost-effective alternatives to traditional runoff management systems, requiring less investment in construction and maintenance. Swales were first implemented in the Netherlands in the late 1990s but have experienced a resurgence in popularity corresponding with the shift towards climate adaptation and resilience measures. According to Koning and Boogaard [9], they are now considered the country's "most popular climate adaptive intervention" with more than 1500 swales, the majority of which were installed in the last ten years. Despite this prevalence, however, the implementation of swales in the Netherlands is still hindered by significant knowledge gaps stemming from an overall lack of empirical design, performance, and monitoring research [9].

1.2.1. Bioswale design and function

There is no single, international guidance for the design and construction of bioswales, but several publications describe the characteristics of a functional bioswale. If the system will only be exposed to occasional rainfall events, then it should be able to completely drain and re-aerate between active periods to prohibit clogging biofilm and algae growths on the surface [6]. Permeable soil should be selected such that it promotes infiltration, and sand/loamy sand is therefore recommended. The engineering media layer generally contains mostly sand mixed with some organic matter and the porosity of the layer should be >30% [6], [7]. Multiple recommendations are made regarding the infiltration rate in the engineered layer: Ekka et al. [3] suggest 2.5 - 5 cm/h, while Woods Ballard et al. [6] recommend 10 - 30 cm/h. However, the infiltration rate of this layer should ultimately be determined based on the intended treatment provided by the bioswale. For example, for the sequestration of nitrogen, Ekka et al. [3] still recommend an infiltration rate of 2.5 - 5 cm/h, but for the removal of phosphorous a rate of 2.5 - 10 cm/h is suggested. Moreover, the infiltration rate should be designed such that sufficient treatment can take place inside of the bioswale while also ensuring that the bioswale does not flood in the event of a subsequent design storm. Woods Ballard et al. [6] recommend that removal performance should be suitable for runoff events occurring once a year on average and a swale-emptying time of 24 - 48 hours.

1.3. Literature review

1.3.1. Bioswale water quality

Despite the common assertion that bioswales are effective in treating polluted runoff and improving water quality, research in this area is considerably limited compared to water quantity challenges and benefits [3], [8], [10], [11]. This disparity is representative of the general lack of treatment assessment for field-scale SUDS and other stormwater control measures, particularly in the scope of nutrient removal [12]. With respect to bioswales, research specifically into nitrogen removal presents an interesting challenge: not only is the topic relatively unexamined, but in the research that does exist, there appears to be little consensus as to how reliable bioswales are in treating nitrogen [3], [12]. For example, in a review of available literature on total nitrogen (TN) treatment in swales, Lucke et al. [13] reported removal rates ranging from -25% (swales were responsible for *increasing* nitrogen levels) to 85%.

One example of such variability is the study conducted by Passeport et al. [14], in which the authors investigated nitrogen and phosphorous removal in two full-scale bioswales for a duration of 16 months. The bioswales consisted of slate, sand, and a topsoil layer containing 5% organic yard waste (intended to supply additional carbon, thereby enhancing denitrification). Both bioswales included an anaerobic zone, enforced using upturned underdrains. By the end of the study, one bioswale exhibited an average nitrate removal rate of 33%, while the other showed a substantially lower rate of 8%. Passeport et al. [14] attributed this disparity to longer suspected periods of anaerobic conditions in the former, which would enhance denitrification and thus increase nitrate removal. However, there was no experimental data to prove this hypothesis. Since the two bioswales were fully operational, no continuous monitoring of swale conditions (e.g., soil moisture or oxidation reduction potential) was included in the study.

Wicke et al. [15] also observed highly variable nitrate removal in their study comparing a full-scale bioswale with the performance of two smaller, technical-scale bioswales (technical-scale data sourced from [16]). Referred to as “reactive” swales by the authors, both the full and technical setups incorporated organic matter to promote denitrification. The large swale was filled with mostly sand and gravel and contained a 15 cm thick layer of woodchips, while the two smaller swales were each primarily filled with a mixture of bark mulch (1050 kg) and straw (550 kg). All three setups contained anaerobic zones. Wicke et al. [15] found that the full-scale bioswale nitrate removal ranged from under 5% to over 50%, but that the rates were closely tied to the residence time inside the setup. Removal of less than 5% was observed for much of the study, until the bioswale inflow decreased and the residence time subsequently increased. With residence times over 10 hours and increasing temperature, >25% removal was reported, while residence times beyond 48 hours enabled >50% removal. When operating at similar residence times, the full scale and technical scale setups resulted in comparable nitrate reduction rates. The authors therefore concluded that findings from technical-scale setups could be representative of field conditions. This result, as well as the original study containing the technical-scale findings, will be further discussed in section 1.3.2. Finally, similar to Passeport et al.’s study [14], Wicke et al. [15] did not appear to employ any continuous monitoring of environmental conditions in the full-scale setup. Moreover, neither the full-scale nor the technical-scale setups included a vegetation layer on top of the swale, as often characteristic of bioswales in urban settings.

As evidenced in the two previously discussed studies, the addition of organic matter to bioswales is suggested to be a critical component in facilitating denitrification and increasing nitrate removal rates. Although no literature providing design guidance on the amount or location of the organic matter in field-scale bioswales has been found, its inclusion has been described in several investigations related to nitrogen treatment (see [14], [15], [16], [17], [18], [19]). In 2021, Valenca et al. [12] published a comprehensive review of the links between nitrate removal efficiency in various stormwater control systems and different design and climate conditions. The authors concluded that nitrate removal in bioswales is "highly unreliable", but that the addition of organic amendments alongside the establishment of anaerobic zones could substantially improve removal efficiencies [12]. While field-scale optimisation of such additions has not been widely documented, lab-scale investigations frequently examine the direct effects of organic matter inclusion on denitrification.

1.3.2. Bioswale experimental scaling

Among the limited research on bioswale water quality performance, few studies focus on nitrate removal, and the majority of these are conducted at full scale using operational, "real-world" bioswales (see [11], [13], [14], [15], [20], [21]). These investigations typically do not focus on the direct effects of added organic matter, instead quantifying the overall removal performance of the setup. The analysis of organic amendments is typically left to lab-scale experiments, in the form of bioreactor column research.

One such experiment was conducted by Krause Camilo et al. [17], in which the authors tested straw and bark mulch to assess their impact on the denitrification process at a residence time of only four hours. Bioreactors were provided with NO_3 - spiked influent for one year. At the conclusion of the experiment, the straw bioreactor had generated the highest nitrate removal rate, nearly three times that of the mulch, but was prone to rapid matter depletion. When compared to field-scale removal results, the straw and mulch enhanced systems were found to perform well at significantly lower residence times (6 - 400 times lower). The removal rates accordingly suggest that the combination of straw and mulch as organic additions to a bioswale could greatly enhance denitrification. However, the applicability of this study's results is limited. The authors recognised that the study was performed entirely under lab conditions at temperatures between 20 $^\circ C$ and 22 $^\circ C$, which may have favourably influenced denitrification rates. It was therefore recommended that further long-term testing of the organic matter mixtures be conducted in field conditions [17].

Two further lab-scale studies were found which tested various organic amendments' impacts on nitrate removal (see [18], [19]). As with the research of Krause Camilo et al. [17], the results of these studies are likely not representative of actual behaviour in field conditions. The controlled lab environment is beneficial in that it can generate very specific test conditions that may well be consistent with conditions in particular bioswale setups. Nonetheless, it is not possible for lab-scale testing to fully replicate the heterogeneity inherent to full-scale bioswales and their surroundings. It is therefore unlikely that lab-scale findings can translate directly to full-scale application. The divide between lab and full-scale application can be bridged by the implementation of technical-scale setups, sometimes referred to as open-air labs (OALs). OALs are a valuable addition to bioswale research, as they allow for both representative and controllable experimentation, essentially combining the benefits of the lab- and field-scale setups. Furthermore, as evidenced in the research of Wicke et al. [15], findings from OALs can be applied to the function of full-scale setups.

Despite the apparent need for and relevance of technical-scale research, only two cases of OAL use were found in a review of current bioswale literature. The first was previously discussed in section 1.3.1 in relation to research conducted by Wicke et al. [15]. This OAL, consisting of two technical-scale swales, was operated from 2010 - 2012 at a field site belonging to the German Federal Environmental Agency (UBA). The findings generated during this period and used as a comparison for the work of Wicke et al. [15] were published by Krause Camilo et al. in 2014 [16]. Although relevant as a comparison for the field-scale setup of Wicke et al. [15], the UBA OAL did not exhibit the characteristics specifically relevant for OAL use in bioswale nitrate removal assessment. The setup served as a testing site for a variety of water quality challenges, not only nitrate removal [16]. Moreover, as described in [15], the two swales were almost completely filled with organic matter, and did not appear to contain any soil or vegetation as expected in a bioswale.

A second OAL was used in the research of Landon et al. [10]. This study investigated several bioswale features in relation to hydrological performance and the removal of total suspended solids (TSS), nitrogen, and phosphorous. Similar to Wicke et al. [15], the OAL served as a comparison for full-scale bioswales. However, the setup was subject to certain limitations. First, in-situ data was only collected at the inlets and outlets of the technical swales and appeared to have been limited to water quantity parameters. Sensors were not placed inside of the swales to monitor internal conditions. It was unclear whether water quality parameters were measured in-situ as well, or if samples were collected for lab analysis. Finally, although nitrate removal was assessed, organic matter amendments were not included in either the full-scale or technical setups [10].

1.3.3. Bioswale monitoring

Despite the recent prevalence of bioswale construction in the Netherlands, most municipalities do not currently monitor swale performance and there are no national monitoring guidelines in place. The little monitoring that does occur does not follow a standard methodology, and findings are often not well-distributed or comparable [9]. The result of this deficit is a distinct lack of information on Dutch bioswale performance, as noted by Oosterveld [22]. While conducting research into bioswale operation in the city of Rotterdam, Oosterveld [22] found limited empirical data that could be used to validate modelled findings. The author specifically stressed the requirement for continuous and long-term monitoring, encompassing a variety of environmental conditions. The Dutch data gap appears to be representative of a larger scarcity in performance data when it comes to sustainable stormwater infrastructure. Meixner et al. [23] attribute this deficiency to the “the logistical limitations of scientific research teams to be able to mount the large-scale sampling efforts needed to collect data” and argue that this data gap impedes the widespread implementation of SUDS.

The number of parameters required for bioswale performance assessment is also a barrier to widespread monitoring. Research suggests that data on water level, discharge, temperature, soil moisture, infiltration, and precipitation should be collected to properly characterise municipal bioswale function [9], [22], [23]. If available personnel are limited for data collection, as suggested by Meixner et al. [23], then these parameters should be measured in-situ, using automated monitoring equipment. However, municipalities may also experience funding limitations for the purchase and maintenance of such equipment [24], [25]. Additionally, since bioswales are often installed as a cost-effective alternative to traditional stormwater infrastructure [9], significant spending on sensors does not necessarily align with the overall philosophy of bioswale implementation.

Hamel et al. [24] contend that the combination of budget considerations and constrained monitoring “know-how” presents an opportunity for the use of low-cost sensors. The authors presented benefits and drawbacks in low-cost sensing based on their own experiences and broader literature applications. They cite the increasing popularity of the devices due to the cost-effectiveness enabled by the mass production of electronics, as well as the possibility of using the sensors for real-time monitoring. However, the authors also discuss several barriers in using low-cost sensors, namely the continued requirement for technical knowledge to properly collect research-grade data and the lack of in-field testing of the sensors [24]. Indeed, performance data on low-cost sensing in the context of bioswale monitoring is sparse. Meixner et al. [23] used low-cost sensing equipment (including soil moisture and temperature sensors) to measure bioswale water infiltration and retention as part of a greater citizen science program. While the installation of these systems was described by the authors, the performance of the equipment was not.

The relevance of field versus lab testing was briefly highlighted in the work of Slongo et al. [26], who compared the performance of 19 low-cost water quality sensors with that of an industry standard multiparameter probe. The authors found relative measurement errors ranging from -0.33 - 33.77%, with most error results remaining below 5% and concluded that the low-cost devices could serve as “complementary alternatives” for monitoring real-time changes in water quality [26]. However, this study was not performed under field-conditions. Performance of these 19 sensors may therefore be significantly different if applied in monitoring water quality in actual bioswales. Slongo et al. [26] concluded that further research was required to assess the long-term reliability and durability of low-cost sensors, as well as their ongoing maintenance requirements.

1.4. Gaps in bioswale research

Research regarding bioswale performance in improving water quality is limited, exceptionally so in the context of nitrate removal. Moreover, the existing findings on nitrate removal ability are not consistent. This variability is attributed to differences in environmental conditions, as well as whether or not organic matter amendments are included in bioswale setups. Findings are generated primarily in full-scale and lab-scale setups, with the latter being the principal setting for the detailed study of organic matter effects on denitrification. The use of technical-scale setups, or OALs, to study nitrate removal in bioswales with a specific consideration of organic amendments is limited. As discussed in section 1.3.2, OALs are particularly advantageous for bioswale research as they combine the controlled conditions of lab-scale experiments with the heterogeneity (and therefore representativeness) inherent to full-scale experiments. However, the few existing OAL setups are limited in their representativeness of urban bioswales and feature only minimal monitoring.

Furthermore, the lack of continuous in-situ monitoring in existing OALs is paralleled by insufficient municipal monitoring of active bioswales. This combination presents the opportunity to equip OALs with continuous sensing equipment not only for experimental monitoring, but also to evaluate the suitability of sensors for municipal implementation. Given the resource limitations that municipalities may face in launching bioswale monitoring programs, OALs could serve as a testing ground for low-cost sensors. Such a setup would also provide the additional benefit of generating in-field data for low-cost sensors that may otherwise only be assessed in lab conditions.

1.5. Research questions and approach

Based upon the identified research gaps, the current study sought to answer the question: **Can an open-air lab (OAL) facilitate the development and monitoring of bioswales?**

The project involved the design and construction of an experimental facility to be used for the current research as well as for future studies pertaining to bioswales and water quality. It was also of interest to investigate how best to monitor bioswales and their effectiveness, particularly from the standpoint of a municipality employing them to address local water quality concerns. Finally, the project included a combined experimental and modelling case, intended to demonstrate the OAL’s ability to effectively test strategies for urban water quality remediation.

The following sub-questions were defined to divide the project into distinct steps, culminating in an answer to the primary research question. In practice, these questions were ultimately addressed by a combination of results from the different project sections (Figure 1.2).

1. Can the OAL be designed and constructed such that it can replicate the hydrological behaviour of a bioswale?
2. Do low-cost sensors provide a suitable alternative to industry standard sensors for local monitoring of water quantity and quality within active bioswales?

3. Can the OAL be employed to assess the potential for enhanced nitrate removal via the addition of organic carbon sources, based purely on hydrological insights?

Using the above questions as guidelines, the following project sections were determined.

Design and construction: The design and construction of the OAL was partially based in the plans developed by former MSc student Cahill in their thesis research (see [27]), but several modifications and additions were made to meet the current project's needs. The goal of this phase was to complete the OAL facility in a manner that replicated the physical setup and hydrological behaviour of a bioswale, but still allowed for the experimental control and monitoring characteristic of a lab setup.

Instrumentation: The instrumentation phase of the project consisted of the selection, installation, and testing of the sensors and equipment required to monitor the OAL during the subsequent experimental phase. Key performance indicators were developed and used to assess whether sensors were suitable for implementation in a municipal setting. Included in this assessment was the direct comparison between low-cost and industry standard instrumentation.

Experiments and modelling: The final part of the project focused on the hydrological characterisation of the OAL, as well as assessing the setup's potential to promote enhanced nitrate removal. The former included infiltration testing and salt injection tracer testing. Findings generated by the tracer tests were used as a basis for a nitrate removal model. A qualitative decomposition analysis was also performed in the OAL to indicate whether nitrate removal processes were feasible.

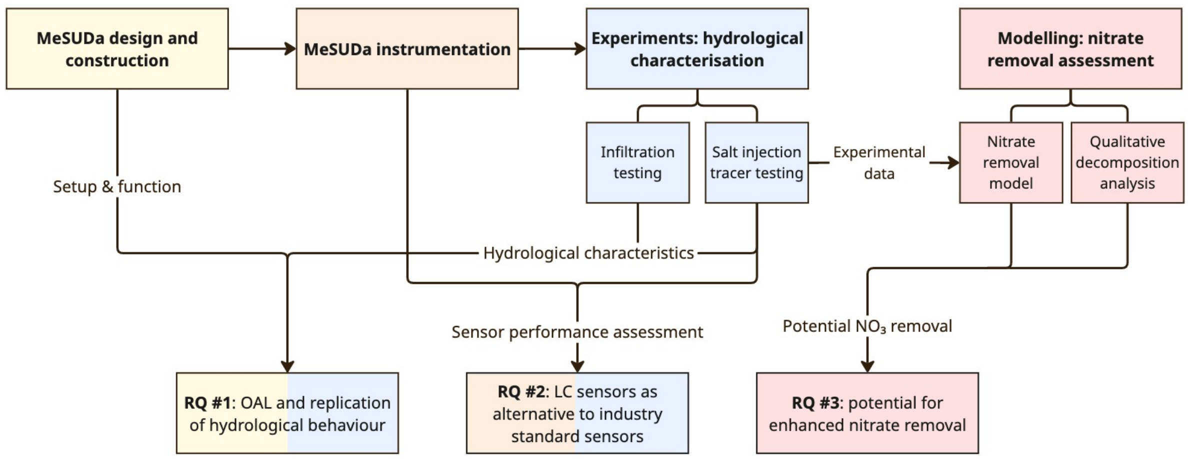


Figure 1.2: Approach to answering research questions with results from different project sections.

1.6. Structure of the report

Chapter 2 introduces the basis for the facility in conjunction with the contributions of Cahill and describes the methodology behind each project section. Cahill's contributions feed into the design and construction of the OAL. The sensor selection process is outlined, including functionality requirements and an explanation of purchasing decisions. The calibration and validation procedures for the selected sensors are also detailed, as is the assessment strategy used to compare the low-cost and industry standard sensors. Sensor installation is also detailed in this chapter. Finally, the methods used to assess the OAL's hydrological and nitrate removal characteristics are presented. This includes the tracer testing procedure, decomposition analysis strategy, and modelling methodology.

Chapter 3 presents the results of each project section. First, the resulting setup and function of the OAL is detailed, including challenges encountered during construction and testing. The findings pertaining to sensor validation, observations, and performance assessment are then presented. Thereafter, the findings of the experimental tracer tests are reported followed by the results of the nitrate removal assessment. This chapter includes brief, surface-level discussion of the results, with a focus on how they relate to each other.

Chapter 4 builds upon initial discussion points and ties relevant results together in the context of the three research sub-questions. Each section corresponds to a sub-question and includes in-depth analysis of results and relevant limitations. Chapter 5 presents the project conclusions and answers the research sub-questions, as well as the overarching research question, while chapter 6 details recommendations for future research.

2
Methodology

2.1. Experimental Setup

The current project is a continuation of the research performed in 2023/24 by former MSc student Cahill [27]. Their work was primarily focused on the development of a Live Pole Drain (LPD, a type of nature-based slope drainage system) OAL, but they also created the preliminary structural designs and built the foundations for a bioswale OAL. This section will detail Cahill's contributions to the current project, as well the physical state and boundary conditions of the facility site at the beginning of the present research.

2.1.1. Existing setup and philosophy

Cahill worked on the development of the two setups at Flood Proof Holland (FPH), a site run by TU Delft's Green Village and VP Delta for outdoor experiments and demonstrations. FPH is located on the southern end of the TU Delft campus. The OALs were intended to "inform research, practice, and educational activities for eco-based sustainable water management" [27] and Cahill placed specific emphasis on the enablement of long-term performance monitoring while still supporting short-term experiments. This remains the overarching philosophy for the continued development of the site. At the end of Cahill's research, and thus at the beginning of the current project, the foundation and walls of the bioswale OAL facility had been constructed (Figure 2.1). Cahill also included a number of design recommendations in their work, which are discussed in the following section. In the course of the current project, the bioswale setup was named the **M**anaged **e**xperimental **S**ustainable **U**rban **D**rainage **a**rea, or MeSUDa.

Figure 2.1: MeSUDa facility at start of current project

2.1.2. Brief summary of Cahill's recommendations

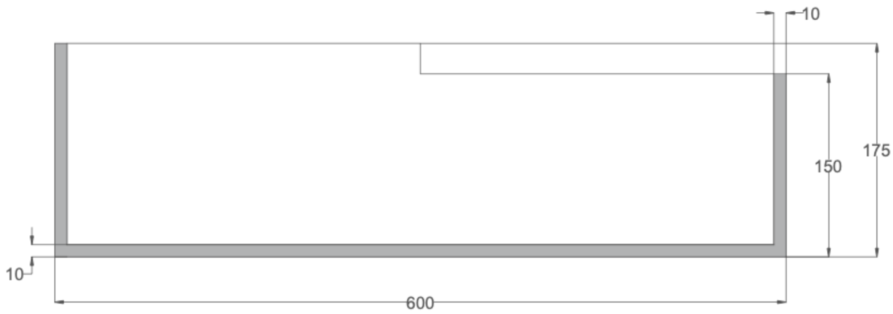
MeSUDa was designed as a gradual slope split into "three sections to monitor long-term vegetated swale hydrological and bio-chemical behaviour" [27]. Cahill's original draft design for MeSUDa can be seen in Appendix 7.1. The outer walls of the facility consisted of precast L-shaped concrete elements, forming a sort of container. As to the continued construction, Cahill recommended partitioning the container into three isolated sections using wooden dividers, lined with plastic to form an impermeable barrier for each section. The partitioning would allow for the conduction of parallel experiments with simultaneous long-term and short-term monitoring in the setup. The isolation of the setup was also recommended in order to prevent external interactions with the natural subsurface, specifically the local groundwater table. It would also allow for the measurement of infiltration fluxes. Thus, Cahill recommended the installation of controlled drainage locations where the outflow could be monitored as necessary.

2.1.3. Boundary conditions

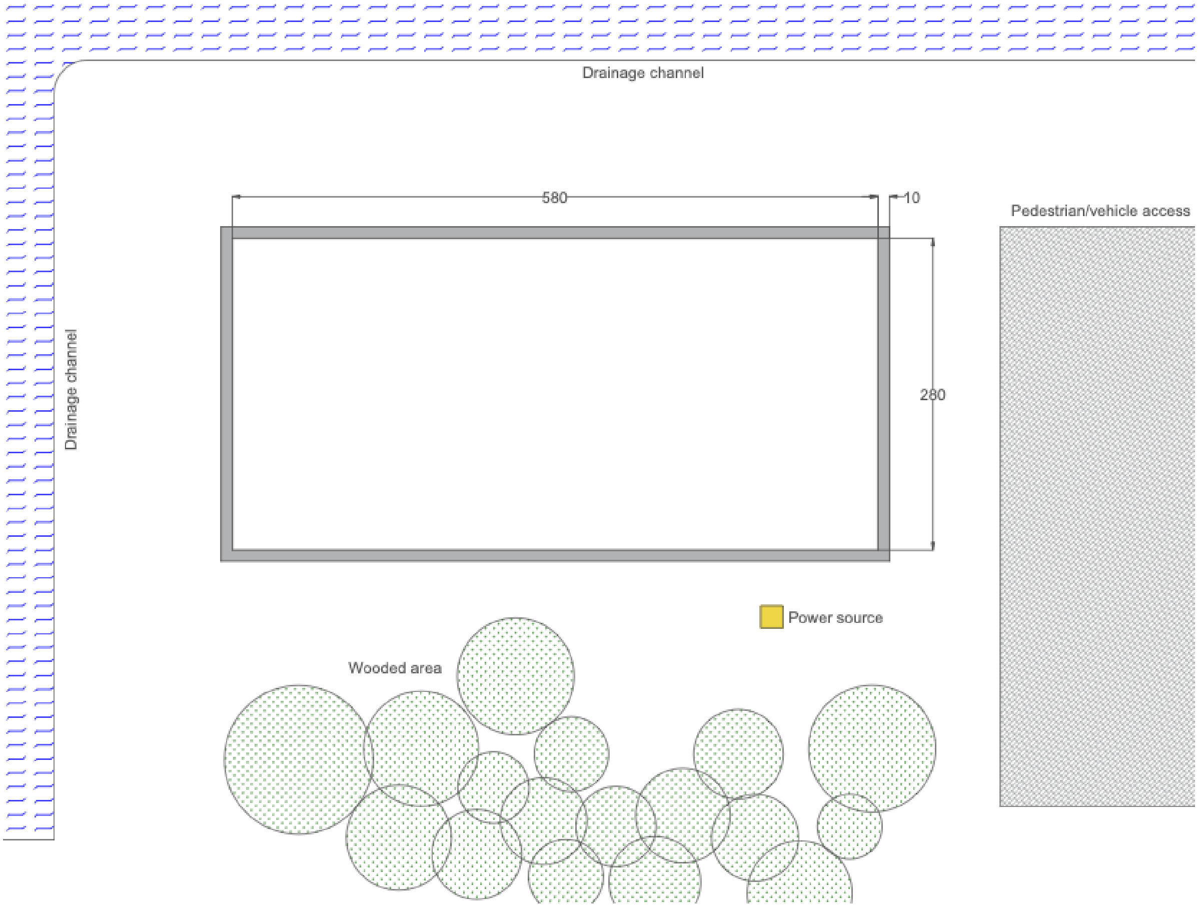
Spatial constraints and considerations

The physical dimensions of MeSUDa (Figure 2.2a) were determined by the pre-constructed concrete outer walls. Further installations were possible on all sides of the setup, but limited. MeSUDa is bordered by water-filled ditches on two sides, a wooded area on one side, and a vehicle/pedestrian access on the other (Figure 2.2b). Taking into account the need for users to move around and access the setup from all sides, approximately one meter of additional installation space was available along the perimeter of the structure.

Within the container itself, experimental space was limited to the structure volume (24.4 m^3). The division of the container into isolated sections was carefully considered to balance the number of parallel swale experiments with what was structurally feasible and scientifically representative. The selection of the fill media for each section was limited by material availability, particularly when accounting for existing materials at the site and what could be locally purchased. Additionally, the input of water for experiments was constrained based on available sources (discussed in the following section) and the space available to deliver and distribute the water across the swale sections. Any additional installations or equipment required for the water input would need to fit in the area next to the setup, or be mounted vertically above the structure.



(a) MeSUDa side view (all dimensions in cm).



(b) MeSUDa top view, with surroundings (all dimensions in cm).

Figure 2.2: MeSUDa boundary conditions.

Water availability and quality

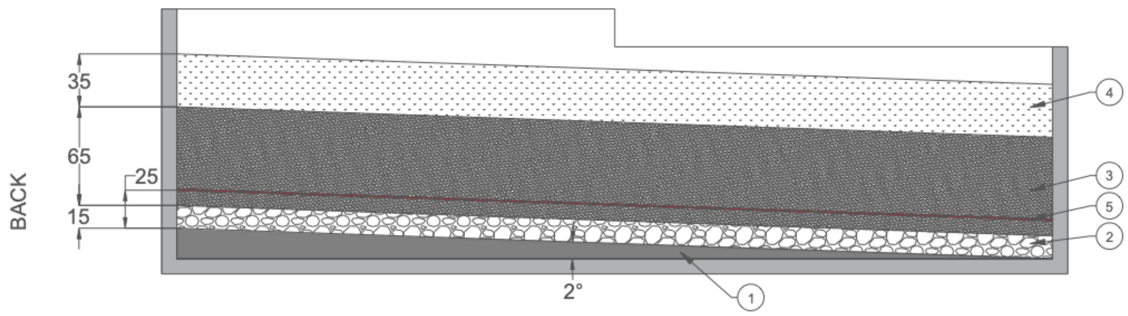
Three different sources were available for use as input water for MeSUDa experiments: rainwater, groundwater, and surface water from a nearby drainage channel. Since the present project aims to simulate the behaviour of a bioswale in an urban setting, the source needed to be of similar composition to urban runoff (specifically low salinity) and readily available for experiments. Rainwater was ruled out, as it would be too variable to use as a reliable source and would also require the construction of a harvesting and storage setup. Groundwater, accessed via a set of existing wells at FPH, was initially selected based on the assumption that it would be largely unpolluted and could be adjusted via chemical dosing to replicate urban runoff, if needed. However, a preliminary water quality check conducted in the field indicated that the groundwater was brackish rather than fresh, with an average electroconductivity (EC) of 7400 μS/cm. Since brackish water can negatively affect vegetation and soil health [28], [29], both critical factors in the functioning of a bioswale, groundwater was eliminated as an input source. As the only available remaining water source, the surface water from a nearby channel was selected for use in the present study. The primary concern in using the surface water was an anticipated high concentration of organic matter that would affect denitrification experiments. A chemical analysis of the water was therefore performed by the TU Delft WaterLab, consisting of ion chromatography (IC) and total organic carbon (TOC) testing. These results are discussed in section 3.1.1.

2.2. Design and construction

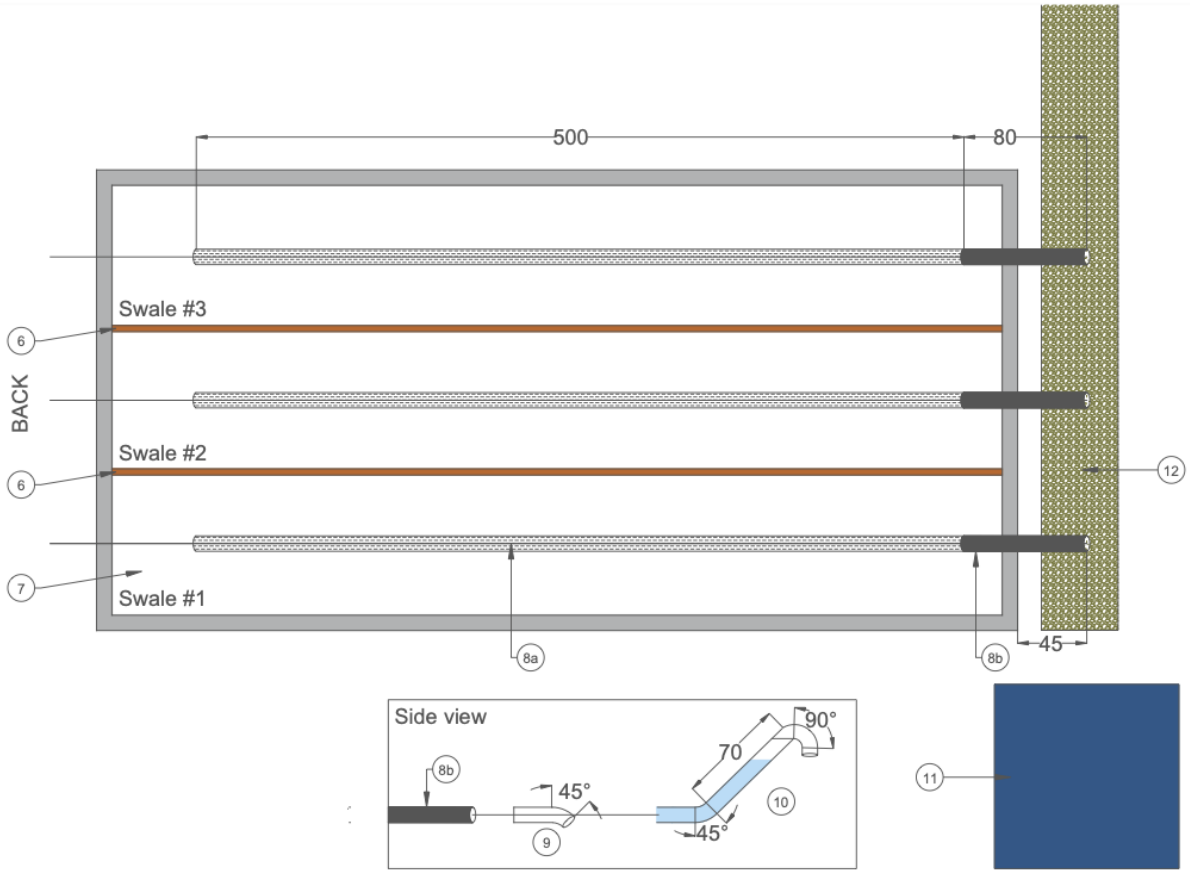
2.2.1. Design

A comprehensive overview of the implemented design is shown in Figure 2.3. Specific design features are denoted with numbers in the drawing and correspond to the information displayed in Table 2.1. MeSUDa was divided into three isolated sections: one benchmark swale (swale #2) and two experimental swales (swales #1 and #3). The base slope was set to 2% to facilitate the drainage of each swale. Since the slope layer was located underneath the impermeable lining and would not come into direct contact with drainage flow, it was formed using a heterogenous mixture of sand and rocky soil readily available at FPH. The fill media (Figure 2.4) were selected based on the work of Cahill [27], who also conducted a grain size analysis of similar materials (Table 2.2), as well as local availability. The topsoil was specifically chosen for its lower infiltration capacity to promote a longer residence time, and therefore an extended treatment period, in MeSUDa. The porosity of the media was not assessed in the current research and a composite value of 0.30 for the topsoil and fine sand was assumed based on analysis of similar materials performed by Cahill [27]. Woodchips were included in swale #3 as a supplementary organic carbon source intended to enhance nitrate removal. Woodchips were selected as a carbon source over other substrates (e.g., straw or mulch) because of their successful implementation in previous denitrification research [15], [16], [17], [18] combined with their availability on site at FPH.

A 1000 L water reservoir was placed next to MeSUDa to provide an easily accessible source of water for experiments. The drainage and outflow system consisted of a perforated PVC pipe placed at the bottom of each swale, emerging on the front side of the setup via a pre-drilled outlet. Two variations of outlet drains were included in the design, ”free-flowing” and ”enforced water level”, allowing for each swale to have either a fully drained setup or an anaerobic zone. Vegetation was selected for MeSUDa's swale, but was not seeded (and therefore not included in the project scope) due to time constraints. Recommended vegetation types and the reasoning for their selection can be found in Appendix 7.2.



(a) Side view (all dimensions in cm).



(b) Top view (all dimensions in cm).

Figure 2.3: MeSUDa design features.

Table 2.1: Design specifications for MeSUDa.

#	Feature Name	Specification	Description
1	base material	Heterogenous mixture of sand and rocky soil.	Available at FPH, used to construct 2% slope.
2	gravel + coarse sand	See Table 2.2 for details.	Available at FPH.
3	fine sand	Classified as sandy loam. See Table 2.2 for details.	Available at FPH.
4	topsoil	Tree planting soil (*bomengrond*). Classified as sandy clay loam. See Table 2.2 for details.	Selected for low infiltration capacity (approx. 0.25 cm/h), as tested in [27].
5	woodchips	Installed in swale #3 with thickness = 1 cm, depth = 90 cm.	Available at FPH. Calculated mass of approx. 16.5 kg. Assumed to be conifer species.
6	wooden divider	Wood planks, approx. 3 cm in thickness.	Available at FPH.
7	impermeable lining	LDPE 0.5 mm plastic sheeting.	Used to isolate each swale and prevent leakage.
8	underdrain	$\varnothing_{internal}$ 50 mm PVC, total length = 5.8 m. 5 m perforated (8a) and 0.8 m non-perforated (8b).	Placed at the bottom of each swale to collect and drain water at outlets.
9	outlet drain, free-flowing	$\varnothing_{internal}$ 50 mm PVC joint with 45° bend.	Directs outflow downwards to drainage ditch (12).
10	outlet drain, enforced water level	$\varnothing_{internal}$ 50 mm PVC. 45° joint connected to 70 cm of straight pipe with 90° joint at end.	Creates water table of 50 cm in the connected swale. Excess water is directed downwards.
11	water reservoir	1000 L (1m x 1m x 1m) plastic container.	Filled with surface water pumped from southern FPH canal. Located near MeSUDa for easy access.
12	drainage ditch	Dug-in ditch, width = 30 cm, depth = 25 cm.	Drains water from MeSUDa into nearby channel.

(a) Gravel (b) Coarse sand (c) Fine sand (d) Topsoil (e) Woodchips

Figure 2.4: Fill media.

Table 2.2: Grain size analysis findings from Cahill [27].

	Topsoil	Fine sand	Gravel
Bulk density [g/m^3]	0.96	1.81	1.60
Organic matter [%]	7	0.13	0.20
Clay [%]	15	<0.1	<0.1
Silt [%]	10	<0.1	<0.1

2.2.2. Construction

Physical construction of the setup began in late-March 2024 and was completed in mid-July 2024. The construction process began with the establishment of the 2% slope using the mixed base material (Figure 2.5a). Wooden barriers were installed in the structure, dividing it into three sections of equal size, each approximately 1 m wide. LDPE plastic lining was draped over the barriers and the outer walls of the structure, isolating each section (Figure 2.5). The lining was affixed to the barriers and walls and smoothed as much as possible, to prevent any folding that might result in preferential flow paths along the swale sides.

(a) Base layer with 2% slope. (b) Securing plastic lining over wood planks.

Figure 2.5: Construction of base layer and impermeable swale linings.

Underdrains were installed at the bottom of each swale, each inside of a filter stocking to prevent clogging (Figure 2.6a). The pipes were slid through the lining and the drilled outlets at the front of each section. A special plastic adhesive was used to seal the lining around each pipe to prevent leakage. Gravel was smoothed around the drainage pipes, with the coarse sand on top of it forming a slight angle inward towards the pipe to promote drainage. Following the drains and gravel/coarse sand, the fine sand was filled into the swales (Figure 2.6b). Between each material addition, the filled media was smoothed to make the individual layers as homogenous as possible.

(a) Installation of underdrains. **(b)** Gravel, coarse sand, and fine sand filled.

Figure 2.6: Underdrain installation and filled media.

The outlet of each swale was constructed in such a way that it would allow multiple outflow configurations and, therefore, some control over hydraulic behaviour inside of the setup. Initially, enforced water table configurations were applied to swales #2 and #3 to create anaerobic zones of 50 cm, while swale #1 was fitted with the free-flowing configuration. However, after leakage was observed in swale #3 (as further discussed in section 3.1.2), its outlet was changed to the free-flowing configuration.

(a) Outlets of underdrains. **(b)** Free-flowing configuration. **(c)** Enforced water table configuration.

Figure 2.7: MeSUDa outlet configurations.

The completed construction of MeSUDa is shown in Figure 2.8. Note: the installation of piezometers and other sensing hardware seen in the figure will be discussed in the next section.

Figure 2.8: Final construction.

2.3. Instrumentation

2.3.1. Instrumentation requirements and selections

A number of parameters were of interest for both current and future research, derived from relevant hydrological (infiltration and storage) and biological (denitrification) processes (Figure 2.9). The shown parameters were selected either as general indicators of water quantity and quality (water level, discharge, soil moisture, EC, TDS) or as indicators of denitrification activity (temperature, redox potential, dissolved oxygen, pH, nitrate concentration).

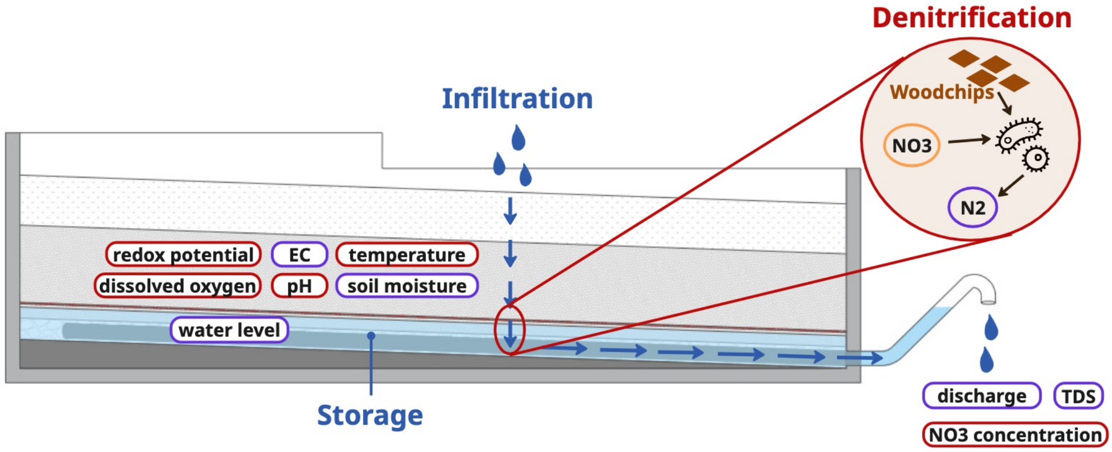


Figure 2.9: Relevant processes and parameters.

Sensor criteria and selections

When considering a sensor for purchase, in either the low-cost or industry standard categories, several criteria were used to assess its suitability (Table 2.3). Desired specifications for each parameter measurement were also taken into account, based on presumed experimental conditions inside of MeSUDa.

A full list of the considered and purchased sensors can be found in Appendix 7.3. Several of the industry standard sensors were purchased from the manufacturers METER and van Essen Instruments, as recommended by Cahill, who cited their prior implementation at other FPH setups [27]. Sensors from these companies, including the ECRN-100 tipping bucket and the TD-Diver had also been successfully used to monitor environmental conditions in Cahill's LPD setup. The remaining nitrogen and redox potential sensors were purchased at the recommendation of various faculty members at TU Delft. The low-cost sensors listed in the table were purchased from Seeed Studio, a company specialising in IoT (Internet of Things) hardware. These sensors were all readily available, and could be paired with Seeed's data loggers in a plug-and-play manner.

The current research involved the assessment of only some of the purchased sensors (Table 2.4). These were selected based on applicability to the planned experiments and ease of installation/implementation. Both industry standard and low-cost sensors were chosen so their performance could be compared.

Table 2.3: Sensor selection criteria.

Assessment criterion	Explanation
Suitability for application	The sensor: • measures desired parameter(s) • is suitable for in-field installation • has a low maintenance requirement
Cost	The sensor fits within the budget for its category. **Low cost:** up to 300 euros per sensor **Industry standard:** 300+ euros per sensor
Availability	Enough sensors are available for purchase to meet project needs, including the ability to purchase additional units at a later time, if needed/desired.
Requirement for additional purchase(s)	If additional components are required for sensor use (e.g., data loggers, cables), these components are also available for purchase and fit within the project budget.
Reviews/reliability	The sensor and/or selling company has positive reviews and appears to be reliable. The company has a designated support channel/structure in case of issues or questions.

Table 2.4: Sensors assessed in present research.

Sensor	Parameters	Category
METER ECRN-100	Discharge	Industry standard
METER TEROS 12	SM, T, EC	Industry standard
van Essen TD-Diver	Water level	Industry standard
Seeed EC/TDS	EC, TDS	Low-cost
Seeed SM/T/EC	SM, T, EC	Low-cost
Seeed water level	Water level	Low-cost

2.3.2. Instrumentation assessment

Sensors were evaluated for applicability to municipal bioswale monitoring based on a set of performance indicators (Table 2.5). These indicators encompassed quantitative data and qualitative observations gathered throughout the project. Sensors were scored for each indicator according to their performance in that category. Once assessed, the scores of the low-cost and industry standard sensors were compared, and personal recommendations for municipal implementation were made.

Table 2.5: Sensor performance indicators and scoring values.

Performance indicator	Assessment components
Cost	sensor price data logger price cables/accessories prices data transfer fees
Setup/installation	calibration/validation procedure ease of installation
Usability	adjustment of sensor settings ease of data retrieval availability of troubleshooting materials (manuals, online resources)
Reliability	data acquisition sensor accuracy software connectivity (data transfer) calibration/validation performance
Robustness	hardware longevity weatherproofing hardware ease of use
Scoring	**1** = not satisfactory, **3** = partially satisfactory, **5** = fully satisfactory

2.3.3. Calibration and validation procedures

Prior to installation in MeSUDa, most of the sensors were either calibrated or validated. The sensors were placed into one of four categories (Table 2.6), depending on their manufacturer calibration status and whether they could be calibrated in a lab or if in-field validation was required. The latter entailed temporarily installing the sensor at FPH and comparing measurements to those of other sensors or known values. The designated categories for each sensor can be found in Appendix 7.4.

Table 2.6: Calibration categories.

Category	Description
Manufacturer Calibrated	Sensors already properly calibrated according to manufacturer statement.
Lab Calibration	Sensors suitable for lab calibration using standard buffer solutions.
In-Field Validation with Industry Standard	Sensors with an industry standard counterpart in the ”Manufacturer Calibrated” category that can be used to directly validate performance.
In-Field Validation	Sensors with no industry standard counterpart, that require physical validation or validation with an external sensor.

The exact calibration and validation procedures for each sensor can be found in Appendix 7.5. For sensors whose performance was validated, the percent error was calculated to compare theoretical/-

expected and experimental findings. The general from of the percent error equation is shown below (Equation 2.1), where v_{exp} is the experimental value and v_{theo} is the theoretical value. In cases where performance was compared to that of an industry-standard counterpart, v_{theo} was the industry standard measurement.

$$\% \, error = \frac{v_{exp} - v_{theo}}{v_{theo}} \times 100\% \tag{2.1}$$

Seeed EC/TDS Calibration

Since the Seeed EC/TDS sensors had built-in EC calibration buttons, they were calibrated in the TU Delft WaterLab with standard buffer solutions of 1413 μS/cm and 12880 μS/cm. Due to time constraints and limited buffer solution availability, the EC/TDS sensors' calibration could not be validated in the lab. The calibration was assumed to have been successful, but performance was monitored closely throughout the experimental phase by consistently checking readings against other EC sensors.

METER ECRN-100 Tipping Bucket Validation

Although considered an industry standard sensor, the METER tipping buckets were still validated in the field. This validation was intended to test the sensitivity of the buckets' internal tipping spoons. According to METER, the rain gauges had a volume per tip of 4.02 ml, or 0.2 mm of rainfall. This volume was verified by adding small amounts of water to a tipping bucket with a syringe until the spoon was triggered, at which point the volume of added water was recorded. This was repeated 10 times for each tipping bucket.

Seeed SM/T/EC Validation

The Seeed soil moisture/temperature/EC (SM/T/EC) sensors were validated in the field against the (industry standard) METER TEROS 12 sensors. Soil moisture and EC were considered the most relevant parameters for the planned experiments and were therefore the focus of the validation. Temperature was left for future validation, should it be required. Testing the sensors entailed progressively saturating a bucket of soil and taking measurements at set intervals from a single METER TEROS 12 sensor and the Seeed sensors.

Seeed Water Level Validation

The Seeed water level sensors were validated in the field at FPH by measuring known water levels inside of a plastic pipe. The pipe was sealed at one end and fixed upright, with markings on the side denoting pre-measured heights.

2.3.4. Installation and setup

Piezometer installation

Prior to sensor placement, standpipe piezometers were installed at various locations in MeSUDa (Figure 2.10). These were required to house the van Essen divers and Seeed water level sensors. In order to maintain a flexible setup, the piezometers were placed at multiple points in each swale, such that sensors could easily be moved when desired.

Each piezometer (Figure 2.11a) was constructed from 40 mm diameter PVC tubing with approximately 60 cm of perforations cut along the bottom (Figure 2.11b). The base of each pipe was open, but covered with a filter stocking to prevent debris from entering and potentially clogging the pipe. Plastic caps were placed on top of the pipes to block rainwater inflow from above. The piezometers had a total length of 165 cm and were installed to a depth of 110 cm.

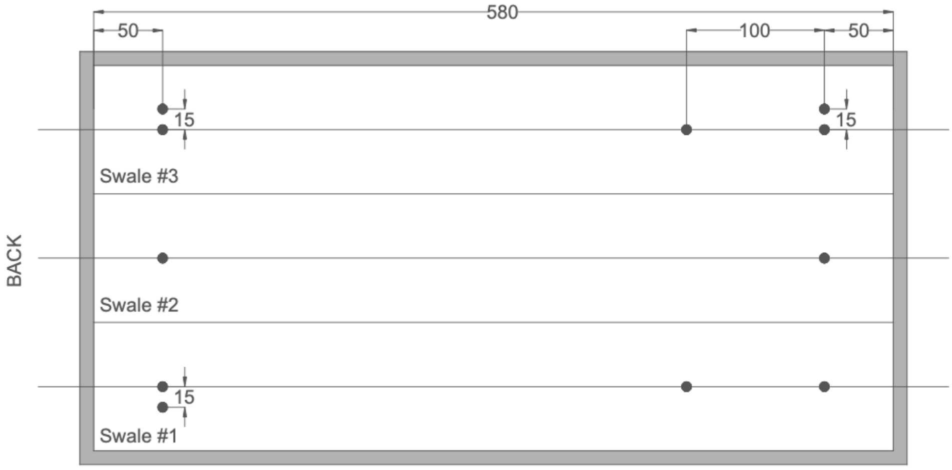


Figure 2.10: Piezometer install locations.

(a) Complete standpipe piezometer. **(b)** Perforations.

Figure 2.11: Standpipe piezometer.

Sensor installation and setup

Two different stages encompassed the process of instrumenting MeSUDa:

1. **Initiation period**: this period served as the familiarisation stage, focused particularly on how the sensors and their respective data loggers and data portals functioned. The instrumentation was set up and allowed several weeks of undisturbed monitoring. Data collected during this time was used to assess sensor performance and reliability in the field.
2. **Experimental period**: this stage was specific to the tracer testing conducted as part of the experimental phase of the project. Collected data would still serve to compare sensor performance, but the primary objective was to obtain the necessary measurements during the tracer tests.

The sensor initiation period occurred over one month while MeSUDa remained largely undisturbed. The following sensors were observed during the initiation period, with tracer-relevant sensors denoted by *:

- METER TEROS 12 *
- METER tipping buckets *
- Seeed SM/T/EC *
- Seeed EC/TDS *
- van Essen TD-Divers

Following the initiation period, instrumentation was relocated to perform experimental tracer testing. The resulting layout was intended to capture salt transport during the tracer test and to provide a comprehensive overview of flow dynamics in the soil of MeSUDa. Testing first took place in swale #1, followed by swale #3, and sensors were moved between the two swales as needed. Sensor locations will be further discussed in section 2.4.2, alongside the tracer testing setup and procedure.

2.4. Experiments: hydrological characterisation

2.4.1. Infiltration testing

A study of MeSUDa's fill media was performed via infiltration testing in the field (see Appendix 7.6 for procedure). These tests were intended to partially predict the setup's behaviour in future experiments, and to inform data interpretation later on. Infiltration testing was conducted using a METER Mini Disk Infiltrometer, a compact infiltrometer designed to measure unsaturated hydraulic conductivity (K) in field conditions. The goal of infiltration testing was to estimate K in the different layers in MeSUDa. As the Mini Disk Infiltrometer applies only a small suction (typically 2 cm), the resulting K estimates reflect only wet-range unsaturated hydraulic conductivity and do not capture saturated flow dynamics. These estimates could then be incorporated into the overall hydraulic characterisation of the setup. Two infiltration tests were performed on each material (topsoil, fine sand, and coarse sand): one under mostly dry conditions and one under partially saturated conditions. The material samples were not dried completely in an oven but were collected and dried naturally in an enclosed space. The fine and coarse sands were dried for six months, while a topsoil sample was dried for one month. When a partially saturated sample was tested, water was added to the dried material until a volumetric water content between 10-15$\%$ was reached.

In addition to testing MeSUDa's fill media individually, the infiltrometer was also used to assess the infiltration rate in the setup at different points in the project. The same location in swale #1 was selected for each infiltration test: 440 cm from the front wall, along the centre line of the swale. The standard infiltration procedure was followed.

2.4.2. Salt injection tracer testing

Salt injection tracer tests were conducted in swales #1 and #3, representing the standard bioswale case and a carbon-enhanced bioswale case, respectively. These tracers were intended to assess the hydrological behaviour of MeSUDa, including the existence of preferential flow paths, the infiltration rate (unsaturated hydraulic conductivity, K), residence time, and dispersion/advection phenomena. Each tracer test was performed as a ponded infiltration test, in which water was filled into an injection ring and infiltrated downward into the swale. The ponded method was selected to prevent overland flow and was intended to isolate transport dynamics in a specific section of the swale. The general stages of each test were as follows: clean water flush, pulse injection of salt water (high EC), clean water flush. EC was continuously monitored at several locations and results consisted of a breakthrough curve (BTC) at each monitoring location, where EC was plotted over time. The BTCs were used to assess the aforementioned hydrological characteristics in MeSUDa. The tracer procedure was iteratively developed, enabling familiarisation with MeSUDa's functioning and refinement of the experimental and data collection procedures. Iterative testing was performed only in swale #1, while the final tracer testing was performed in both swales #1 and #3.

Iterative development

The first set of tracer tests helped optimise the experimental setup and timing. The following improvements were made to the overall tracer procedure:

- full test duration extended to capture arrival and departure of the salt pulse (duration > 24 hours)
- flushing periods maximised (1000 L water input) to ensure steady-state EC levels before/after

salt injection, preventing interference of salt remobilisation

- sensing setup made completely autonomous, enabling remote monitoring of testing
- experimental setup made autonomous by installing peristaltic pump at injection ring (automatic water injection) and embedding a drainage pipe underneath the outlet (automatic outflow removal)
- second vertical profile of SM/T/EC sensors installed to capture any horizontal transport (intended to confirm that vertical flow dynamics dominate transport in MeSUDa)

Final setup and procedure

The injection ring (∅ 50 cm) was installed on the centre line of the swale and sensors were placed at four monitoring points: the injection site, two vertical profiles, and the outlet (Figure 2.12). The vertical profiles were referred to as "location #1" (30 cm from injection point) and "location #2" (80 cm from injection point). The water reservoir was placed near MeSUDa, with the peristaltic pumping connecting it to the injection ring. The pump was set to 7 rpm (measured as 26 L/h) for the entirety of the test to maintain a constant head of 20 cm inside of the injection ring. Each tracer test consisted of 37 hours of initial flushing (corresponding to maximum water input of 1000 L), 30 hours of salt injection, and 37 hours of final flushing. Salt injection was performed by mixing salt into the injection ring until an EC of 10000 μS/cm was reached. A detailed tracer test procedure, as well as relevant sensor settings, can be found in Appendix 7.6.

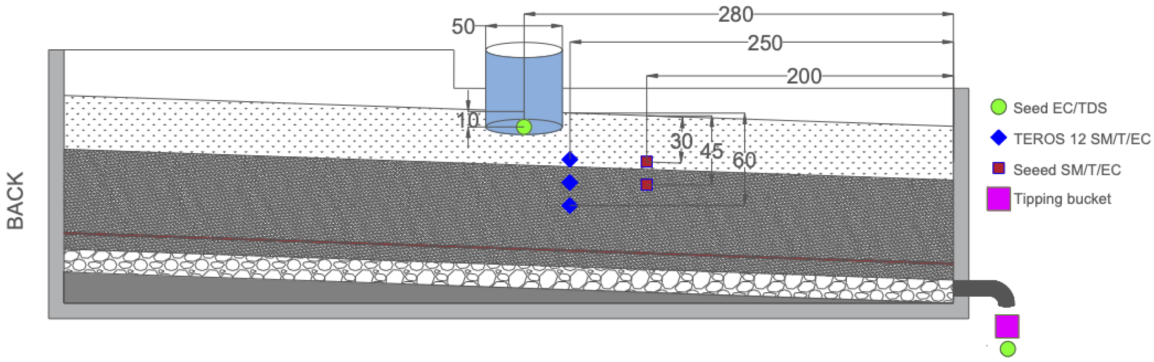


Figure 2.12: Tracer testing setup.

2.5. Modelling: nitrate removal assessment

MeSUDa's potential to remove nitrate from influent water was assessed with a focus on swale #3 (woodchip enhanced swale). The analysis investigated whether the presence of additional organic carbon could promote increased denitrification under the correct conditions. Denitrification could not be directly monitored in MeSUDa; instead, tracer testing results were combined with an analytical model to provide a theoretical estimate of nitrate removal in the swale. Additionally, woodchip decomposition in the setup was assessed, confirming the possibility of organic carbon enhanced denitrification. An overview of the nitrate removal assessment is shown in Figure 2.13.

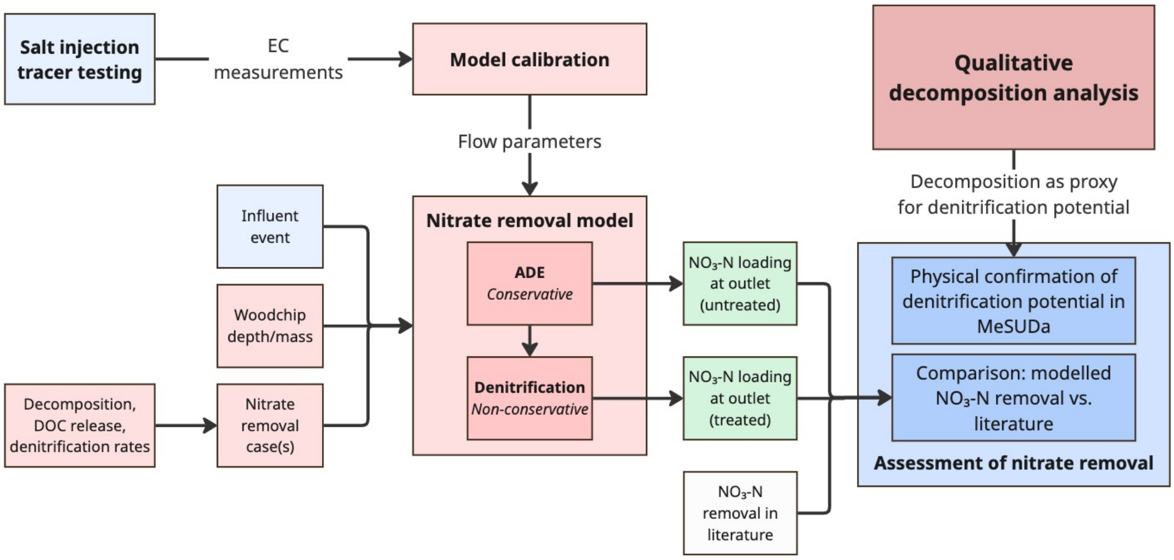


Figure 2.13: Overview of nitrate removal assessment.

2.5.1. Model structure

Analytical solution

Nitrate concentration was modelled analytically, combining conservative transport with non-conservative denitrification. Conservative transport in MeSUDa was simulated using the 1-D Advection Dispersion Equation (1D ADE, Equation 2.2), derived to represent an impulse input of a conservative solute (derivation in Equation 2.3) [30].

$$\frac{\partial C}{\partial t} = -v\frac{\partial C}{\partial x} + D\frac{\partial^2 C}{\partial x^2} \tag{2.2}$$

$$C_{ADE}(x,t) = \frac{M}{\sqrt{4\pi Dt}}e^{-\frac{(x-vt)^2}{4Dt}} \tag{2.3}$$

This equation was suitable for use under the assumptions that flow in the setup was one dimensional in the vertical direction and that salt input during tracer testing was a conservative impulse input. Following denitrification, the nitrate concentration $C(x,t)$ was calculated by convolving influent data $C_{influent}$ with the ADE impulse equation and subtracting a denitrification term k:

$$C(x,t) = \int_0^t C_{influent}(\tau)\cdot C_{ADE}(x,t-\tau)\,d\tau - k(x,t) \tag{2.4}$$

where:

- $C(x,t)$ was the concentration at depth x and time t (also known as the BTC),
- $C_{influent}(\tau)$ was the influent concentration at past time τ,
- $C_{ADE}(x,t-\tau)$ was the impulse response of the ADE,
- τ was the integration variable representing past injection time,
- $k(x,t)$ was the nitrate removed in the woodchip layer.

The following inputs were required for the nitrate removal model:

- **Depth of interest** (x): vertical distance from influent injection point, between 0 cm (inlet) and 100 cm (outlet drain)
- **Time of interest** (t): amount of time following influent injection
- **Flow parameters**: values representing flow dynamics in MeSUDa
  - *v*: flow velocity
  - *D*: dispersion coefficient
  - *M*: mass coefficient
- **Influent data**: concentration ($C_{influent}$) and duration of the influent pulse
- **Nitrate removal capacity** ($k(x,t)$): determined by nitrate removal rate (r_{NO_3}), woodchip layer mass and depth

Model overview

The nitrate removal model (Figure 2.14) consisted of two stages: a conservative stage (transport governed by the 1D ADE) and a non-conservative stage (applied denitrification). Influent concentration data was convolved with the corresponding impulse response (integrated term in Equation 2.4) . The result was used to calculate the BTC, or the $NO_3 - N$ concentration at the depth of interest for each relevant time step. Based on the BTC, the mean residence time (MRT) could be calculated for any depth, or in a specific layer. Nitrate removal was represented by three processes: woodchip decomposition, dissolved organic carbon (DOC) release, and eventual denitrification. Denitrification was simulated in the woodchip layer by applying a given nitrate removal rate to the inflow of contaminated water, yielding a nitrate removal capacity ($k(x,t)$) at every time step. Finally, effluent nitrate loading was calculated by subtracting the removal capacity from the untreated BTC at the outlet. Model outputs included the nitrate BTC (mg/L), cumulative mass loading (mg), and mass flux rate (mg/h) at the swale outlet.

The nitrate removal model simulated only conservative transport, with the 1D ADE valid up to the woodchip layer, where denitrification induced non-conservative behaviour (Figure 2.15). Although the model could not simulate further vertical transport after nitrate removal, the woodchip nitrate removal capacity was applied at the outlet to account for additional dispersion between the woodchips and discharge point.

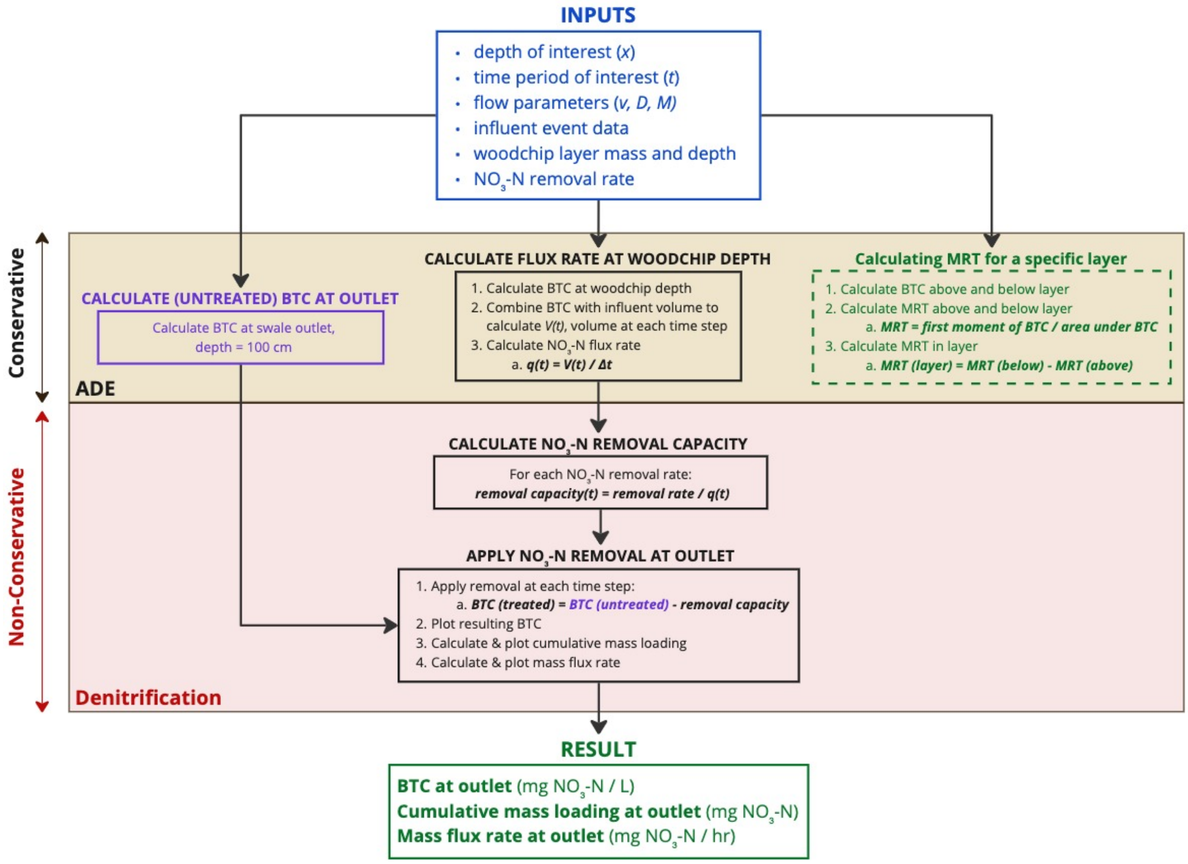


Figure 2.14: Nitrate removal model flow diagram.

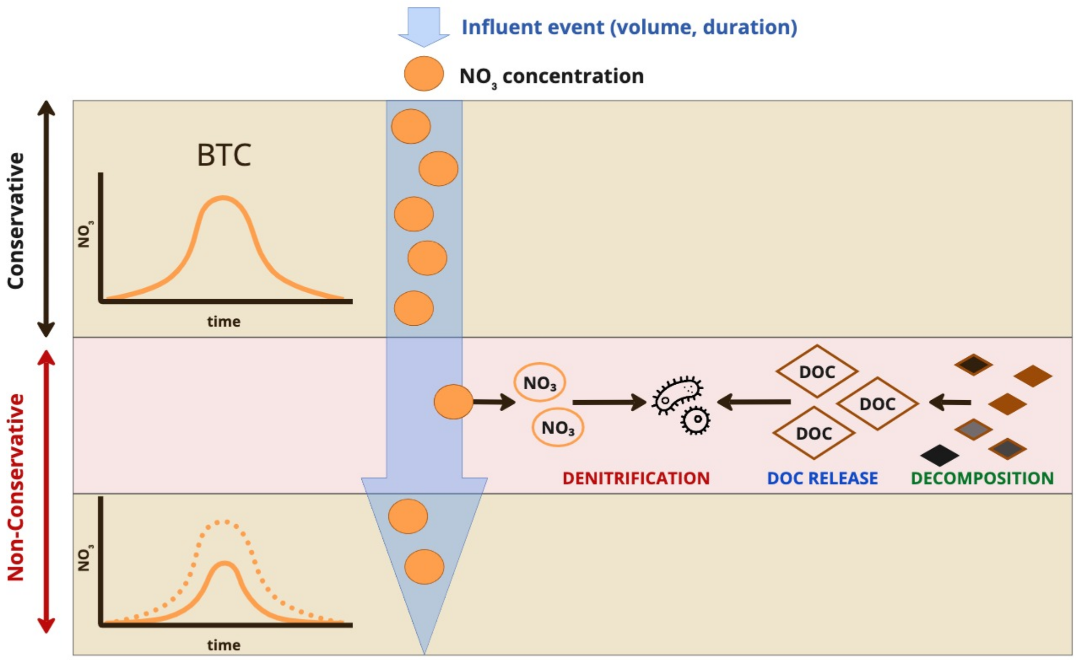


Figure 2.15: Nitrate removal in woodchip layer (red), as modelled.

Model assumptions
Within the woodchip layer (Figure 2.15), both aerobic and anaerobic conditions were assumed, as required for decomposition/DOC release and denitrification, respectively. All three processes occurred simultaneously and instantaneously as water passed through the layer, and the rates of each process remained constant over time. Lastly, it was assumed that flow conditions and nitrate concentrations at depth = 100 cm were equal to those at the swale outlet (i.e., the vertical transport through the 15 cm gravel layer and the lateral movement through the underdrain were negligible).

Model calibration and validation
The parameters *v*, *D*, and *M* were estimated through non-linear least squares fitting using experimental tracer data and results characterised transport in MeSUDa. Experimental data included EC measurements from the injection ring (influent data) and the first vertical profile (salt BTC), located 30 cm downstream of the injection point. Parameter calibration was performed at a depth of 45 cm, selected because it was located in the fine sand layer of the swale which made up most of the depth profile and was expected to exert the most influence over the setup's flow dynamics. Beginning with initial guesses for *v*, *D*, and *M*, a BTC was generated at 45 cm based on the influent tracer data and fitted to the salt BTC. The three parameters were updated until the best fit between the modelled BTC and the experimental BTC was achieved. To account for a sensor placement error of $\pm$3 cm during tracer testing, the fitting process was repeated for depths of 42 cm and 48 cm. Model calibration produced three parameter sets corresponding to: overestimated placement (42 cm), correct placement (45 cm), and underestimated placement (48 cm). The correct placement set was used to capture MeSUDa's transport dynamics while the other two sets represented the error region.

Following calibration, the model was validated against the tracer data at depth 30 cm using all three parameter sets. BTCs were modelled and compared to the salt BTC using the root mean square error (RMSE) and the Nash-Sutcliffe Efficiency (NSE). These findings were used to assess the model's ability to simulate conservative transport with minimal error.

2.5.2. Modelled scenarios
Tracer test MRT validation
Prior to simulating nitrate removal in MeSUDa, the 1D ADE stage of the model was used to validate the residence times calculated from tracer testing results. Influent data consisted of the EC measurements at the injection ring and the woodchip layer depth was set to 90 cm with a mass of 16.5 kg (actual instalment in MeSUDa) . The MRT was calculated for the following areas and depths: topsoil layer (0 - 35 cm), fine sand layer (35 - 100 cm) , woodchip layer (90 cm), and MeSUDa (whole, 0 - 100 cm).

Nitrate removal test cases
The dynamics responsible for nitrogen removal were simplified to three primary processes: woodchip decomposition, DOC release, and denitrification. Various literature sources were consulted to obtain an occurrence rate for each of these processes. Minimum and maximum rates were considered for both DOC release and denitrification in order to capture the possible upper and lower limits of nitrate removal in MeSUDa, as well as account for uncertainty in all three process rates. The interdependent rates of all three stages were combined to create four nitrate removal test cases (Figure 2.16).

Studies quantifying the decay rate of buried woodchips were used as a proxy to characterise the decomposition process inside of MeSUDa. Moorman et al. [31] observed the decomposition of woodchips for nine years, finding that half of the woodchips had decayed after 4.6 years. Combining this result with the mass of woodchips installed in MeSUDa (16.5 kg), a decomposition rate ($r_{decomposition}$) of 4.9 g/day was calculated using Equation 2.5, in which m_{wood} was the mass of woodchips and $t_{50\%}$ was

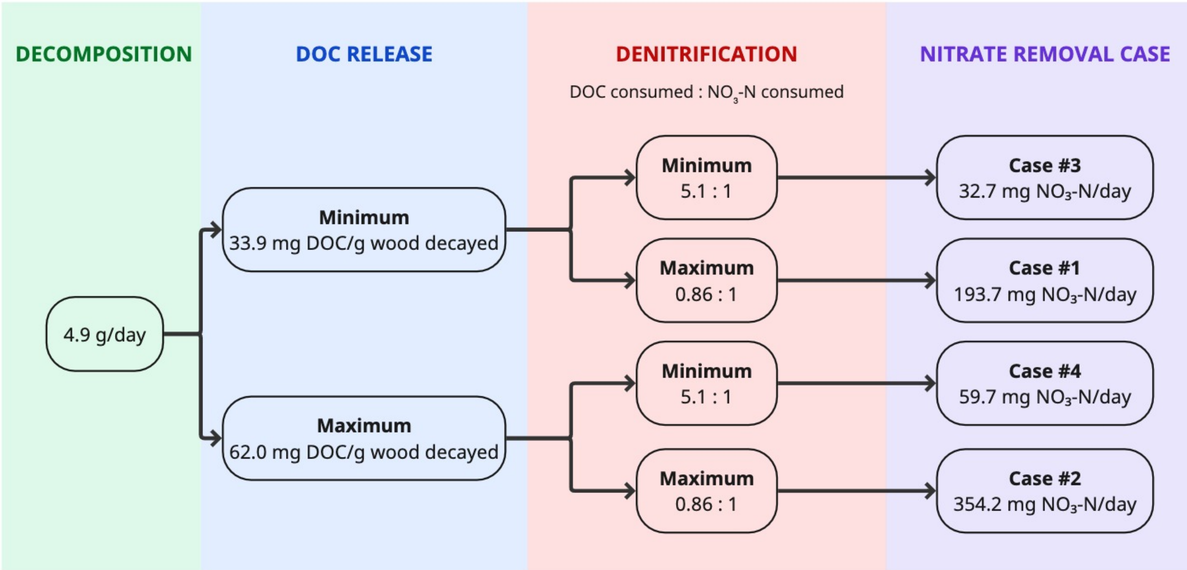


Figure 2.16: Derivation of nitrate removal cases.

equal to 4.6 years.

$$r_{decomposition} = \frac{m_{wood}}{2 \times t_{50\%}} \tag{2.5}$$

Two release rates ($DOC_{release}$) were adopted from Hollands et al. [32], who researched DOC release from forest floor deadwood: a minimum rate, representative of a managed forest site, and a maximum rate, representative of an unmanaged site. Both values were multiplied by the decomposition rate for MeSUDa (Equation 2.6), yielding the daily DOC release (r_{DOC}) for the setup .

$$r_{DOC} = DOC_{release} \times r_{decomposition} \tag{2.6}$$

A simplified relationship between DOC release and nitrate consumption was adopted to predict denitrification rates. Two denitrification scenarios, representing minimum and maximum denitrification efficiencies, in the form of $DOC_{consumed} : NO_3 - N_{consumed}$ (represented by R_{denit}) were sourced from Qin et al. [33]. These scenarios were selected to encompass the possible upper and lower limits of nitrate removal in MeSUDa. The actual ratio of DOC consumption to nitrate consumption was expected to fall between the presented values. The nitrate removal rate, r_{NO_3}, for each removal case was calculated with Equation 2.7.

$$r_{NO_3} = R_{denit} \times r_{DOC} \tag{2.7}$$

The influent event for testing nitrate removal was selected from data collected by Rombeek et al. [34]. This event (50 mm rainfall in six hours) corresponds to an above average, high-intensity rainfall occurring during the summer in the Netherlands. Runoff from a larger area, as usually captured by full-scale bioswales, was not considered and only the influent over the actual swale area was taken into account. Influent data and other model inputs are summarised in Table 2.7. Each test case was simulated with a woodchip depth of 90 cm, considered close enough to the outlet that further non-conservative transport was negligible. The potential ability of MeSUDa to remove nitrate was assessed by comparing the effluent $NO_3 - N$ loading for the different test cases to an untreated case, in which no nitrate was removed. Focus was placed on case #3, as it was considered most representative of the actual swale performance and reduced the risk of overestimating removal capability. The findings from case #3 were also compared to those from literature on nitrate removal in full-scale, active bioswales.

Table 2.7: Input parameters for nitrate removal quantification in MeSUDa.

Input parameter	Value	Reasoning
Woodchip mass	16.5 kg	Mass installed in MeSUDa
Woodchip depth	90 cm	Depth of layer in MeSUDa
Influent event	50 mm in 6 hours (rainfall)	Above average summer rainfall intensity in the Netherlands [34]
Influent $NO_3 - N$ concentration	25 mg/L	Concentration of polluted surface water [15], [16]
$NO_3 - N$ removal cases tested	All (#1, #2, #3, #4)	Comparison of removal rate performances

Variation in woodchip mass and depth

Different woodchip masses and depths were tested to inform the optimisation of nitrate removal in MeSUDa. The same influent data was used as in the modelling of the four removal test cases. Only case #3 was evaluated for different woodchip configurations. A summary of the input parameters is shown in Table 2.8. Since this analysis concerned woodchip depths of less than 90 cm which would require simulation of non-conservative transport until the underdrain, results were limited to effects on MRT and the nitrate removal capacity within the woodchip layer. Based on these findings, the potential resultant nitrate loading at the outlet was qualitatively assessed and recommendations regarding the optimal amount and placement of the woodchips were made. Additionally, the functional time during which the woodchip layer could maintain case #3's nitrate removal percentage was calculated for the different masses, as was the time required for complete wood decay. This part of the analysis considered an average yearly rainfall of 851 mm/year in the Netherlands [34]. Functional time results were also used to inform woodchip amount recommendations.

Table 2.8: Input parameters for testing of woodchip masses/depths in MeSUDa.

Input parameter	Value	Reasoning
Woodchip mass	5, 10, 15, 16.5, 20, 25, 30 kg	Reasonable range of masses
Woodchip depth	20, 40, 60, 80, 90 cm	Reasonable range of depths
Influent event	50 mm in 6 hours (rainfall)	Above average summer rainfall intensity in the Netherlands [34]
Influent $NO_3 - N$ concentration	25 mg/L	Concentration corresponding to polluted surface water [15], [16]
$NO_3 - N$ removal cases tested	Case #3 (min DOC release, min denitrification efficiency)	Case #3 assumed to be closest to actual removal rate in MeSUDa

2.5.3. Qualitative decomposition analysis

Woodchip decomposition in MeSUDa served as a proxy for denitrification potential. The qualitative decomposition analysis assessed whether the woodchips could successfully break down in the swale, thereby releasing additional DOC and initiating the denitrification process. A sample of woodchips was extracted from swale #3 approximately 230 days after installation (Figure 2.17a). Two additional samples were collected from a bucket of woodchips that had been stored indoors in a dry environment for the same amount of time: one sample from the top (Figure 2.17b) and one from the bottom (Figure 2.17c). These served as a reference to assess decomposition of the swale woodchips.

(a) Sample from swale #3. **(b)** Sample from bucket top. **(c)** Sample from bucket bottom.

Figure 2.17: Woodchip samples.

Three tests were used to evaluate the decomposition extent of the samples: two physical-based and one visual-based (Figure 2.9. Detailed information in Appendix 7.7). General observations of extraction conditions and woodchip appearance were also noted. For each test, multiple woodchips per sample were assessed and the results were compiled to assign an overall decay status to the sample (”no decay”, ”minimal decay”, or ”moderate decay”). The DC3 classification focused on the solidity of the woodchip samples as an indicator of decay. Each sample was assigned a classification, with intermediate classifications (i.e., 1.5 or 2.5) given when appropriate [35]. The physical toughness of the woodchips was evaluated using the “pick test”, as described by Anderson et al. [36]. The pick test was designed to detect surface level wood decay. A brief guide on aerobic decay and associated fungi from Schweingruber et al. [37] informed the visual fungal identification. Woodchips from each sample were analysed and any defining features were recorded, along with the suspected responsible fungi species.

Table 2.9: Decomposition tests performed on each sample.

Type	Name	Description	Source
Physical	DC3 Classification	Samples assigned values between 1-3 based on solidity. 1: wood is sound and solid, 2: wood is slight soft with hard inner core, 3: wood is soft and easily crushed.	Eaton and Sanchez [35]
Physical	Pick Test	Wood pierced with sharp tool and assessed based on failure mode, break difficulty, and break sound.	Anderson et al. [36]
Visual	Fungal Identification	Evaluation of wood color variation, patterns, and texture. Identification of brown rot fungi, soft rot fungi, and white rot fungi.	Schweingruber et al. [37]

3
Results

3.1. MeSUDa setup and function

3.1.1. Input water quality

The results of the WaterLab's IC analysis (Appendix 7.8) confirmed that the use of the FPH surface water was appropriate for the current study, as all of the tested anions and cations were within, and in some cases below, the expected ranges for the setting of the channel. The surface water was fresh, with an EC of 980 μS/cm compared to the brackish groundwater EC of 7400 μS/cm. The nitrate concentration (0.85 mg/L, or 0.19 mg/L as $NO_3 - N$) was much lower than anticipated. This could present a challenge in performing any future quantitative denitrification experiments in the carbon-enhanced swale, as removal of such small concentrations could be difficult to observe. The TOC results confirmed that the surface water contained relatively high levels of organic matter, with an average concentration of 22.5 mg/L. This finding was expected for surface water, but prompted concern that the input water would not be representative of urban runoff (see section 4.1.1 for further discussion on TOC levels in runoff). For the present project, the surface water was deemed to be the most suitable source for simulating flow into a bioswale. Its high organic content was considered less detrimental to the overall health of the setup than the brackish groundwater and it was easily accessible at all times. The aforementioned concerns did not impact the experiments conducted in MeSUDa. However, recommendations will be made in chapter 6 as to how future studies might better replicate urban runoff, particularly in the scope of nitrate removal quantification.

3.1.2. Leakage in base structure

Following MeSUDa's construction, a leak in the setup's structure was detected. Significant water loss was first identified when attempting to establish a water table of 50 cm in swale #3 using the enforced water table outlet configuration. Despite a water input of over 3000 L, no water accumulation was observed. Given the estimated porosity of 0.30, this input should have created a water table of at least 20 cm. There were no outward signs of leakage along the exterior of MeSUDa, nor were the outlet pipes leaking. It was concluded that the impermeable plastic lining near the front of the swale had been breached and that water was flowing through the gap between the L-shaped frame and down into the ground.

3.2. Instrumentation

3.2.1. Validation

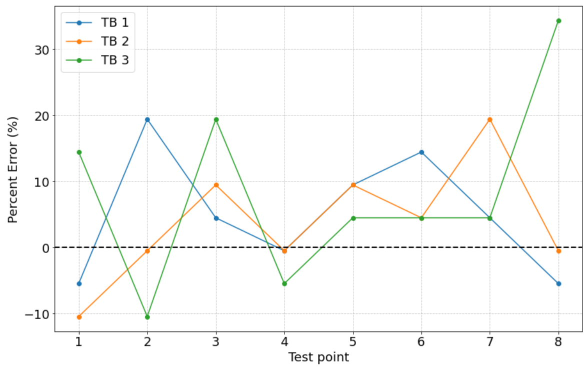
The validated sensors varied in performance, as evidenced by the percent error results (Table 3.1 and Figure 3.1). The METER tipping buckets yielded a larger average error than expected for industry standard devices, which is suspected to be a result of the validation method. The tipping buckets were validated using incremental small volumes of water ($\sim$1 mL), injected into the funnel using a syringe until one of the internal spoons tipped. The manufacturer accuracy of the buckets is unknown, but it is possible that that they were unable to correctly register such small changes in volume. Figure 3.1a does not exhibit similar error values for all three tipping buckets, but each bucket does demonstrate a slight oscillation between subsequent error values. This behaviour may be an indication that the small volumes of added water occasionally got stuck on the funnel only to contribute to the following spoon tip, resulting in an over- then underestimation of water input.

In the low-cost sensor category, the Seeed SM/T/EC sensors also exhibited significant average errors, particularly in measuring EC. Validated against the METER TEROS 12 sensors, their resulting accuracy may have been affected by placement differences in varying soil conditions. This is supported by the similar percent error observed between the two Seeed sensors at all but the first test point, for both soil moisture and EC readings (see Figure 3.1b). These patterns indicate that both Seeed sensors may have been placed in one soil condition, while the METER sensor was inadvertently placed in another. The METER sensors have their own error, which was not accounted for in the current analysis. The assumption that the METER readings represent the "true" values in the validation may have led to an overestimation of the Seeed error, particularly when compared to their stated manufacturer accuracy of $\pm$ 5% for soil moisture and $\pm$ 3% for EC.

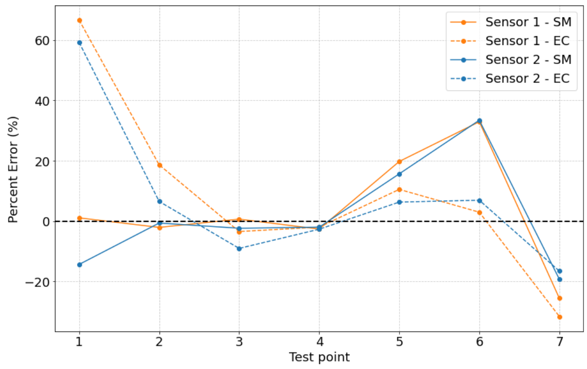
The Seeed water level sensors produced low average errors. Compared to the Seeed SM/T/EC sensors, the lower error could be attributed to the water level sensor's more straightforward function as well as the stability of the measurement conditions. While the measurement of soil moisture and EC is more dependent on soil properties and occurs in a heterogenous medium, the use of a piezoresistive sensor to measure water levels depends only on a single medium (water) with stable properties. It is therefore expected that the low-cost water level sensor might perform relatively better than its SM/T/EC counterpart. The water level sensors also routinely underestimated the water depth in all but two measurements (Figure 3.1c), suggesting some degree of calibration error. Since manual re-calibration of the sensors was not possible, they would either need to be re-calibrated by Seeed or the error would need to be accounted for during future use.

Table 3.1: Percent error ranges for validated sensors.

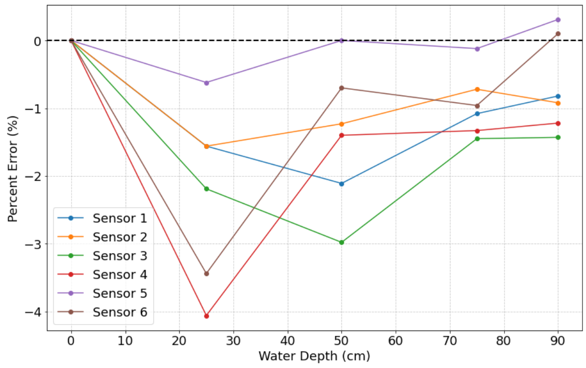
Sensor	Percent error (mean $\pm$ SD)	Manufacturer accuracy
METER tipping buckets (x 3)	9.00 $\pm$ 2.82	$\pm$ 2%
Seeed SM/T/EC sensors (x 2)	**SM**: 12.32 $\pm$ 0.30, **EC**: 17.37 $\pm$ 2.88	**SM**: $\pm$ 5%, **EC**: $\pm$ 3%
Seeed water level sensors (x 6)	1.08 $\pm$ 0.52	$\pm$ 3%



(a) METER tipping bucket validation.



(b) Seeed SM/T/EC sensor validation.



(c) Seeed water level sensor validation.

Figure 3.1: Sensor validation results.

3.2.2. Experimental observations

Initiation period

The industry standard sensors, including the METER tipping buckets, TEROS 12, and the van Essen divers, did not present any issues during the initiation period. In the case of the METER sensors, the connectivity was continuous, and the online data portal (ZENTRA) was simple to set up. This portal proved beneficial, as it enabled not only remote monitoring of the measurements, but also of the equipment itself (sensors and data logger). With respect to the hardware, the data logger, as well as all sensors and cables remained intact and weatherproof throughout the entire period. The van Essen divers also performed well and their initialisation and installation inside the piezometers was straightforward. However, the divers could not transmit data wirelessly and no remote monitoring of the measurements or the divers themselves was possible. Compared to the METER equipment, the diver data retrieval was timelier and more inconvenient, as the water levels in the setup could not be tracked in real time. Any alterations to sensor settings also required sensor removal and the suspension of the current sampling period.

The low-cost sensors experienced several complications during the initiation period. Although installation of the Seeed equipment was straightforward, not all sensors were successfully connected to the data logger. Multiple cable splitters were used to connect over 12 sensors to a single logger, which was theoretically rated to support them. In practice, the logger was unable to accommodate the sensor load, resulting in several sensors not being recognised or displayed on the online data portal (SenseCAP). The Seed equipment experienced further connectivity issues: for the first nine days, the Seeed data logger did not transmit any data. The reason for this interruption was unknown but was deemed an issue with the logger because the data of *all* connected sensors was lost. After nine days, the logger resumed normal function and no further connection problems were observed. In addition to software challenges, the Seeed equipment also experienced hardware complications. Despite a smooth initial installation, the data logger's sensor ports began to loosen over time before eventually detaching completely from the logger casing. The connections between sensors and the logger remained stable, but the logger itself was no longer sealed and weatherproof. It was also increasingly difficult to attach and remove the sensor cables.

Experimental period

During the experimental period, tracer testing was conducted with the following sensors: METER tipping buckets, METER TEROS 12, Seeed SM/T/EC, and Seeed EC/TDS. As during the initiation period, the METER sensors functioned well during the tracer tests and experienced no issues. Moreover, the ZENTRA data portal proved extremely useful during testing. The site provided a comprehensive overview of live and previously collected data, which was needed to plan and assess the salt-injection and flushing phases.

The Seeed sensors functioned suitably during tracer testing, but still exhibited some of the same issues as during the initiation period. First, the data logger ports continued to detach from the logger casing despite several attempts to tighten and secure them. Although the ports were on the underside of the logger, there was concern that rain or moisture would enter the casing and disrupt data transmission during testing. Second, even though a second data logger had been installed, the theoretical maximum number of sensors were still not recognised simultaneously. Fortunately, only the SM/T/EC and EC/TDS sensors were required during tracer testing and these were successfully connected. With respect to the online data portal (SenseCAP), the Seeed equipment worked well. Although this portal was not as customisable or simple to use as its METER counterpart, it still provided the necessary remote monitoring and overview of sensor function.

3.2.3. Performance assessment

Scores for each performance indicator varied significantly between sensors (Figure 3.2, see Appendix 7.9 for complete assessment and resulting scores). The industry standard sensors, shown in magenta, scored exceptionally well in the "Robustness" category, in which the low-cost sensors did poorly. This distinction is the result of continued malfunctions with the Seeed data logger sensor ports, as observed during the initiation and experimental periods. Although these malfunctions did not result in sensor failure during the present research, the potential exposure of internal electronics to the elements during field-use was considered critical. The industry standard sensors did not exhibit such issues. The categories of "Usability" and "Reliability" also highlighted differences between the two sensor groups, with the low-cost sensors routinely scoring lower than the industry standard sensors. The lower "Usability" scores were the result of a lack of troubleshooting resources, which often meant that the manufacturer had to be contacted directly for assistance. This was not the case for the industry standard group, particularly devices from METER. For the "Reliability" category, two of the low-cost sensors received deductions due to the connectivity issues experienced during the initiation period. The industry standard sensors did not exhibit any connectivity problems, but the METER tipping buckets did perform poorly during validation and therefore also received a lower reliability score. Conversely, all low-cost sensors outperformed the industry standard sensors in "Cost", as expected. The low-cost sensors were significantly cheaper than the industry standard sensors, despite requiring some additional accessories (e.g., cable splitters) and the purchase of SIM cards to transfer data. Finally, all tested sensors scored equally well in the "Setup" category, as they were all simple to install and were either already calibrated or could be easily calibrated/validated during the project.

Two sets of sensors enabled direct comparison between the industry standard and low-cost groups.

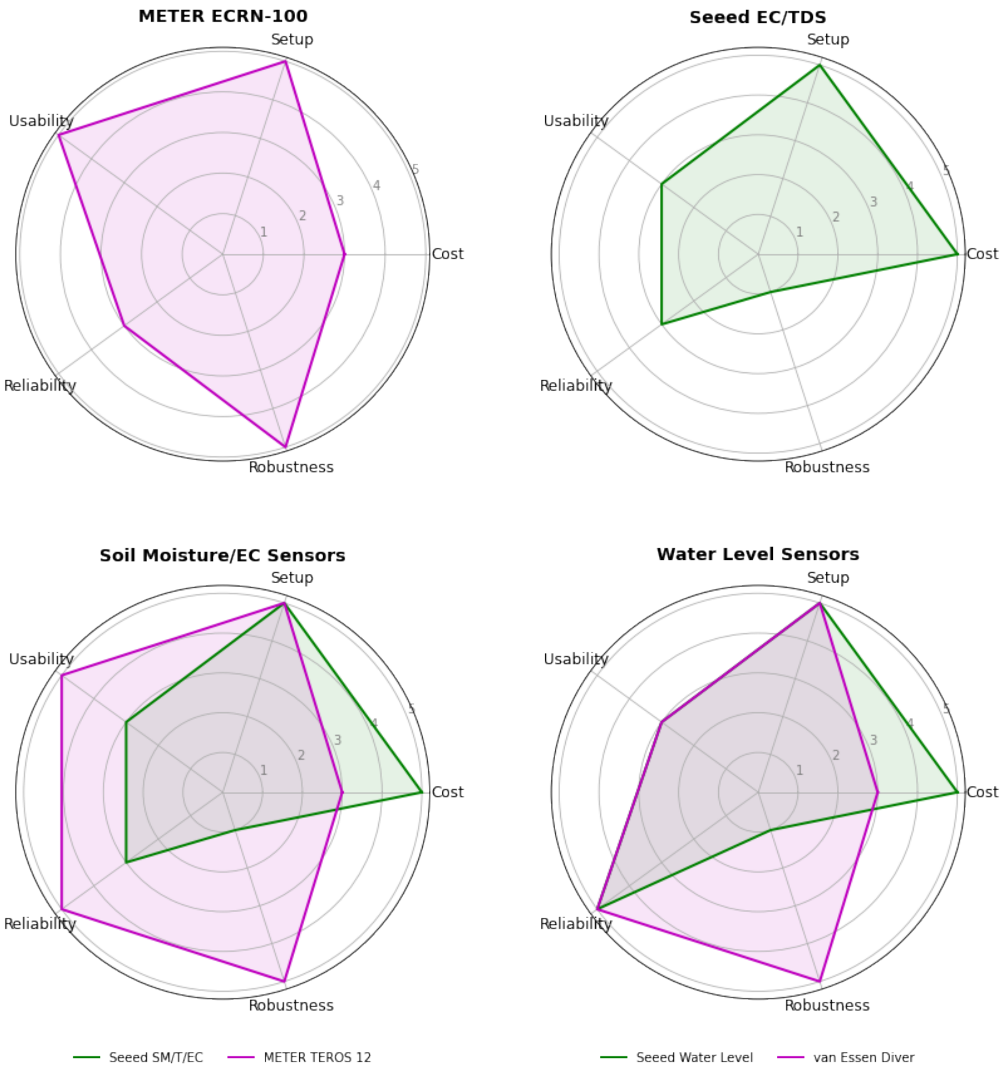


Figure 3.2: Summary of sensor performance results.

The Seeed SM/T/EC sensor was compared to the METER TEROS 12 for the measurement of soil moisture and EC. The TEROS 12 outperformed the Seeed sensor in three out of the five categories, only receiving a lower score for "Cost". Based purely on the present analysis, the METER TEROS 12 is the most suitable sensor for applications in which cost is not the primary selection factor. For the measurement of water level, the Seeed water level sensor was compared directly to the van Essen TD-diver. In this case, the two sensors scored equally well in most categories. The summarised findings suggest that if robustness and longevity are critical, the divers are preferable. If cost is more important, the Seeed sensor would be suitable. However, these sensors both received deductions in the "Usability" category, but for different reasons: the divers for the lack of real-time monitoring capability and the Seeed sensors for a lack of troubleshooting materials. The requirement for real-time monitoring is especially relevant in the municipal setting and should be taken into account when considering these two sensors.

3.3. Hydrological characterisation

3.3.1. Infiltration

Following construction, the fill media inside of MeSUDa settled and compacted, affecting the unsaturated hydraulic conductivity of the setup. Infiltration testing results (Figure 3.3) demonstrated a decrease in K in swale #1 of 58% over a three-month period. In this time, the soil was also routinely wetted (either via rainfall or experimental trials), further contributing to settling and compaction. The gradual decrease in K was expected, given that MeSUDa's bare soil surface lacked vegetation to promote soil structure and improve infiltration. Although only three data points were collected, the unsaturated hydraulic conductivity appeared to be stabilising at the end of the testing period.

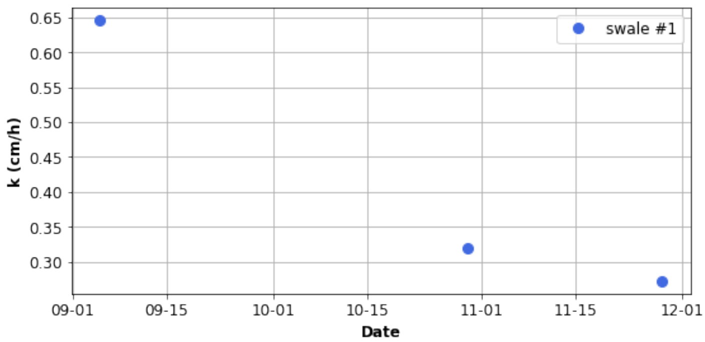


Figure 3.3: Unsaturated hydraulic conductivity (K) in MeSUDa, swale #1. Estimated volumetric water content of 30%.

The unsaturated hydraulic conductivity for each fill material was tested at mostly dry and partially saturated conditions (see Appendix 7.10 for complete results). K varied as expected between conditions and results were treated as indicative. The partially saturated K values, 0.04 cm/h for topsoil and 0.27 cm/h for fine sand, were compared to later experimental results (discussed in the following sections).

Limited infiltration was evident throughout testing, as any significant water influx resulted in ponding on top of MeSUDa. In swale #3, substantial pooling was observed at the front of the swale during attempted water table establishment (Figure 3.4a) and again during tracer testing (Figure 3.4c). Swale #1 experienced more localised pooling, also during tracer testing (Figure 3.4b). While MeSUDa was designed to promote slight ponding and gradual infiltration, the depicted extent of water accumulation was not desirable.

(a) Swale #3, filling attempt.

(b) Swale #1, tracer testing.

(c) Swale #3, tracer testing.

Figure 3.4: Pooling in MeSUDa.

3.3.2. Tracer testing, swale #1

The EC data collected from the tracer test in swale #1 appears to successfully capture both vertical and lateral movement of the salt pulse at it traversed the swale (Figure 3.5). As expected, the EC at the inlet remained constant at 10,000 μS/cm before slowly decreasing to baseline values, as the salt pulse infiltrated. The pulse is clearly represented in the breakthrough curves (BTCs) at locations #1 and #2, having arrived at shallower depths prior to reaching lower layers. A sharp increase in EC was observed at the outlet as well, signalling the arrival of the salt injection at the final monitoring point.

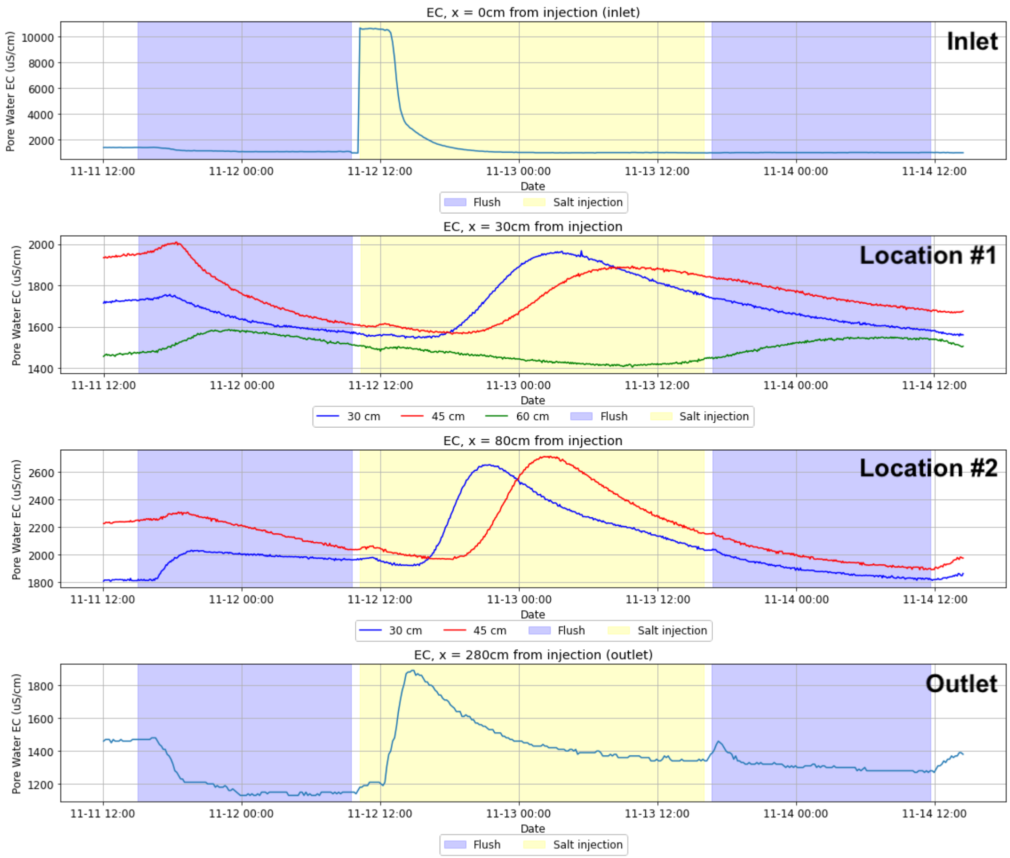


Figure 3.5: EC level at different monitoring locations for tracer testing in swale #1.

Three dynamics of Figure 3.5 were not anticipated. First, the salt pulse appeared to reach the 60 cm depth of location #1 later than anticipated and is identifiable by only a slight increase in EC. This delay and dampening of the response is attributed to the significant dispersion of the salt pulse that can already be observed at depths 30 cm and 45 cm (represented by widening of the BTCs). Second, the peak EC of the salt pulse at depth 45 cm, location #2, appears to be higher than that at depth 30 cm, indicating an unexpected increase in EC with increasing depth. When accounting for the baseline EC levels (those recorded just before salt injection), however, the magnitude of the pulse decreases from depth 30 cm to 45 cm. Third, and most notably, the salt pulse reached “downstream” locations prior to “upstream” monitoring points. In fact, the salt was detected first at the outlet, just two hours post-injection. Six hours after injection, it was observed at location #2, before finally reaching location #1 at eight hours. If the injected salt had traversed the swale as expected, this order of appearance would have been reversed. Findings and observations suggest that overland flow combined with the effects of preferential flow paths were responsible for the unexpected timing.

Overland and preferential flow

Although the water inflow was set to the lowest possible pumping rate to avoid overflow, enough spillage occurred to influence the tracer results. Further analysis concluded that some water from the injection ring had overflowed and travelled overland towards the front of the swale. Depressions in the topsoil, such as those above sensor installation points, led to localised pooling (Figure 3.6) and subsequent infiltration directly into the vertical sensor profiles. It was suspected that this vertical infiltration occurred more quickly at location #2 compared to location #1 because these sensors were freshly installed at the time of the tracer and the soil had not yet settled enough to eliminate any preferential flow paths. The salt pulse therefore appeared to arrive at location #2 prior to location #1.

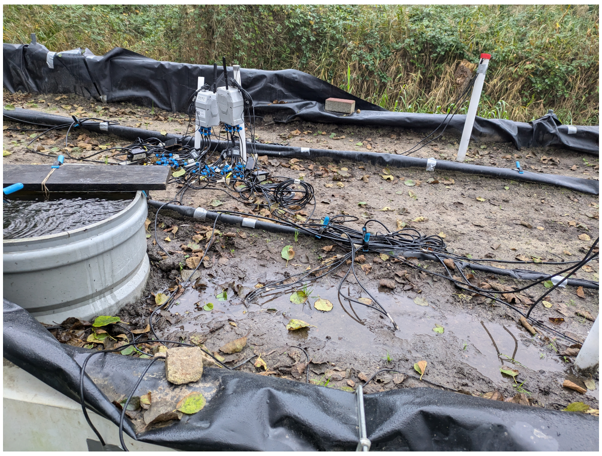


Figure 3.6: Pooling above sensor profiles during tracer testing in swale #1.

Significant pooling and subsequent ponded infiltration also occurred at the front of the swale. As at the sensor profile at location #2, preferential flow paths promoted rapid infiltration and an accelerated response at the outlet. This is reflected in the monitored discharge (Figure 3.7), where the connection between pumping status and outflow is apparent. Almost immediately after pumping was stopped on two occasions, the outflow sharply declined. Once pumping resumed, the outflow returned to previous higher rates after a slight delay. This pattern indicates that when the pump was turned on, water flowed over the ring sides and travelled to the front of the swale, where it quickly infiltrated and was discharged through the drain outlet. Water infiltrated most rapidly at this location and resulted in salt detection before the other monitoring locations.

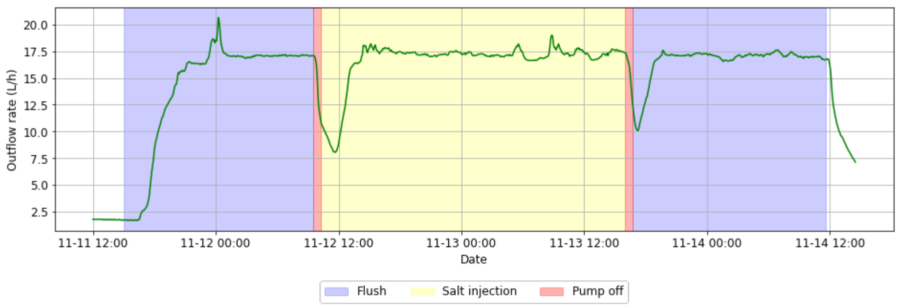


Figure 3.7: Discharge at outlet for tracer testing in swale #1.

Analysis of vertical transport at location #1

Given this analysis, it was determined that the tracer flow dynamics were dominated by overland flow in combination with vertical transport. Lateral transport was considered negligible and the use of a 1D vertical model was appropriate. The data collected at location #1 was retained as representative of vertical flow in MeSUDa with little influence from preferential flow paths. These findings also confirmed the use of location #1 data to calibrate the nitrate removal model, the results of which are discussed in section 3.4.

A range of indicative MRTs was calculated for the topsoil and fine sand layers, based on the BTC results in Figure 3.5. This calculation considered only advective transport, using the velocity of the peak concentration through the layer to compute the residence time. The estimated MRT in the topsoil was between 17 - 20 h with a mean time of 19.1 h, while fine sand was estimated between 13 - 30 h with a mean of 19.5 h. The large difference in range is likely due to the fact that the fine sand layer is considerably thicker than the topsoil layer. The estimated mean MRT for MeSUDa's entire vertical profile was 35 h. This finding was consistent with the required duration of the tracer testing.

The average vertical velocities through both layers and the corresponding unsaturated hydraulic conductivities (Table 3.2) are within the expected ranges for the experimental setup. The experimental K values are larger than the infiltration testing findings, but are considered more representative of MeSUDa's function than the infiltrometer measurements. Conditions during tracer testing were closer to saturated that those during infiltration testing, which also likely contributed to the higher conductivities. However, the unsaturated hydraulic conductivity of both materials is less than the inflow rate set by the pump. Operating the pump at 7 rpm (or 26 L/h), the unsaturated hydraulic conductivity needed to be at least 4 cm/h to prevent accumulation and eventual overflow in the injection ring.

Table 3.2: Average velocity and unsaturated hydraulic conductivity (porosity = 0.3) for swale layers. Infiltration testing K included for reference.

Layer	Ave. v [cm/h]	K [cm/h]	K, infiltration test [cm/h]
Topsoil	1.83	0.55	0.04
Fine sand	3.40	1.02	0.27

EC data at the inlet and outlet were used to calculate a mass balance, which yielded a salt retention of 67% inside swale #1. This finding was consistent with suspected retention and remobilisation of salts during initial flushing periods. Critically, this high retention suggests that the tracer testing was not conservative, as originally intended.

3.3.3. Tracer testing, swale #3

The tracer test conducted in swale #3 was not successful. Following salt injection, significant overland flow and pooling at the front of the swale were observed. The pump was switched off to avoid further overflow and allow for natural infiltration of the standing water. After four days, the water level inside the injection ring had not decreased and the tracer test was concluded.

EC levels at the four monitoring locations (Figure 3.8) did not indicate salt transport through swale #3. Inlet measurements clearly showed high EC levels in the injection ring, but no infiltration of the pulse, even after the pump was switched off ("salt infiltration" phase). Although locations #1 and #2 experienced variations in EC throughout the test, none of these fluctuations pointed to the transport of the salt pulse. Increases in EC were observed at the outlet, but were attributed to the overland flow and subsequent preferential flow of salted water down the front of the swale (as observed in the swale #1 tracer).

Low infiltration

The failure of the swale #3 tracer test was linked to extremely slow infiltration in the swale, which resulted in significant overflow of the injection ring, overland flow, and heavy pooling at the front of the setup. Observations during testing supported this theory. The soil around and just downstream of

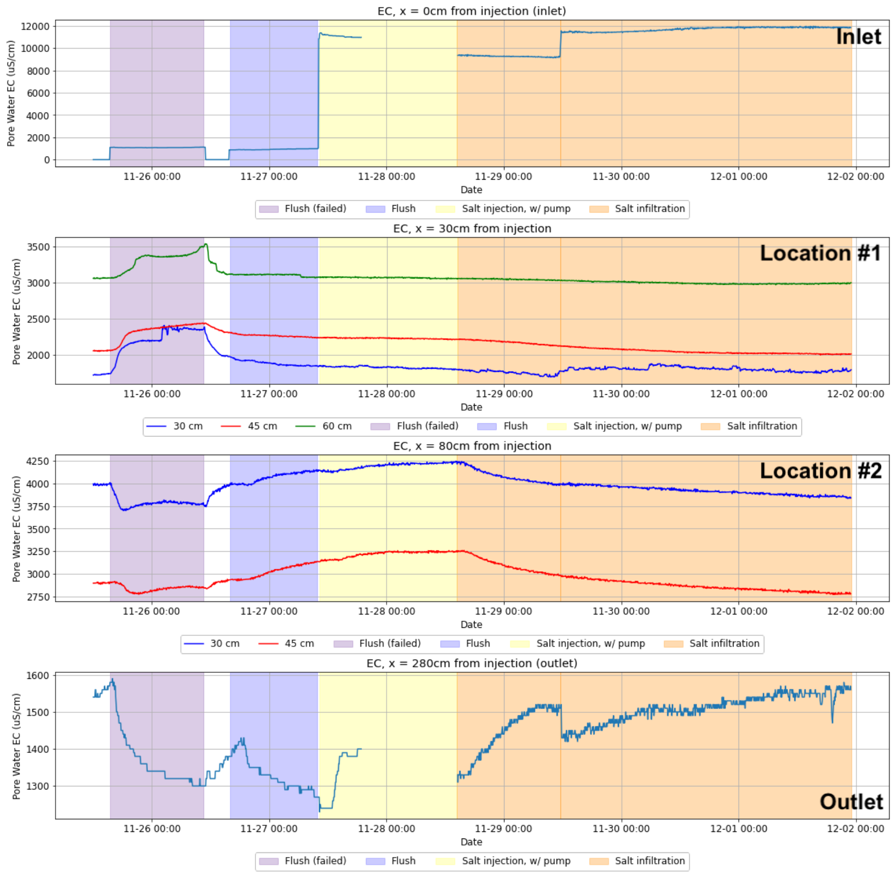
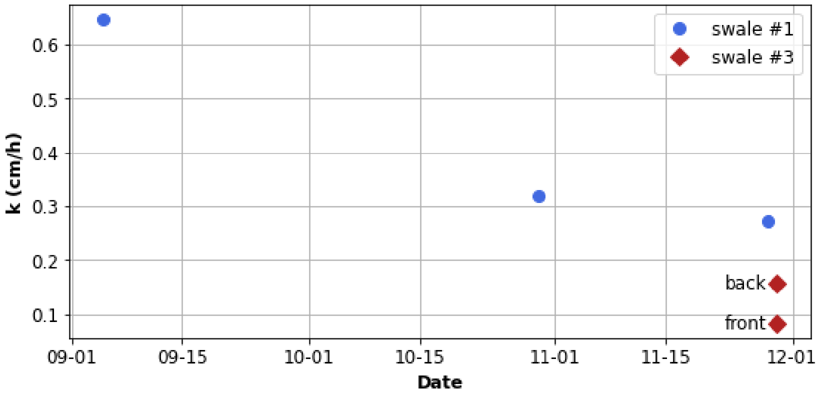


Figure 3.8: EC level at different monitoring locations for tracer testing in swale #3.

the injection ring had turned into mud (Figure 3.9a) and overland flow and pooling, more extreme than during the swale #1 tracer, were observed throughout the test. A nearly constant 3 - 4 cm of standing water was present at the front of the swale (Figure 3.9b) and the water in the ring had still not completely infiltrated six weeks after the conclusion of the tracer. Two infiltration tests were conducted in swale #3 at the end of the testing period, using the METER infiltrometer. One test was performed in front of the injection ring, 130 cm from the outlet, while the other was conducted behind the ring, 380 cm from the outlet. The unsaturated hydraulic conductivity in swale #3 was found to be considerably lower than that in swale #1 (Figure 3.3). The front location, where mud-like consistency was observed, had a conductivity of 0.08 cm/h, while the back location had a conductivity of 0.16 cm/h. It was expected that the infiltration capacity in the front of swale #3 would be the lowest, considering that water had been pooling there for an extended period. However, the back of swale #3 having half the infiltration capacity of swale #1 suggests an irregularity between the two swales.

(a) Mud around and in front of injection ring.

(b) Pooling at front of swale.



(c) Unsaturated hydraulic conductivity in MeSUDa, swales #1 and #3.

Figure 3.9: Evidence of low infiltration in swale #3.

3.4. Nitrate removal assessment

3.4.1. Model calibration and validation

Three parameters sets (Table 3.3) were calibrated to represent flow conditions in MeSUDa, based on the experimental tracer data. These sets, fitted at depths 42 cm, 45 cm, and 48 cm, represented the overestimation, correct, and underestimation, respectively, of the physical sensor placement during tracer testing. The error sets (over- and underestimated) differ from the correct set by 7%, 14%, and 8% for *v*, *D*, and *M*, respectively. These differences are considered small enough to suggest there were no large variations in flow characterisation resulting from a ± 3 cm sensor placement error.

Table 3.3: Fitted parameters sets for nitrate removal model.

	Velocity (*v*)	**Dispersion coefficient (*D*)**	**Mass coefficient (*M*)**
Overestimated (42 cm)	1.70	6.82	0.36
Correct (45 cm)	1.82	7.82	0.39
Underestimated (48 cm)	1.94	8.90	0.42

Validation was performed by comparing modelled and experimental data at a depth of 30 cm (Figure 3.10) using the RMSE and NSE (Table 3.4). Slight deviation between findings was expected, as the model was calibrated in the fine sand layer at 45 cm, but validated in the topsoil layer at 30 cm. Nonetheless, all parameter sets exhibit accurate concentration predictions (high NSEs) and relatively small error (low RMSEs). The nitrate removal model was deemed suitable for simulating conservative transport within MeSUDa.

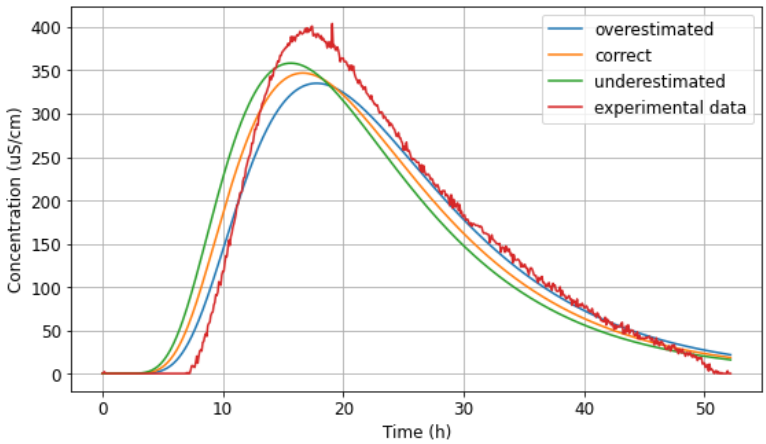


Figure 3.10: Modelled BTCs at 30 cm, validated against experimental data.

Table 3.4: Results for parameter sets validated at 30 cm.

Depth (cm)	RMSE (μS/cm)	NSE
overestimated	24.365	0.966
correct	29.458	0.950
underestimated	42.066	0.898

3.4.2. Tracer test MRT validation

Tracer test data was used with the 1D ADE stage of the model to simulate the MRT in each layer of the setup, as well as in MeSUDa as a whole (Table 3.5). Modelled findings include the effects of both advection and dispersion. As expected, the modelled MRTs are longer than the upper ends of the indicative ranges, which were derived directly from tracer data and only considered advection effects. This distinction demonstrates the important role of dispersion in transport through MeSUDa. Although the modelled MRTs were not vastly larger that the indicative values and advection still appears to be the dominant transport dynamic, the increase in residence time is enough to validate that dispersion is not negligible. This finding confirms the use of a combined advection-dispersion model (1D ADE) to simulate transport in MeSUDa.

Table 3.5: Modelled and indicative MRTs in MeSUDa.

Location	MRT [h]	Indicative MRT [h]
Topsoil	24.9 - 27.4	17 - 20
Fine sand	33.5 - 38.2	13 - 30
Woodchips	1.0 - 1.2	-
MeSUDa (whole)	58.3 - 65.6	26 - 50

3.4.3. Nitrate removal test cases

The four nitrate removal test cases (Table 3.6) were applied to calculate nitrate removal capacities within the woodchip layer (Figure 3.11). Each curve represents the concentration of $NO_3 - N$ that can be removed from the input pulse over time. The removal capacity (mg/L) was influenced by the incoming flow rate at 90 cm depth but constrained by the case-specific nitrate removal rate (mg/day). At early timesteps, low inflow rates resulted in a high theoretical removal capacity, allowing for complete removal due to low incoming nitrate concentrations. As the inflow rate increased, the constant removal rate became insufficient to match the incoming nitrate load, reducing the effective removal capacity and leading to partial breakthrough of $NO_3 - N$.

Table 3.6: Nitrate removal test cases.

	Case #1	Case #2	Case #3	Case #4
Nitrate removal [mg $NO_3 - N$/day]	193.7	354.2	32.7	59.7

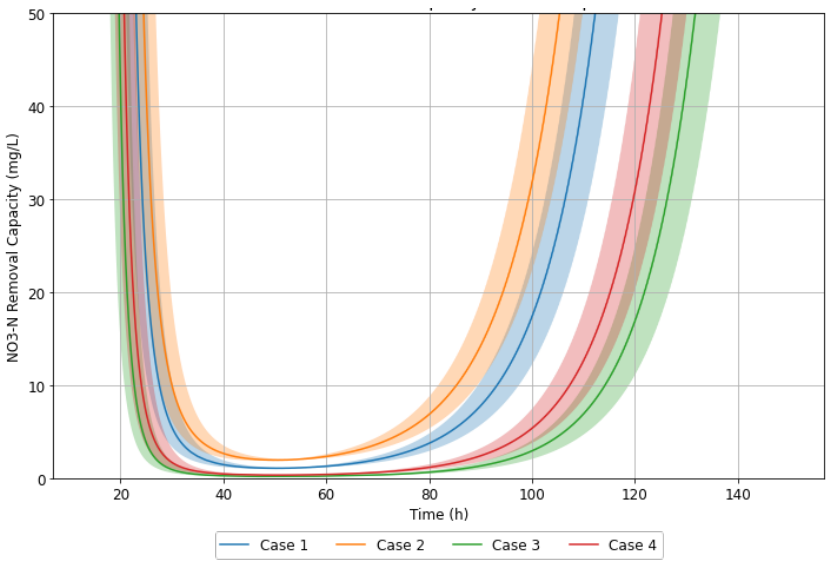
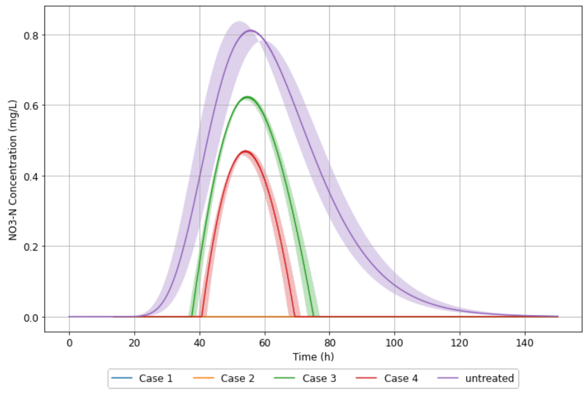
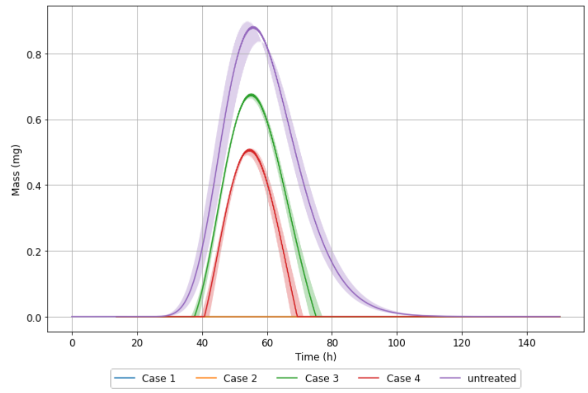


Figure 3.11: Theoretical $NO_3 - N$ removal capacity in woodchip layer.

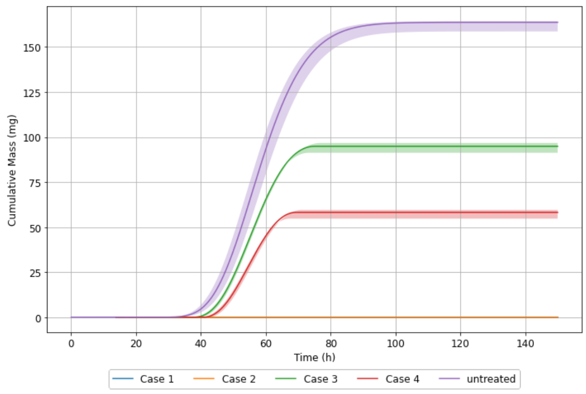
Following treatment in the woodchip layer at 90 cm, the $NO_3 - N$ breakthrough at the swale outlet was calculated (Figure 3.12a). This result was combined with the flow rate at the outlet to calculate the resulting mass loading (Figure 3.12b), the cumulative mass loading (Figure 3.12c), and the mass flux (Figure 3.12d). For the tested inflow conditions, cases #1 and #2, which both featured ideal denitrification, enabled complete removal of $NO_3 - N$. Case #4 resulted in a nitrate reduction of 64%, while case #3 only removed 42% of nitrate. With improved removal performance, the breakthrough of nitrate at the outlet was also delayed and shorter in duration.



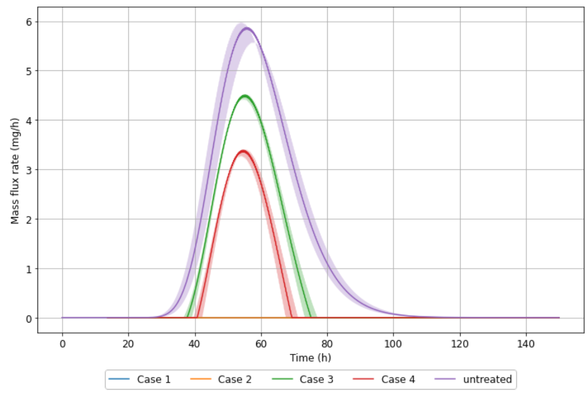
(a) $NO_3 - N$ concentration at outlet.



(b) $NO_3 - N$ mass loading at outlet.



(c) Cumulative $NO_3 - N$ mass loading at outlet.



(d) $NO_3 - N$ mass flux at outlet.

Figure 3.12: Modelled $NO_3 - N$ breakthrough at swale outlet.

3.4.4. Variation in woodchip mass and depth

To inform design recommendations, the effects of different woodchip masses and depths on the combined woodchip and denitrification zone MRT (Figure 3.13), as well as the nitrate removal capacity (Figure 3.14) were investigated. Since the size of the denitrification zone (between the woodchip layer and the outlet) was significantly larger than that of the woodchip layer in each depth/mass combination, it had the most influence over the combined MRT. It was therefore expected that the depth of the woodchip layer was far more relevant to the MRT than the mass of the layer was. For example, while the woodchip layer was set at 90 cm in the current swale setup, decreasing the depth by just 10 cm boosted the combined MRT by 92%. Leaving the layer at 90 cm but increasing the mass of woodchips from 16.5 kg to 30 kg only resulted in a 6% MRT increase. These results suggest that increasing the MRT, and theoretically the subsequent nitrate removal, would only require placing the woodchips at a smaller depth. However, in practice and considering the limitations of the current model, this relationship was not so simple and will be further discussed in chapter 4.

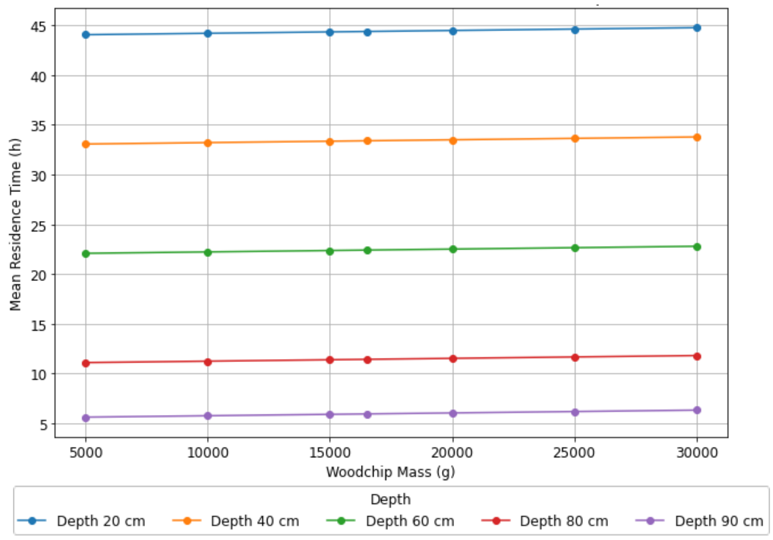
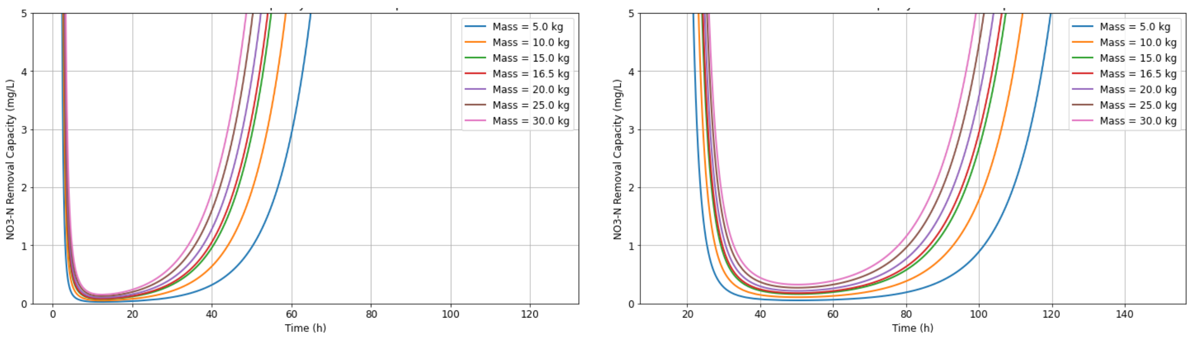


Figure 3.13: MRT for various woodchip masses and depths.

Changing the depth and mass of the woodchip layer also affected the layer's nitrate removal capacity (Figure 3.14). Here, the influence of increasing mass was easier to identify. As the woodchip mass increased, so did the minimum nitrate removal experienced during peak contaminated water flow. This likely would have resulted in less nitrate breakthrough at the swale outlet. However, the depth of the woodchips also played a role as it impacted the shape and duration of the removal capacity curves. An increase in depth resulted in a higher minimum removal capacity, suggesting less eventual nitrate breakthrough. The removal capacity curves also widened with increasing depth, indicating longer contact time between the woodchips and contaminated inflow.



(a) $NO_3 - N$ removal capacity at 20 cm depth.

(b) $NO_3 - N$ removal capacity at 90 cm depth.

Figure 3.14: $NO_3 - N$ removal capacities at different depths for various woodchip masses and depths.

The effect of changing the woodchip mass was most pronounced with respect to functional time and decay time (Figure 3.15). Given highly polluted rainfall (25 mg/L) and maintaining case #3's 42% removal rate, swale #3 could successfully operate between 0.5 - 4 years based on the average annual Dutch precipitation. Higher woodchip masses corresponded to increased functional times and decay times. This result was expected, as larger volumes of woodchips would need more time to decompose and would release greater amounts of DOC to support nitrate removal.

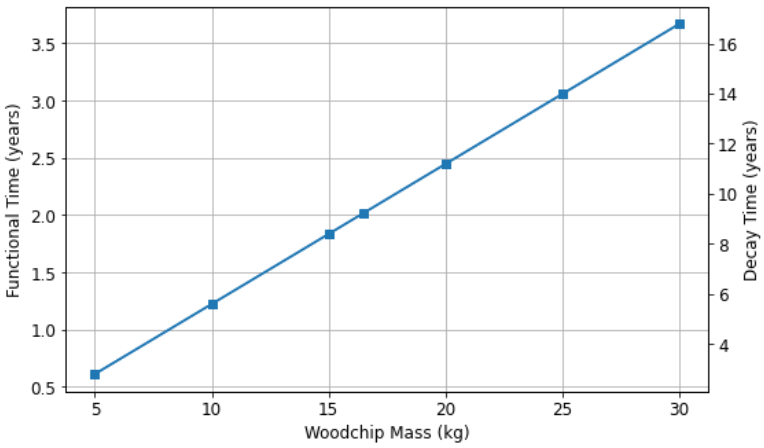


Figure 3.15: Effects of mass on function and decomposition of woodchip layer, 42% nitrate removal.

3.4.5. Qualitative decomposition analysis

Three samples of woodchips were assessed for decomposition indicators and assigned a decay status accordingly (Table 3.7, see Appendix 7.12 for complete results). The final decay status was selected based on the maximum intermediate status assigned during the assessment (no decay, minimal decay, or moderate decay, with the latter being the maximum status). General observations shown in Table 3.7 were not taken into account deciding the final status of a sample; these observations will instead be discussed in conjunction with the status in the following sections.

Table 3.7: Summary of decomposition assessment results. See Appendix 7.12 for full findings.

	Control, top	Control, bottom	Swale #3
DC3 classification	No decay	Minimal decay	Minimal decay
Pick test	No decay	Moderate decay	Minimal decay
Fungal inspection	Minimal decay	Moderate decay	No decay
General observations	Chips were stored in mostly dry environment. Did not clump together when collected, and feel very strong. Chips are fibrous, with distinct and sharp edges.	Chips were moist when collected and clumped together. Feel softened and smell earthy. Chips are fibrous.	All pieces are very small, smooth, and compact. Soil around extraction point was wet, with dark grey colour and distinct sulphuric smell. 5 different locations in swale drilled for samples, but woodchips only found at 2 locations.
Decay status	**Minimal decay**	**Moderate decay**	**Minimal decay**

The control sample collected from the top of a stored woodchip bucket ("Control, top") was intended to serve as a reference for no or minimal decay. These woodchips were fibrous with sharp, distinct edges, and were predominantly light coloured (Figure 3.16b). Some white and black spots were identified as potential early signs of rot fungi. The sample was assigned a final status of minimal decay, which was consistent with the stated observations and the expectation for this particular sample. A second sample was collected from the woodchip bucket, but from the bottom of the stored chips. Although originally intended to function as a second reference for minimal decay, this sample (called “Control, bottom”) was instead considered a benchmark for the “moderate decay” status. Similar to the other reference sample, these woodchips were still fibrous and rough. However, the bucket bottom chips were significantly softened and exhibited signs of decay in the DC3 classification and the pick test. Visual signs of decomposition were also evident: many pieces were blackened and some featured white, fuzzy patches (see Figure 3.16c), both indications of white rot fungi.

The final sample was extracted from swale #3 and was assigned a decay status of “minimal” based on the results of the DC3 classification and pick test. The chips did not exhibit any visual indicators of rot fungi. These findings were unexpected, as it was assumed that the woodchips in the swale had decomposed at least partially in the 230 days since their installation. However, unlike the other two samples, the general observations made at the time of extraction and analysis were not consistent with the assigned decay status. The extracted woodchips were compact and smooth, lacking the fibrous edges present in the reference samples (Figure 3.16a). The chips were also quite strong and hard. These observations indicate that the woodchips in swale #3 had indeed decayed, to the extent that the softer, more fibrous outer layers of the woodchips had completely broken down, leaving only the harder, more decay-resistant inner wood behind. Observations made during the extraction, namely that the soil had “a dark grey colour and distinct sulphuric smell”, are consistent with decomposition activity in swale #3. These observations are all considered indicators of anaerobic soil [38], suggesting that the microorganisms responsible for decomposing the woodchips had consumed the surrounding oxygen. Finally, only a handful of woodchips were retrieved from the swale, despite drilling in several locations. It is possible that many of the original woodchips had decomposed, and those that had not were too small to easily extract. Combined with the initial expectation of decomposition in swale #3, these observations indicate significant decay in the setup.

(a) Sample extracted from Swale #3.

(b) Sample extracted from top of stored bucket.

(c) Sample extracted from bottom of stored bucket.

Figure 3.16: Woodchip samples assessed for decomposition.

The analysed samples could serve as reference points for different stages of woodchip decomposition: the top bucket sample was representative of minimal decay, while the bottom bucket and swale samples were representative of moderate and significant decay, respectively. These findings are consistent with expectations and confirm that conditions inside of MeSUDa are suitable to decompose woodchips.

4 Discussion

4.1. MeSUDa function and bioswale representation

4.1.1. Setup and function

Measured TOC levels in the surface water (22.5 mg/L) were initially thought to exceed those of typical urban runoff. However, TOC concentrations in urban runoff can vary significantly. For example, in a study conducted along a congested road in Munich, Germany, runoff samples contained TOC concentrations ranging from 10 to 355 mg/L, with a mean of 71 mg/L [39]. It was therefore determined that the surface water TOC levels would not exclude its use as a proxy for urban runoff. Still, another concern remains that the elevated TOC levels could promote additional nitrate removal in MeSUDa by introducing supplemental carbon to the system. Should this carbon result in increased denitrification, it would be impossible to differentiate the effect from any woodchip-promoted denitrification occurring in the swale.

Another challenge in quantifying nitrate removal in MeSUDa is providing nitrate-rich influent, such that differences between inflow and outflow concentrations can be observed. The surface water only contained 0.19 mg/L of $NO_3 - N$, while potential nitrate removal in the swale was modelled with a heavily polluted inflow with 25 mg/L $NO_3 - N$. Although future experimental conditions do not need to match those modelled in the current research, literature indicates that inflow values between 0.4 and 50 mg/L $NO_3 - N$ will enable quantifiable changes in concentrations [14], [15], [17], [18]. Recommendations regarding the replication of nitrate-rich urban runoff will be discussed in chapter 6.

The leakage detected in MeSUDa during the attempted filling of swale #3 also affected the function of the setup and influenced experimental results. Most notably, the inability to establish a water table in swale #3 prevents experimental quantification of nitrate removal. Instead, the setup's denitrification potential was assessed using the nitrate removal model coupled with a physical analysis of woodchip decomposition. Although this approach was suitable for the present research, future studies regarding nitrate removal in MeSUDa will require the establishment of specific water levels to create an anaerobic denitrification zone.

Finally, while MeSUDa could exemplify the fill media of a bioswale, the setup currently lacks the vegetation characteristic to bioswales. The inclusion of vegetation in the setup may have significantly altered the soil-water dynamics observed during tracer testing. For example, vegetation can mitigate soil compaction and enhances infiltration capacity by strengthening subsurface structure through root growth [40]. This may have prevented the extremely slow infiltration and resulting pooling observed during tracer testing in swale #3. Vegetation is also relevant for any future studies regarding experimental nitrate removal in MeSUDa, as plants play a critical role in the nitrogen cycle [41].

4.1.2. Hydrological characterisation

Although MeSUDa was designed to facilitate gradual infiltration, the extent of pooling and slow infiltration observed in swales #1 and #3 is not desirable. While swale #1 exhibited only localised ponding during tracer testing, swale #3 experienced substantial pooling stemming from exceedingly low infiltration. Decreased infiltration in both swales #1 and #3 is likely caused by soil compaction, which was expected after constructing MeSUDa and allowing the materials to settle. Moreover, both swales had only bare soil on the surface and no vegetation to promote infiltration and develop soil structure. However, the stark difference in infiltration capacity between the swales indicates another compaction cause is present in swale #3. The only difference in treatment of the swales was the attempted establishment of a 50 cm water level in swale #3, as described in section 3.1.2. Despite the suspected leakage in this swale, it was possible that the large amount of water added during the fill attempt contributed to additional compaction of the topsoil and fine sand layers. It was also evaluated whether the fill media of swale #3 was more saturated, or possibly supersaturated, compared to swale #1. However, soil moisture measurements collected during both tracer tests (see Appendix 7.11) showed that this was not the case: swale #3 had lower soil moisture levels than swale #1 in both the topsoil and fine sand layers.

Suspected compaction in swale #1 affected the unsaturated hydraulic conductivity (K) and MRT, both used to characterise MeSUDa in the context of bioswale design guidance. The unsaturated hydraulic conductivities of the topsoil and fine sand layers were calculated as 0.55 cm/h and 1.02 cm/h, respectively. These results are consistent with theoretical conductivities for sandy clay loam (topsoil layer) and sandy loam (fine sand layer) [42], but are lower than the suggested design parameters of bioswales. Ekka et al. [3] suggest an infiltration rate of 2.5 - 5 cm/h, while Woods Ballard et al. [6] suggest 10 - 30 cm/h. However, infiltration rates in active swales can vary considerably in practice [43] and MeSUDa's experimental values remain in the same order of magnitude as the design guidance. With respect to the MRT, modelled findings indicated that MeSUDa's MRT is between 58 and 66 hours. This range is considerably higher than the emptying time of 24 - 48 hours recommended by Woods Ballard et al. [6]. Since the emptying time is set to prevent bioswales from flooding with runoff from subsequent rain events, the elevated MRT in MeSUDa is likely not representative of a full-scale bioswale. In this context, the pooling observed in MeSUDa is also conducive to increased flooding risks and undermines the representativeness of the setup.

4.1.3. Limitations

Several limitations stemmed from the tracer testing methodology. Data collection, specifically for EC measurements, was limited to only two vertical profiles (five sensors in total) and one sensor each at the inlet and outlet. The installation of additional vertical profiles, including sensors at the bottom of the swale (next to the underdrain), would have allowed for a more comprehensive overview of transport dynamics in two dimensions. Had the sensors been installed well before tracer testing commenced, the preferential flow paths along the installation drill points may have been avoided and any lateral flow would have been captured. Instead, the current setup limited usable data to a single vertical profile of sensors (those at location #1), which were not impacted by the overland flow and preferential infiltration.

The injection strategy was also a limiting factor in tracer testing. Ponded infiltration using an injection ring was selected to avoid any overland flow and contain transport dynamics to a small, observable section of the swale. This technique was more manageable than a full-swale ponded tracer. It also allowed for the optimisation of monitoring points, considering the limited number of available sensors. Despite the favourable injection method, the use of the peristaltic pump to continuously add water to the injection ring was a suboptimal choice. Even with the pump operating on the lowest speed (infiltrating at 4 cm/h), the unsaturated hydraulic conductivity in the swale was too low to infiltrate the pumped inflow. The pump gradually overfilled the ring, resulting in overflow and subsequent overland flow/pooling. Combined with rapid infiltration via preferential flow paths observed during the swale #1 tracer, this event invalidated the EC measurements at location #2 and the outlet. Due to the interference of the vertical flow at these locations, any lateral flow dynamics in the swale were impossible to identify. Only the data at location #1 was considered sound and further analysis was reduced to one dimension. Additionally, the continued use of the location #1 data was limited to the assumption that all influent had travelled down that vertical profile, which was not the case. This assumption was critical in using the tracer data as an input to model conservative transport. The impacts of this limitation on the modelling of nitrate removal will be further discussed in section 4.3.3.

4.2. Assessment of low-cost sensors

4.2.1. Findings and municipal application

All low-cost sensors received lower total scores than industry standard counterparts in the performance indicator assessment. Every low-cost sensor underperformed in the categories of “Robustness” and “Usability”, and two out of three were not considered as reliable as the industry standard devices. Each low-cost sensor performed well in the “Setup” category and outperformed the more expensive industry standard sensors in “Cost”. These findings indicate that savings in equipment costs were routinely tantamount to reduced hardware quality and longevity. This trade-off is consistent with literature on the topic of low-cost sensing, as reported by Zhu et al. [25]. Even if a low-cost sensor performed well in validation and field testing, equipment failures introduced an element of unreliability that is not desirable in experimental work, particularly long-term studies. For example, the inexpensive Seeed water level sensor performed exceptionally well during validation testing, was easy to install, and enabled real-time remote monitoring of water levels. This sensor scored the highest out of the three low-cost sensors and had it not been associated with hardware issues, it would have likely outperformed most of the industry standard sensors (including the van Essen divers, its high-cost counterpart). However, the water level sensor requires the use of the Seeed data logger, which experienced continuous sensor port malfunction, nullifying the weatherproofing of the logger casing. This introduced risk of sensor failure and reduced the overall dependability of the water level sensor. Another commonality between the low-cost sensors was the lack of general documentation, particularly troubleshooting resources. Although the availability of such materials depends on the manufacturer, the tested industry standard sensors were very well-documented and supported by a multitude of online resources. The low-cost sensors had limited documentation and accompanying online information. In the case of the currently assessed sensors, this distinction is likely the result of comparing specialised environmental sensing companies with a significantly larger retailer of multi-disciplinary hardware and sensing products.

The suitability of low-cost sensors for municipal implementation is highly dependent upon the requirements and limitations of a given monitoring campaign. For any application in which cost is not a limitation, industry standard sensors are recommended provided they meet all other project requirements. If cost is a priority (as found in literature [24], [25]), then low-cost sensors are considered suitable for implementation in short-term monitoring projects involving general measurement parameters (i.e., soil moisture, EC, discharge, or water level). However, according to Oosterveld [22], continuous and long-term municipal monitoring is needed. For these applications, investment in industry standard sensors is preferable as longevity benefits outweigh potential low-cost sensor replacement and repair costs, as well as accompanying monitoring interruptions.

These recommendations apply to the three low-cost sensors assessed in the current project. In the case of soil moisture and EC monitoring, the Seeed SM/T/EC sensor is an appropriate selection for short-duration projects in which high accuracy is not critical and instead only approximate ranges and trends are of interest. The Seeed EC/TDS sensor is equally applicable for projects entailing EC and/or TDS measurements, although could support higher accuracy sensing since on-board calibration is possible. Finally, the Seeed water level sensor is also suitable for shorter projects. However, given this sensor's strong validation performance and its capability for remote monitoring, it is also recommended over its industry standard counterpart (the van Essen diver) for longer term projects. In this case, the ability to assess water levels continuously and remotely is assumed to be critical to municipalities (as stated in [23], [24]) and would outweigh the risk of data logger malfunction.

4.2.2. Limitations

The assessment of low-cost sensors was primarily limited by insufficient direct comparison to industry standard counterparts. Only two sets of parameters (SM/EC and water level) in the current research were measured with both low-cost and industry standard sensors. Although these comparison results and overall sensor performance offer insights into both groups, the collected data is insufficient to definitively determine which is better suited for municipal applications. For example, although the Seeed water level sensor performed well during validation, it and the van Essen diver were not tested in parallel over a longer duration to compare measurements. The same is true for the Seeed SM/T/EC sensor. Furthermore, in the cases of discharge and EC/TDS, only a single industry standard sensor and a single low-cost sensor were assessed, respectively. Without direct comparison between the two groups, it is impossible to distinguish whether the measurement parameter influences the suitability of a low-cost sensor. As discussed in section 3.2.1, it is suspected that the Seeed water level sensor performed better in validation than the Seeed SM/T/EC sensor because the measurement of water level with a piezoresistive sensor is more straightforward than the measurement of soil moisture and EC. A low-cost sensor might prove to be a suitable alternative for water depth measurements, while an industry standard sensor is required for soil moisture and EC. Similar distinctions might be true for other measurement parameters, but the current data is not adequate to perform such an analysis.

Another limitation of the sensor assessment is the inclusion of only a single validation test (when applicable) and a short duration experimental period. Sensor performance was not assessed over time, and therefore no insights regarding stability or drift could be developed. Additionally, the validation methodology used for two of the sensors (METER tipping buckets and Seeed SM/T/EC sensors) likely contributed to a poorer performance, thereby affecting final indicator scores.

4.3. Nitrate removal in MeSUDa

4.3.1. Nitrate removal model

The modelled $NO_3 - N$ breakthrough at the swale outlet demonstrated that all four test cases would effectively reduce effluent nitrate levels compared to an untreated case. This result was expected, as was the complete removal enabled by the two cases featuring "ideal denitrification". Case #3 results, considered to be the most representative of conditions in MeSUDa, suggested that the setup could support $NO_3 - N$ removal up to 42% from contaminated influent. This removal rate is higher than the average rates found in two full-scale setups: Passeport et al. [14] obtained nitrate removal up to 33%, while the setup of Wicke et al. [15] removed less than 5% on average. However, the latter observed removal rates greater than 50% when the residence time exceeded 48 hours [15]. Considering MeSUDa's MRT of 58 - 66 hours, a removal of 42% is consistent with a bioswale featuring an extended residence time.

Modelled MRT in the combined woodchip and denitrification zones was primarily governed by woodchip layer depth. This result suggests decreasing the woodchip depth would significantly increase the MRT and therefore nitrate removal. However, the MRT is not the only parameter that dictates denitrification.

The flow conditions, particularly the dispersion, and resulting MRT in the woodchip layer alone determine DOC uptake, which subsequently constrains denitrification. Therefore, if the woodchip layer were placed at a smaller depth, the movement of water through the layer would closely resemble a pulse input with little dispersion. The MRT of the water at this depth would be relatively smaller than that at a deeper woodchip layer, meaning that DOC uptake would be limited, as would the resultant nitrate removal capacity. This effect was demonstrated in Figure 3.14. The removal capacity at a depth of 20 cm was approximately 50% smaller than that at a depth of 90 cm. Although the difference in concentration removal appears small, the reduced removal capacity would be applied to a higher, non-dispersed peak concentration. Therefore, by decreasing the depth of the woodchip layer, the risk of a higher $NO_3 - N$ concentration breakthrough at the outlet is increased. This effect might be counteracted by increasing the mass of woodchips, but the model's limitations prevented the analysis of $NO_3 - N$ transport post-removal, so this claim was not possible to confirm. The results of the assessment do clearly indicate, however, that denitrification in MeSUDa could be optimised by pairing a shallower woodchip depth with a greater woodchip mass.

Functional time analysis indicated that larger woodchip masses enable longer durations of nitrate removal under average rainfall. However, this calculation assumed complete availability of DOC released during decomposition of the entire woodchip mass. In practice, decomposition and denitrification occur simultaneously, potentially constraining nitrate removal at any given moment. As a result, the functional time would be distributed over the woodchip layer's decay period, which also increased with mass. When this distribution is accounted for, the ratio of functional time to decay time would remain constant across different masses. This suggests that installing smaller woodchip masses at regular intervals in MeSUDa may be preferable, avoiding extended periods of insufficient nitrate removal.

4.3.2. Decomposition assessment

The woodchips stored in the bucket presented different levels of decay depending on whether they were on the open top or enclosed bottom of the container. The chips on top were likely kept at a colder temperature and in drier conditions, thereby preventing the progression of decomposition; these were considered the reference for minimal decay. Woodchips taken from the bottom of the container had likely been exposed to warmer conditions and were noted to be slightly damp upon extraction. These conditions are conducive to decomposition [44] and the woodchips showed signs of moderate decomposition, for which they became the benchmark. The third sample, which was extracted from swale #3, presented signs of significant decay which suggests favourable conditions for decomposition inside of the swale. This physically confirms the first step for DOC release and subsequent denitrification in MeSUDa. Moreover, the observation of decay provides insight into the conditions at the woodchip layer. Based on the publication of Käärik [44], wood decomposition necessitates a moist and warm (20 - 32 degrees Celsius) environment. Furthermore, soil pH should be between 4 and 9 and oxygen should be readily available (i.e., aerobic conditions). The observation of decomposition in swale #3 suggests that these conditions were met. Although sufficient to confirm the conditions necessary for decay inside of MeSUDa, the results of the assessment were limited to purely qualitative observations. Each of the three stages of assessment (DC3 classification, pick test, and fungal inspection) was subject to interpretation and therefore vulnerable to personal observation error.

4.3.3. Limitations

The most significant limitation of the nitrate removal model is the oversimplification of the processes and conditions responsible for nitrate removal. Removal in MeSUDa was approximated by only three processes: decomposition, DOC release, and denitrification. All three processes are highly dependent upon environmental conditions such as temperature, pH, and microbial activity [37], [45], [46]. It was unlikely that the conditions in each study site from which process rates were derived were identical to those in MeSUDa. Furthermore, the wood species used in the woodchips placed in MeSUDa was unknown, although assumed to be some type of coniferous species. This is relevant to both the decomposition rate, as well as the DOC release rate [45], [47]. Also relevant to these processes was the location of the woodchips. The woodchips in Moorman et al.'s study [31] to quantify decomposition were surrounded by a combination of other chips and soil, while the wood in Hollands et al.'s DOC release research [32] was located on the forest floor. The woodchips in MeSUDa were located in the subsoil, compressed between layers of fine sand, likely affecting decomposition and DOC release [48].

Moreover, conditions in the swale were assumed to be suitable for DOC release and subsequent denitrification. The woodchip layer was assumed to remain aerobic, providing sufficient oxygen for the decomposition of the woodchips, while the volume directly underneath the woodchips was assumed to be entirely anaerobic, as required for denitrification. Realistically, the transition between aerobic and anaerobic occurs in a gradient. The model also simulated DOC release and subsequent denitrification as an instantaneous process as the flow passed through the woodchip layer. The modelled nitrate removal capacity was calculated based only upon the flow dynamics through the layer and the size of the anaerobic zone was not considered in the model. In practice, denitrification would not be solely dependent upon dynamics in the woodchip layer, but also the flow dynamics (e.g., dispersion and the resulting MRT) in the anaerobic zone. Moreover, denitrification would not occur constantly throughout this zone, as the oxygen content (as well as other conditions, like pH and temperature) would vary with depth.

Several other limitations stemmed from the input conditions provided for the model. Only a single successful tracer experiment was performed in swale #1, supplying the data needed to calibrate the model. No tracer data from swale #3 could be used, despite being the model's target swale. Furthermore, actual conditions inside of MeSUDa (such as pH, redox potential, and DO) were unknown and not included in the analysis input. The model is also limited to one-dimensional, conservative transport. Although the tracer results from swale #1 indicated predominantly vertical flow and lateral flow was assumed negligible, it could not be entirely ruled out.

As briefly discussed in section 4.1.3, the observed overland flow during the swale #1 tracer suggested that the collected data at location #1 was not representative of conservative transport. A mass balance also demonstrated overall high salt retention (67%) in the swale. Since this tracer data was used to fit conservative transport parameters, the modelling of advection and dispersion is likely not reflective of actual flow conditions in MeSUDa.

Furthermore, because the model is limited to conservative transport, it is unable to predict further nitrate transport during and following removal. While the model could simulate effluent nitrate concentrations for a woodchip depth of 90 cm under the assumption that changes between 90 cm and the outlet were negligible, it could not do the same for shallower depths. Finally, the findings presented in section 3.4.3 are only applicable to removal potential in MeSUDa and cannot be extrapolated to full-scale bioswales. The inflow conditions provided to the model represented a rainfall event impacting only the surface area of MeSUDa. In implementation, bioswales are used to treat runoff from larger collection areas, which would increase inflow volume and reduce the nitrate removal ability of the swale.

5
Conclusions

An OAL, dubbed MeSUDa (Managed experimental Sustainable Urban Drainage area), was constructed at Flood Proof Holland with the objective of representing a full-scale bioswale. Instrumentation was selected and purchased for MeSUDa, with a specific focus on low-cost sensors as an alternative to industry standard devices. The sensors were validated and a series of performance indicators was used to assess their overall function as well as their suitability for municipal bioswale monitoring. Finally, experimental tracer testing was conducted to characterise MeSUDa's hydrological function, and an advection-dispersion based model was developed to assess the potential for enhanced nitrate removal in the setup.

The following research questions were addressed in the study:

1. *Can the OAL be designed and constructed such that it can replicate the hydrological behaviour of a bioswale?*

   Yes, MeSUDa can be designed and constructed to replicate the behaviour of a bioswale. This research has demonstrated that, as an OAL, MeSUDa can be used to conduct controlled water quality and water quantity experiments with continuous monitoring employed. The setup's applicability extends beyond hydrological characterisation of bioswales; MeSUDa offers a flexible platform that is suited to a range of bioswale research topics. Moreover, the site is highly customisable, and design features such as fill media, water levels, vegetation, and sensing locations can all be tailored to project needs.

   However, MeSUDa's *current* design prohibits complete representation of a full-scale bioswale. At the outset of the present project, design criteria for infiltration should have been more clearly defined. Gradual infiltration was preferred for the setup, but was unacceptably slow in practice and led to overland flow and pooling. This impacted hydrological testing, as did the presence of preferential flow paths at the fronts of the swales and along the vertical sensor profiles. The extent of pooling, overland flow, and preferential flow in MeSUDa limits the replication of bioswale behaviour. Consequently, the setup's infiltration rate and MRT do not fall within recommended design ranges. The experimental infiltration rate is lower than recommended, but not substantially. Infiltration rates of active bioswales can vary significantly, so this finding alone does not limit representativeness. Conversely, the modelled MRT for the setup is larger than the recommended bioswale emptying time and does detract from the hydrological replication. Finally, a leakage was detected that prevented the establishment of an anaerobic denitrification zone.

   With specific design changes, MeSUDa can become representative, physically and hydrologically, of a bioswale. The leakage can be repaired and a denitrification zone created, while the addition of vegetation may improve the infiltration rate (and MRT), reducing pooling and overland flow. Preferential flow paths are likely to disappear over time as fill media continue to settle around sensors and the swale fronts.

2. *Do low-cost sensors provide a suitable alternative to industry standard sensors for local monitoring of water quantity and quality within active bioswales?*

   The suitable application of low-cost sensors in municipal bioswale monitoring is dependent upon the requirements and limitations of the measurement campaign in question. This study has demonstrated an inverse relationship between sensor cost and quality/longevity. When cost is not a priority, industry standard sensors are still recommended over their low-cost counterparts. Since municipalities may have limited resources, however, low-cost sensors can be successfully employed as an alternative to more expensive sensors in specific cases. For projects in which only short-term monitoring is required, low-cost sensors are considered acceptable to measure general water quantity and quality parameters (i.e., water level, discharge, soil moisture, and EC). In cases where continuous long-term monitoring is necessary, low-cost sensors do not provide a suitable alternative. For these applications, investing in higher quality sensors is suggested to avoid potential replacement and repair costs, as well as the risk of monitoring interruptions.

3. *Can the OAL be employed to assess the potential for enhanced nitrate removal via the addition of organic carbon sources, based purely on hydrological insights?*

   Yes, this research demonstrated that MeSUDa can be used to assess enhanced nitrate removal based solely on hydrological findings. The qualitative decomposition assessment performed on woodchips from the setup confirmed favourable conditions for wood decay and therefore the potential of additional DOC release and subsequent denitrification (given the correct conditions). Modelled nitrate transport and removal indicated that the presence of woodchips in MeSUDa would result in elevated nitrate removal compared to an untreated case. Furthermore, the findings for case #3 (most indicative of the actual setup) suggest a nitrate load reduction of up to 42%. Lastly, an analysis of woodchip mass and placement demonstrated that the nitrate removal ability of the setup could be further improved by increasing the mass of woodchips and moving them to a shallower depth. However, the functional time versus the decay time of the woodchip layer should be taken into account when selecting a woodchip mass.

The current research has proven that an OAL can facilitate the development and monitoring of bioswales. A solid foundation for further research on the interactions between bioswales and water quality has been established and continued study on woodchip-enhanced denitrification is expected. Moreover, MeSUDa is equipped to serve as a testing ground for monitoring equipment in the context of both academic and municipal applications. Despite the promising initial results regarding MeSUDa's general function and potential for enhanced nitrate removal, several limitations were present in the current study. Recommendations for addressing these constraints are provided, as are suggestions for continued research into nitrate removal and the assessment of low-cost instrumentation.

6 Recommendations

6.1. MeSUDa and future research

MeSUDa has demonstrated its value as an OAL, allowing for controlled yet customisable bioswale research. It is therefore strongly recommended that the facility be employed in future endeavours. MeSUDa can support the study of numerous topics, including water quality, water quantity, monitoring, vegetation, and design optimisation. In particular, the topics addressed in the current project warrant continued investigation, as the presented findings are foundational and several uncertainties remain. Recommendations discussed in the following sections are based in the continuation of this research.

6.2. MeSUDa setup

It is recommended that the leakage in swale #3 be located and repaired as soon as possible. Once MeSUDa is entirely isolated and the plastic lining is confirmed to be impermeable, tracer testing can be performed without risk of water/mass imbalances. It will also be possible to establish a water table in swale #3 and create an anaerobic zone underneath the woodchip layer, as required for any experimental work involving denitrification. Vegetation should be added to the setup (see recommended types in Appendix 7.2) to better replicate a full-scale bioswale. The root structures of the vegetation should promote infiltration and prevent excessive compaction of the fill media, reducing the overland flow and pooling observed in the setup. Improvement of the infiltration capacity should improve MeSUDa's unsaturated hydraulic conductivity and MRT, aligning them with design guidance. The addition of vegetation will also provide opportunities to study its effects on MeSUDa's hydrological behaviour. Findings from the present research can then be used as (bare soil) baseline results for comparison. If the addition of vegetation does not sufficiently increase infiltration, then replacement of the current topsoil with a more permeable alternative may be considered.

6.3. Instrumentation

To further assess which low-cost sensors might serve as alternatives to industry standard sensors, municipalities should first be consulted. It is recommended that municipalities be surveyed to gather data on their monitoring needs and priorities regarding active bioswales. Based on the findings of this survey, relevant low-cost sensors and their industry standard counterparts can be tested in MeSUDa. Testing should occur in parallel and continuously over a time frame of at least one year to assess seasonal effects, under variable conditions (moisture levels, temperature, etc.). Combining direct sensor performance comparisons with municipal requirements will better support a definitive recommendation between low-cost and industry standard sensors.

6.4. Experiments

6.4.1. General

A limitation of the current setup is that the selected water source for experimental influent (nearby surface water) may not contain high enough nitrate levels to quantify removal in MeSUDa. Furthermore, elevated TOC levels may influence denitrification inside of the setup. The pre-treatment/alteration of the surface water or the use of different sources entirely could be considered. Pre-treatment to increase nitrate concentrations would entail spiking the influent, a strategy employed in various studies [15], [16], [18], [17]. However, if removal in MeSUDa is insufficient, the nitrate-contaminated effluent cannot be discharged into surroundings as it currently is. Treatment or removal of the effluent would be required, which is not seen as logistically or financially feasible. Moreover, the removal of TOC prior to experimental use is also not possible, as this requires advanced treatment technologies [49]. It is therefore suggested that different water sources be considered for use. Alternatively, the effects of TOC-rich influent on nitrate removal in bioswales could be investigated.

The continued installation of standard water quality and quantity parameter sensors is also advised. In particular, it is suggested that vertical profiles for temperature, pH, DO, and soil moisture be installed at regular intervals along the entire length of at least one of the experimental swales. Water level sensors should be included at these intervals as well, particularly if a water table is established in the setup. The expansion of the current instrumentation would enable comprehensive, long-term monitoring with the goal of developing a reliable baseline dataset for MeSUDa.

6.4.2. Hydrological characterisation

Further tracer testing is recommended to refine the characterisation of flow dynamics and assess variability between swales in MeSUDa. Experimental data should also be collected for swale #3 to inform transport in the nitrate removal model. Following improvement of infiltration, tracer tests should be repeated using the current methodology. If infiltration remains insufficient, then the methodology can be modified to include a longer testing duration (> five days) with a smaller ponding head in the injection ring to mitigate any lateral flow effects (10 cm is suggested). Additionally, an alternative pumping mechanism should be installed to decrease the inflow amount and avoid overflow at the injection point. If the injection ring methodology is still not suitable, then a ”buried tracer test” is recommended. This would entail installing tracer chemicals in crystalline form at various locations/depths, with different chemicals used for different locations (distinction of results per location). Water can be ponded on top of the swale and the crystals will dissolve into the flow as the water infiltrates. No existing literature was found employing this technique, but using salt and rhodamine is suggested. Both substances are conservative and are commonly employed in traditional tracer testing.

6.4.3. Nitrate removal

Several experiments are suggested to quantify nitrate removal in MeSUDa. The transition from an analytical to a physically-based model would be advantageous, but requires the collection of sufficient environmental data from swale #3. pH, DO, and temperature sensors should be installed at various depths (ideally in all layers of the swale) and lateral locations. A redox potential probe should also be placed such that it can gather data from multiple depths to locate the aerobic and anaerobic zones of swale #3. Data on environmental conditions should be linked to decomposition, DOC release, and denitrification rates in MeSUDa. It is recommended that these rates be experimentally determined in swale #3. The combined condition-rate findings can be fed into the model to more accurately quantify nitrate removal and produce results that are representative of conditions in MeSUDa.

A long-term decomposition study should be performed to investigate how often the woodchip layer would need to be replaced to sustain denitrification in the setup. Since decomposition is not a linear process (as evidenced in [31]), the study should include regular assessment of decomposition progress to build an accurate trend. Based on the qualitative decomposition assessment and the analysis of functional and decay times for different woodchip masses, it is estimated that at least a year will be needed to perform this study. It is recommended to refer to the experimental procedure detailed by Moorman et al. in [31], as the authors conducted a similar, long-term evaluation.

6.5. Modelling

6.5.1. MeSUDa design

An unexplored application of the nitrate removal model is the optimisation of the design and setup of MeSUDa. Since the range of suggested infiltration rates for bioswales is large, it is recommended to optimise infiltration for MeSUDa. The current nitrate removal model could be modified to consider the design guidance infiltration rates and MRT, as well as the resulting potential nitrate removal, and suggest an infiltration rate accordingly. Findings could be used to inform design changes to the current setup. A thorough analysis of the woodchip layer depth and mass is also recommended, with the goal of optimising MeSUDa's woodchip layer. The findings of the currently modelled nitrate removal indicate that shallower depths (< 90 cm) and increased woodchip masses are conducive to improved nitrate removal. The results of the optimisation could be considered in tandem with the outcome of the long-term decomposition study, to estimate how long the optimal woodchip layer would remain effective.

6.5.2. Nitrate removal

A significant limitation of the nitrate removal model is that it is limited to conservative transport. The incorporation of non-conservative transport (i.e., active and variable denitrification) throughout the anaerobic zone is critical to improving the prediction of effluent nitrate loading. Within this addition, the experimentally determined process rates (decomposition, DOC release, and denitrification) and environmental conditions should be integrated such that removal can be modelled based on actual conditions in MeSUDa. This will become particularly relevant once a water table is established in swale #3, as the physically-based removal model could then simulate the effects of varying anaerobic zone sizes.

Finally, the current model considers an inflow of only rainfall on the surface of the swale, which is not representative of a standard urban bioswale. Moreover, the presented nitrate removal findings were calculated for a heavily polluted, above-average rainfall event. It is recommended that inflow volumes corresponding to various runoff collection areas are modelled, as are a wider variety of inflow events.

7 Appendices

7.1. Original draft design for MeSUDa

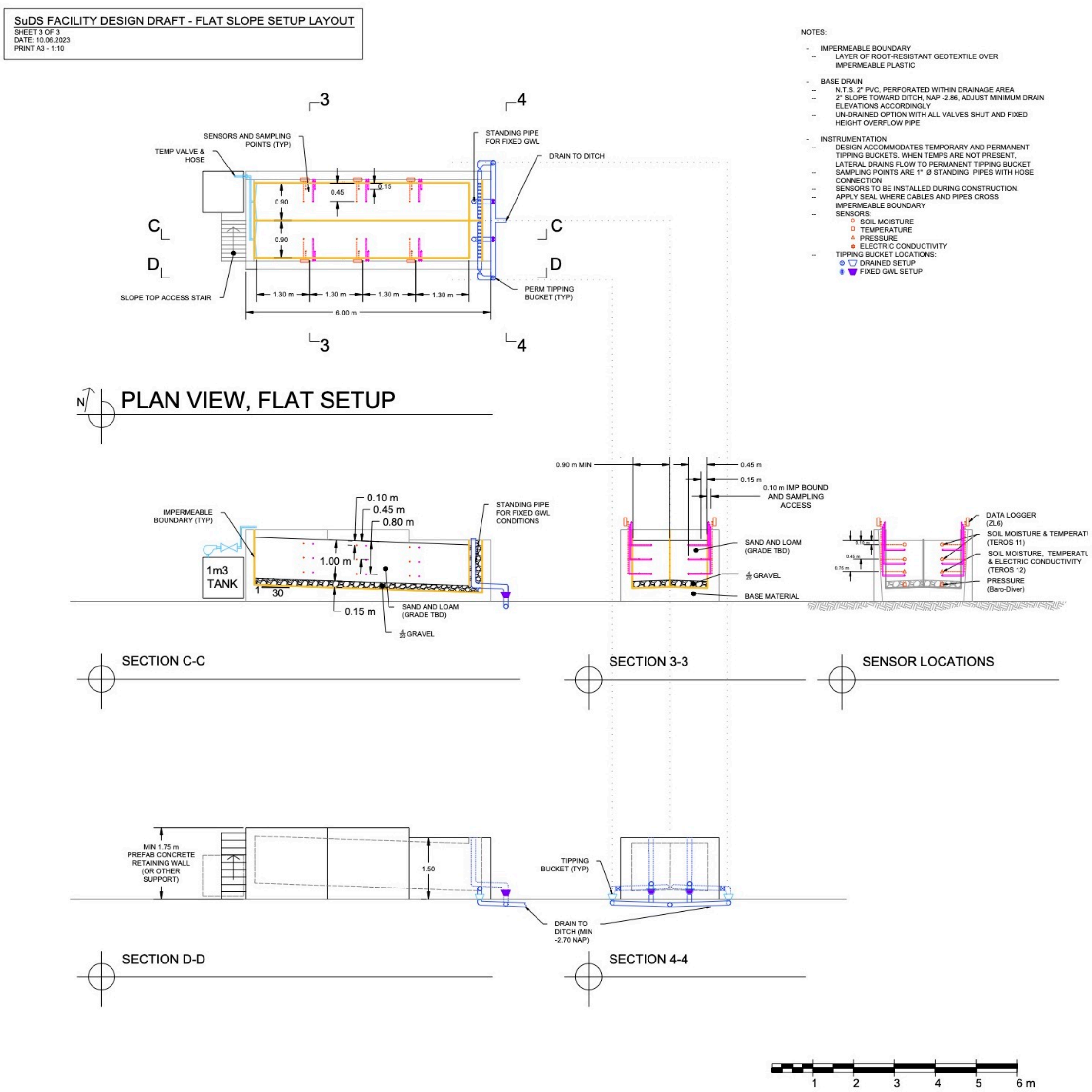


Figure 7.1: Preliminary design for MeSUDa [27]

7.2. Vegetation recommendations for MeSUDa

Two different vegetation types are suggested in order to compare their influence on swale performance. Swale #1 should be seeded with a flower-rich grassland mixture, designed for wet soils (Cruydt Hoeck, *Bloemrijk grasland - natte grond (G3)*). Swales #2 and #3 should be seeded with standard Dutch play grass (MRS Seeds & Mixtures, Allround grass seeds). The flower-rich mixture is recommended because it is identical to the vegetation planted in a monitored site on the TU Delft campus, in a swale located in front of the Echo educational building. By using the same vegetation in a MeSUDa swale, the representativeness of the OAL can eventually be assessed by comparing collected data to that from the Echo swale. Standard play grass is recommended because it is most often employed in Dutch urban swales and therefore serves as the benchmark vegetation.

Table 7.1: Details of swale characteristics and experimental purposes

Swale Number	Layering	Vegetation	Experimental Purpose
Swale #1	Gravel/coarse sand - fine sand - topsoil	Flower-rich grassland (Echo mixture)	Assessment of representativeness of OAL via comparison with Echo swale
Swale #2	Gravel/coarse sand - fine sand - topsoil	Play grass	Benchmark swale
Swale #3	Gravel/coarse sand - fine sand with woodchip layer - topsoil	Play grass	Testing enhanced nitrogen removal via supplementary carbon source (woodchips)

7.3. Sensor purchasing options

Tables 7.2 and 7.3 contain the sensors considered for purchase for water quantity and water quality parameters, respectively. The tables include standard as well as specialised sensors in both the low-cost and industry standard categories.

Table 7.2: Water quantity: low-cost and industry standard sensor options. Selected sensors in **bold.**

Parameter	Low-cost	Industry standard
Soil Moisture	• **Seeed: Industrial Soil Moisture & Temperature & EC Sensor**	• METER: TEROS 11 • **METER: TEROS 12** • METER: TEROS 54 Profile Probe • HOBO: HOBO MX Soil Moisture Sensor
Discharge		• **METER: ECRN-100 Rain Gauge**
Water Level	• **Seeed: Piezoresistive Liquid Level Sensor**	• **Van Essen Instruments: TD-Diver** • Van Essen Instruments: Baro-Diver • METER: HYDROS 21

Table 7.3: Water quality: low-cost and industry standard sensor options. Selected sensors in **bold.**

Parameter	Low-cost	Industry standard
Temperature	• **Seeed: Industrial Soil Moisture & Temperature & EC Sensor**	• METER: TEROS 11 • **METER: TEROS 12** • METER: TEROS 54 Profile Probe • HOBO: HOBO MX Soil Moisture Sensor
EC	• **Seeed: Industrial EC & TDS Sensor** • **Seeed: Industrial Soil Moisture & Temperature & EC Sensor**	• **METER: TEROS 12** • METER: HYDROS 21
TDS	• **Seeed: Industrial EC & TDS Sensor**	
pH	• **Seeed: Industrial pH Sensor**	
DO	• **Seeed: Industrial Dissolved Oxygen Sensor**	• Eureka Water Probes: Dissolved Oxygen Sensor • HOBO: Dissolved Oxygen Sensor
Nitrogen Concentration		• **TriOS Mess- und Datentechnik GmbH: NICO UV photometer**
Redox Potential		• **SWAP Instruments: ORP 80-4-D** • In-Situ: WaterTech Redox8000

7.4. Calibration statuses for purchased sensors

Table 7.4: Calibration categories for each sensor

Sensor	Category	Reasoning
METER: TEROS 12	Manufacturer Calibrated	Pre-calibrated by METER.
Van Essen: TD-Diver	Manufacturer Calibrated	Pre-calibrated by Van Essen.
TriOS: NICO UV Photometer	Manufacturer Calibrated	Pre-calibrated by TriOS.
METER: ECRN-100 Rain Gauge	Field Validation	Manufacturer calibrated, but physical validation desired to ensure function with MeSUDa setup (i.e., allowable flow rate).
Seeed: Soil Moisture/Temp/EC	Validation with Industry Standard	Possible to validate all parameters using the METER TEROS 12 sensor.
Seeed: EC/TDS	Lab Calibration	Featured on board EC auto-calibration for buffer solutions 1413 uS/cm and 12880 uS/cm. TDS remained uncalibrated since was not a parameter of interest for current experiments.
Seeed: Water Level	Field Validation	No calibration settings available, so physical validation was required.
Seeed: pH	Lab Calibration	Featured on board auto-calibration buttons for buffer solutions pH 4.01, pH 7.00, and pH 10.01.
Seeed: DO	Field Validation	No calibration settings available. Field validation with an external sensor (Greisinger DO meter) was possible.
SWAP: Redox Potential	Lab Calibration	Redox probe was not needed for the present research, so remains uncalibrated. Lab calibration is possible for future experimental use.

7.5. Sensor calibration and validation procedures

7.5.1. Seeed EC/TDS Calibration

1. Fill one glass beaker with 60 mL of 1413 uS/cm buffer solution and another with 60 mL of 12880 uS/cm buffer solution.
2. Rinse the electrode of the sensor with distilled water.
3. Submerge the electrode in the solution and wait for approximately 60 seconds, allowing the EC reading to stabilise.
4. Press the sensor calibration button corresponding to the current buffer solution.
5. Once the process is complete (approximately 30 seconds), remove the electrode and rinse with distilled water.
6. Repeat steps 2-4 for both buffer solutions.

7.5.2. METER Rain Gauge Validation

1. Clear the tipping bucket of any debris or dirt that might prevent water flow into the spoons.
2. Fill a 10 mL syringe with water.
3. Very slowly empty the syringe into the funnel of the tipping bucket, as closely to the centre as possible (this prevents water from beading on the inside of the bucket).
4. Continue adding water until one of the spoons tips.
5. Record the amount of water added.
6. Repeat steps 1-5 for a total of 10 times for the bucket, resulting in 10 data points.
7. Repeat the entire procedure for each of the tipping buckets.

7.5.3. Seeed SM/T/EC Validation

1. Fill a 3 L bucket with soil and remove any debris (stones, pieces of wood, etc.). Gently pack the soil into the bucket, ensuring there are no large pockets of air or gaps.
2. Carefully slide the METER sensor and the Seeed sensors into the soil and wait for 60 seconds to allow the sensors to adjust their readings.
3. Record the measurements for each sensor from the corresponding data platform.
4. Remove the sensors.
5. Add 250 mL of water to the bucket using a graduated cylinder and mix until homogenous.
6. Repeat steps 2-5, until soil is saturated and water can no longer be added (begins to pool on top).

7.5.4. Seeed Water Level Validation

1. Fill the pipe with 25 cm of water.
2. Lower the sensor gently into the pipe, until the tip is resting at the bottom.
3. Record the increase in water level due to the addition of the sensor (typically 7-8 cm).
4. Record the sensor water level measurement.
5. Repeat steps 2-4 for the following water heights: 50, 75, and 90 cm.
6. Repeat entire procedure for each water level sensor.

7.6. Hydrological characterisation procedures

7.6.1. Infiltration testing

1. Prepare the Mini Disk Infiltrometer and accompanying Excel workbook according to the METER instruction manual.
2. Fill a bucket with a known volume of material and gently pat down, such that the material is smoothed on top and slightly consolidated.
3. Place the infiltrometer on top of the material and start a stopwatch.
4. Record the volume in the infiltrometer at set time intervals, typically every 10-60 seconds, depending on the speed of infiltration.
5. Continue recording until at least 30 mL of water has infiltrated (as recommended by METER).
6. Consult the Excel workbook for the final unsaturated hydraulic conductivity.

Procedure derived from METER Mini Disk Infiltrometer manual [50].

7.6.2. Salt injection tracer testing

Sensor settings

Sensor	Location	Sampling rate
METER tipping bucket	Outlet	1 point / 5 minutes
METER TEROS 12	Location #1	1 point / 5 minutes
Seeed EC/TDS	Inlet, Outlet	1 point / 5 minutes
Seeed SM/T/EC	Location #2	1 point / 5 minutes

Test procedure

Initial flush

1. Install injection ring 280 cm from the front of the swale, on the centre line, at a depth of 10 cm.
2. Mark desired water level in injection ring using a piece of tape (recommended: 20 cm).
3. Prepare and position peristaltic pump next to MeSUDa, placing one hose into the filled water reservoir and one into the injection ring.
4. Turn on pump at relatively high speed and fill injection ring with water until desired water level is reached. Note current time as official test start time.
5. Record EC of surface water in the injection ring as the baseline flushing level.
6. Adjust pump speed to 7 rpm.
7. Continue to pump water into injection ring, maintaining a constant water level until entire 1000 L reservoir has been emptied.
8. Refill reservoir with surface water, using a second pump if necessary.

Salt injection

1. Switch off the peristaltic pump.
2. Slowly and stir salt into the injection ring until the water has an EC of approximately 10,000 μS/cm.
3. Record the resulting EC of the water inside the injection ring.
4. Switch the pump back on and set to 7 rpm.

5. Pump water into the injection ring continuously, maintaining a constant water level until the EC in the injection returns to baseline (surface water) levels.
6. Refill reservoir with surface water, using a second pump if necessary.

Final flush

1. Continue to pump water into injection ring, maintaining a constant water level until entire 1000 L reservoir has been emptied.
2. Monitor EC values in the swale effluent to ensure that salt pulse has passed through and that values are returning to pre-tracer levels.

7.7. Decomposition assessment tests

7.7.1. DC3 Classification

Table 7.5: DC3 classifications, adopted from Eaton and Sanchez [35].

DC3 Classification	Description
1	wood is sound and solid
2	wood is slightly softened with most of the inner core still hard
3	completely soft and easily crushed

7.7.2. Pick Test

Table 7.6: Pick test decay detection guide as described by Anderson et al. [36]

	Non-decayed	Decayed
Failure mode *appearance of the splinter*	**Fibrous break**: long splinters, breakage extends parallel to the grain beyond the penetration point **Splintering break**: multiple short splinters, breakage occurs at the penetration point	**Brittle break**: very little or no splintering, breaks perpendicular to the grain at the penetration point
Break difficulty *ease of penetration with the screwdriver*	Difficult to penetrate	Easy to penetrate
Break sound *sound that wood makes when splintering*	Audible and clear sound of wood breaking (the “expected” sound of splintering)	Very quiet sound of breaking or completely inaudible

7.7.3. Fungal Identification

Table 7.7: Indicators of different fungi species [37]

Fungi species	Indicators
Brown rot fungi	Brown, cubiform pattern
Soft rot fungi	Cubiform and/or fibrous patterns, expected in humid conditions
White rot fungi	White and spongy, possible dark spots/lines or clear dark outlined “decay zones”

7.8. Surface water quality testing results

7.8.1. IC analysis

Table 7.8: Ion chromatography testing results for two surface water samples, all concentrations in mg/L.

	F	Cl	NO_2	Br	NO_3	PO_4	SO_4	Na	NH_4	K	Mg	Ca
S 1	0.36	59.92	0.46	0.86	0.86	1.17	59.98	41.83	0.88	7.49	13.47	105.70
S 2	0.36	59.86	0.46	0.86	0.84	1.17	59.90	41.82	0.89	7.61	13.46	105.95

7.8.2. TOC analysis

Table 7.9: Total organic carbon testing results for two surface water samples, all concentrations in mg/L.

	NPOC	TN
S 1	22.21	3.75
S 2	22.79	3.83

7.9. Sensor performance indicator results

7.9.1. METER ECRN-100 tipping buckets

Table 7.10: Tipping bucket indicator results.

Indicator	Score	Notes
Cost	3	Cost is high, 967.20 € in total. Purchase of data logger required. Annual payment of 179 € for online data portal required.
Setup/installation	5	Manufacturer calibrated, very straightforward to install.
Usability	5	Settings easy to adjust. Data retrieval and use of the ZENTRA portal simple. Troubleshooting materials available and manufacturer very responsive/helpful to questions.
Reliability	3	Data acquisition is reliable (no issues with sensors or data logger) and manufacturer accuracy is suitable. Software connectivity is reliable. Validation performance was considerably worse than stated accuracy.
Robustness	5	Hardware is durable, no issues with breakage or failure. All components are completely weatherproof and easy to use.
Total Score	**21**	

7.9.2. METER TEROS 12

Table 7.11: TEROS 12 indicator results.

Indicator	Score	Notes
Cost	3	Cost is high, 693.60 € in total. Purchase of data logger required. Annual payment of 179 € for online data portal required.
Setup/installation	5	Manufacturer calibrated, very straightforward to install.
Usability	5	Settings easy to adjust. Data retrieval and use of the ZENTRA portal simple. Troubleshooting materials available and manufacturer very responsive/helpful to questions.
Reliability	5	Reliable data acquisition (no issues with sensors/data logger). Good sensor accuracy and software connectivity. No calibration/validation performance to assess.
Robustness	5	Hardware is durable, no issues with breakage or failure. All components are completely weatherproof and easy to use.
Total Score	**23**	

7.9.3. Seeed SM/T/EC sensor

Table 7.12: Seeed SM/T/EC indicator results.

Indicator	Score	Notes
Cost	5	Low cost, 167.36 € in total. Purchase of data logger required. Data transfer payment required via SIM card.
Setup/installation	5	No simple calibration, but could be validated against industry standard. Straightforward installation.
Usability	3	Sensor settings easy to adjust via online portal. Data download on portal also simple. Not many troubleshooting materials available, manufacturer contact usually required.
Reliability	3	Data acquisition from sensor is reliable and manufacturer accuracy is good. Validation performance was suboptimal. Experienced some issues with data transfer (software connectivity).
Robustness	1	Hardware is easy to use, but is not robust and does not apper to last very long. Connection between cables and data logger failed multiple times, exposing inside of logger to weather.
Total Score	**17**	

7.9.4. Seeed water level sensor

Table 7.13: Seeed water level indicator results.

Indicator	Score	Notes
Cost	5	Low cost, 205.46 € in total. Purchase of data logger required. Data transfer payment required via SIM card.
Setup/installation	5	No simple manual calibration, but could be validated using known water levels. Very straightforward installation.
Usability	3	Sensor settings easy to adjust via online portal. Data download on portal also simple. Not many troubleshooting materials available, manufacturer contact usually required.
Reliability	5	Data acquisition reliable and manufacturer-stated accuracy is suitable. No issues experienced with software connectivity. Excellent performance during validation.
Robustness	1	Sensor itself seems robust enough and easy to use, but connection between cables and data logger was unstable at times. Concerns that if connection does not stay tight, inside of logger will be exposed to weather.
Total Score	**19**	

7.9.5. Seeed EC/TDS sensor

Table 7.14: Seeed EC/TDS indicator results.

Indicator	Score	Notes
Cost	5	Low cost, 215.46 € in total. Purchase of data logger required. Data transfer payment required via SIM card.
Setup/installation	5	Very simple calibration procedure, with on-board calibration buttons. Sensors straightfoward to insall.
Usability	3	Sensor settings easy to adjust via online portal. Data download on portal also simple. Not many troubleshooting materials available, manufacturer contact usually required.
Reliability	3	Data acquisition from sensor is reliable and manufacturer accuracy is good. Experienced some issues with data transfer (software connectivity). No validation to assess, but performance during experiments was suitable.
Robustness	1	Hardware is easy to use, but does not apper to last very long. Connection between cables and data logger failed multiple times, exposing inside of logger to weather.
Total Score	**17**	

7.9.6. van Essen diver

Table 7.15: van Essen diver indicator results.

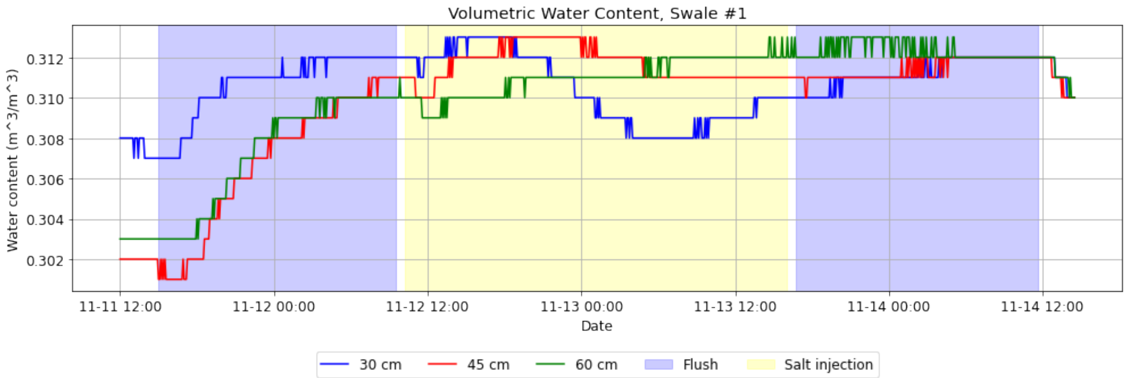
Indicator	Score	Notes
Cost	3	Cost is high, 894.96 € in total. Purchase of additional diver reader accessory required.
Setup/installation	5	Manufacturer calibrated, very straightforward to install.
Usability	3	Diver must be retrieved from setup and connected to computer to change settings and retrieve data. Troubleshooting materials available and manufacturer responsive.
Reliability	5	Reliable data acquisition/transfer and manufacturer accuracy suitable. No calibration or validation performance to assess, but good performance in field.
Robustness	5	Sensor and accompanying hardware is durable and easy to use. No concerns regarding weatherproofing and longevity.
Total Score	**21**	

7.10. Infiltration testing results

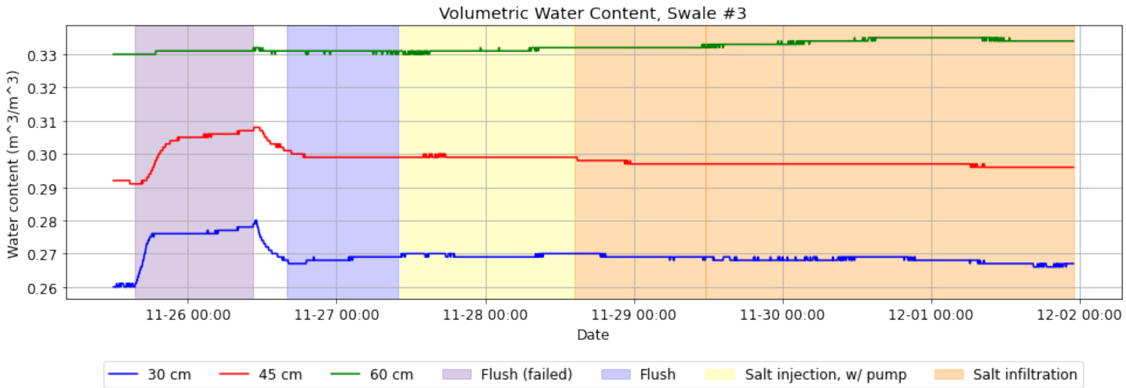
Table 7.16: Unsaturated hydraulic conductivity (K) for fill media. All values in cm/h.

Layer	Mostly dry	Partially saturated
Topsoil	4.32	0.04
Fine sand	10.36	0.27
Coarse sand	115.2	72.85

7.11. Soil moisture measurements, tracer testing



(a) Soil moisture at location #1, swale #1.



(b) Soil moisture at location #1, swale #3.

Figure 7.2: Soil moisture data at location #1 from both tracer tests.

7.12. Qualitative decomposition assessment results

Table 7.17: Decomposition assessment results, swale #3.

Swale #3	Observations	Result
DC3 classification	Slightly softened and not sound-feeling, but quite compact. Class 2.	Minimal decay
Pick test		
Failure mode	Mostly fibrous failure, but few pieces with brittle failure	Minimal decay
Break difficulty	Very difficult to penetrate with pick	No decay
Break sound	Mostly snap when splintering, but few pieces inaudible	Minimal decay
Visual	No clear indications of rot fungi. Difficult to identify colours, but no clear discoloured patches. No distinct patterns.	No decay
General observations	All pieces are very small, smooth, and compact. Soil around extraction point was extremely wet, with dark grey colour and distinct sulphuric smell. 5 different locations in swale drilled for samples, woodchips only found at 2 locations.	

Table 7.18: Decomposition assessment results, control (top) .

Control (top)	Observations	Result
DC3 classification	Almost all pieces are solid and sound. Very small pieces are slightly softened. Class 1.	No decay
Pick test		
Failure mode	Both splintering and fibrous breaks observed. No brittle failure.	No decay
Break difficulty	Very difficult to penetrate with pick	No decay
Break sound	Mostly snap when splintering. Few smaller pieces are slightly quieter. None inaudible.	No decay
Visual	Few pieces have small, white patches or black/dark brown spotting. Difficult to identify, but might indicate one or multiple types of rot fungi. No distinct patterns.	Minimal decay
General observations	Chips were stored in mostly dry environment. Did not clump together when collected, and feel very strong. Chips are fibrous, with distinct and sharp edges.	

Table 7.19: Decomposition assessment results, control (bottom).

Control (bottom)	**Observations**	**Result**
DC3 classification	Some pieces hard and sound, some softed and easy to crush (especially smaller samples). Class 1.5.	Minimal decay
Pick test		
Failure mode	Mostly fibrous failure, but few pieces with brittle failure.	Minimal decay
Break difficulty	Large pieces difficult to penetrate, small pieces easy to penetrate.	Minimal decay
Break sound	Only very large pieces snap when broken. All others very quiet or completely inaudible.	Moderate decay
Visual	Few fuzzy, white patches, and more frequent black spots. Small and thin pieces almost completely blackened. Smaller and softer pieces exhibit cubiform patterns. Indications of white rot fungi, as well as either soft or brown rot fungi.	Moderate decay
General observations	Chips were moist when collected and clumped together. Feel softened and smell earthy. Chips are fibrous.	

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