Fachgebiet Ressourceneffiziente Abwasserbehandlung

Technische Universität Kaiserslautern

Final report
to the Willy-Hager Foundation on the project:
"*Evaluation of a nature-based, resource-efficient process for the treatment of household waste-water streams*"

Runtime: June 2024 until February 2025

FG Ressourceneffiziente Abwasserbehandlung
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Kaiserslautern, Mai 2025.

Summary

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List of Abbreviations

Term	Unit	Description
SSF	–	Specialized Soil Filter
CW	–	Constructed Wetland
CAL	–	Calcium Lactate Extraction Method
$CaCl_2$	–	Calcium Chloride
eff CEC	–	Effective Cation Exchange Capacity
MAP	–	Magnesium Ammonium Phosphate (Struvite)
COD	mg/L	Chemical Oxygen Demand
TN	mg/L	Total Nitrogen
TP	mg/L	Total Phosphorus
OD	mg/L or % saturation	Dissolved Oxygen
NH_4-N	mg/L	Ammonium Nitrogen
NO_3-N	mg/L	Nitrate Nitrogen
NO_2-N	mg/L	Nitrite Nitrogen
PO_4-P	mg/L	Phosphate-Phosphorus
TOC	%	Total Organic Carbon
CEC	$cmol^+/kg$	Cation Exchange Capacity
KBE	KBE/100 mL	Colony-Forming Units (German: "koloniebildende Einheiten")
MPN	MPN/100 mL	Most Probable Number (used for microbial counts)
d_{10}	mm	Particle diameter at 10% cumulative undersize
BET	m^2/g	Brunauer–Emmett–Teller (Surface Area)
kfA	m/s	Permeability Coefficient
EC	µS/cm	Electrical Conductivity
ROS	–	Resource-Oriented Sanitation
RPTU	–	Rheinland-Pfälzische Technische Universität Kaiserslautern-Landau
DBU	–	Deutsche Bundesstiftung Umwelt (German Federal Environmental Foundation)
UFMS	–	Universidade Federal do Mato Grosso do Sul
NTU	NTU	Nephelometric Turbidity Units
VDLUFA	–	German Agricultural Analytic and Research Institutes Association
DIN EN ISO	–	German/European/International Standardization Organization
IWA	–	International Water Association

Kurzfassung

Die vorliegende Studie bewertet die Reinigungsleistung eines dezentralen ressourcenorientierten Sanitärsystems, das in einer ländlichen Gemeinde in Rheinland-Pfalz, Deutschland, implementiert wurde. Das System wurde ursprünglich im Rahmen eines früheren Forschungsprojekts, gefördert durch die Deutsche Bundesstiftung Umwelt (DBU), am ländlichen Standort Reinighof entwickelt und erprobt.

Das im Jahr 2018 für im Durchschnitt 16 Einwohnerwerte ausgelegte System nutzt einen Ansatz der Teilstromtrennung. Grauwasser aus Haushalten und vom Campingplatz wird mittels einer bewachsenen Bodenfilteranlage behandelt, während Urin aus Trockentrenntoiletten in einem Reaktor zur Produktion von Struvit (Magnesium-Ammonium-Phosphat; MAP) verarbeitet wird. Mehr als 90 %, häufig über 99 %, des Phosphors im Urin werden zuverlässig als Struvit zurückgewonnen. Nach der MAP-Fällung fließt der ammoniumreiche Urinüberstand in einen Spezial-Bodenfilter (SBF) – einen unbepflanzten, modifizierten Bodenfilter –, der zwischen dem MAP-Reaktor und dem Bodenfilter zur Grauwasserbehandlung angeordnet ist. Ziel des SBF ist die Reduktion der Ammoniumkonzentrationen auf ein Niveau, das eine nachgeschaltete Mitbehandlung in dem Grauwasser-Bodenfilter ermöglicht. Gemäß den lokalen Grundwasserschutzauflagen erfolgt die Ableitung des gereinigten Abwassers in einen Verdunstungsteich, wodurch ein abflussfreies System („Zero-Discharge“) gewährleistet wird. Fäzes werden einem zweistufigen Kompostierverfahren unterzogen.

Diese Systemkombination fördert die Ressourcengewinnung und verbindet technische mit naturnahen Lösungen. Die gewonnenen Produkte – Kompost und MAP-Dünger – können im Gartenbau und in der Landwirtschaft eingesetzt werden und ersetzen Mineraldünger aus endlichen Rohstoffquellen wie Phosphatgestein.

Beide Filtersysteme, SBF und der Grauwasser-Bodenfilter, waren ursprünglich mit zeolithhaltigem Lavasand (0-4 mm) befüllt. Dieses Material weist eine außergewöhnlich hohe Kationenaustauschkapazität von 50–70 $cmol^+/kg$ auf. Laut dem früheren Projektbericht [4] zeigte der SBF kurz nach der Inbetriebnahme des eine Reinigungseffizienz von über 99 % für NH_4-N, noch bevor sich nitrifizierende Bakterien etablieren konnten. Im weiteren Verlauf erreichte die NH_4-N-Elimination im SBF über 75 %, während die Reinigungsleistung des Grauwasser-Bodenfilters hinsichtlich des Ammoniumstickstoffs (NH_4-N) zwischen 92 und 99 % lag [1].

Die Filtersysteme sind seit über sechs Jahren in Betrieb und obwohl es ursprünglich auf 16 Nutzer täglich ausgelegt waren, versorgen sie nun saisonal eine wachsende Anzahl von Nutzern auf einem Campingplatz, die in den Sommermonaten Spitzenwerte von 80–100 Personen erreicht. Im Mai 2024 führte eine Systemüberlastung zu einer irreversiblen Kolmation und damit zum Überstau im SBF, was einen vollständigen Austausch des Filtermaterials erforderlich machte. Da der ursprünglich verbaute Lavasand nicht mehr verfügbar war (dauerhafte Schließung des Steinbruchs), wurde ein neuer Lavasand mit unbekanntem Zeolithgehalt aus einem anderen Steinbruch verwendet.

Im Rahmen dieser Studie wird seit Juli 2024 die Leistungsfähigkeit des ertüchtigten SBF mit dem neuen Lavasand sowie die des Gesamtsystems untersucht. Ein weiterer Schwerpunkt liegt auf der Untersuchung der Ursachen für die Filterkolmation im SBF und der Entwicklung von Optimierungsstrategien für Wartung und Betrieb, um das zukünftige Kolmationsrisiko zu minimieren.

Das System zeigte während des Untersuchungszeitraums von Juli 2024 bis Januar 2025 hohe Eliminationsraten für verschiedene Parameter. Gesamtstickstoff (N_{ges}) wurde von bis zu 5.420 mg/L im Urinüberstand auf eine durchschnittliche Konzentration von 126 mg/L im Ablauf des Grauwasser-Bodenfilters reduziert. Ammoniumstickstoff wurde von Werten bis zu 4.774 mg/L im Urinüberstand auf 0,7 mg/L im Ablauf des Grauwasser-Bodenfilters reduziert. Die Gesamtphosphorkonzentrationen (P_{ges}), die aufgrund der vorgeschalteten Struvitfällung bereits niedrig lagen, fielen von im Durchschnitt 11,4 mg/L im Urinüberstand auf 0,09 mg/L im Ablauf des Grauwasser-Bodenfilters. Der Chemische Sauerstoffbedarf (CSB) wurde im gesamten System konsistent reduziert, von etwa 5.046 mg/L im Urinüberstand auf 17,4 mg/L im Ablauf des Grauwasser-Bodenfilters. In Verdunstungsteich lag die Konzentration des Nitratstickstoffs (NO_3-N) durchschnittlich bei 9 mg/L, während die Konzentration des Nitritstickstoffs (NO_2-N) konstant niedrig blieb, meist unter 0,1 mg/L. Die Reduktion der pathogenen Keime war ebenfalls signifikant, dabei sanken die Konzentrationen von *E. coli* und Enterokokken von $2,3 \times 10^5$ KBE/100 ml bzw. $2,3 \times 10^4$ KBE/100 ml im Zulauf des SBF auf < 60 KBE/100 ml im Ablauf des Grauwasser-Bodenfilters.

Die Forschungsarbeiten erfolgten in Kooperation mit der Universidade Federal do Mato Grosso do Sul (UFMS) in Brasilien. Eine Doktorandin evaluierte in diesem Rahmen die Reinigungsleistung des Systems und dessen Übertragbarkeit auf Brasilien und andere Entwicklungsregionen. Die Studie liefert Erkenntnisse zur Robustheit dezentraler Abwasserbehandlungslösungen und unterstützt nachhaltige Sanitärstrategien für ländliche Gebiete sowie Regionen ohne Anschluss an die zentrale Kanalisation.

Abstract

This study evaluates the performance of a decentralized resource-oriented sanitation system (ROS) implemented in a rural community in Rhineland-Palatinate, Germany. The system was originally developed and tested at Reinighof, a rural site, as part of a previous research project funded by the Deutsche Bundesstiftung Umwelt (DBU).

Designed in 2018 to accommodate an average of 16 users daily, the system employs a multi-stream separation approach. Greywater from households and the camping site is treated using a constructed wetland, while urine from urine-diverting dry toilets is processed in a reactor to produce struvite (magnesium-ammonium-phosphate; MAP). More than 90%, and often over 99%, of the phosphorus in the urine are reliably recovered as struvite. After MAP precipitation, the treated urine supernatant, which contains high ammonium concentrations, flows into a specialized soil filter (SSF) – an unplanted modified constructed wetland – placed between the MAP reactor and the constructed wetland for greywater treatment (CW). The goal of the SSF is to reduce the ammonium concentrations to a level that would allow subsequent co-treatment in the constructed wetland for greywater treatment. To address local groundwater protection regulations, the treated effluent is managed through an evaporation pond, ensuring a zero-discharge system. Feces are managed through a two-stage composting process, including rapid in-vessel composting followed by open-air post-composting.

This combination facilitates resource recovery while blending technical with nature-based solutions. The recovered products, compost, and MAP fertilizer, can be used in gardening and agriculture, replacing mineral fertilizers derived from finite resources such as phosphate rock.

Both systems, the SSF and CW, were originally filled with a 0-4 mm zeolite-containing lava sand filter layer. This material has an exceptionally high cation exchange capacity of 50 – 70 cmol+/g, and according to the former project report [1], the SSF demonstrated a treatment efficiency of over 99% for NH_4-N at the beginning of the process. At this time, nitrifying bacteria were not yet established. As the project progressed, the SSF´s purification efficiency for NH_4-N exceeded 75%. In its turn, the treatment performance of the CW with respect to NH_4-N, demonstrated an efficiency of 92 to 99% [1].

After those previous results, the system has been in operation for over six years and, even though it was built for 16 users daily, it accommodates now seasonally a growing number of users at a camping site, which peaked at 80–100 people during summer months. In May 2024, system overload resulted in irreversible clogging and impoundment of the sand layer in the SSF, necessitating its full replacement. The former lava sand is not available anymore, as the mining site that it had been extracted from was permanently closed, and therefore a new lava sand from another quarry was employed. Its zeolite content is not known.

Since July 2024, this study has focused on evaluating the performance of the refurbished SSF with a different lava sand as filter layer as well as the entire system performance, diagnosing the causes of the previous filter's clogging, and proposing maintenance and operational optimizations to prevent recurrence of similar issues.

The system demonstrated high removal efficiencies across various parameters during the monitoring period from July 2024 to January 2025. Total nitrogen (TN) was reduced from up to 5420 mg/L in the urine supernatant to an average concentration of 126 mg/L at the outlet of the constructed

wetland. Ammonium nitrogen (NH_4-N) was reduced from values as high as 4,774 mg/L in the urine supernatant to 0.7 mg/L in constructed wetland effluent. Total phosphorus (TP) concentrations, already low due to prior struvite recovery, dropped from 11.4 mg/L in the urine supernatant to an average value of 0.09 mg/L in the final effluent. Chemical Oxygen Demand (COD) was consistently reduced across the system, with concentrations around 5046 mg/L in the urine supernatant to 17.4 mg/L in the effluent of the CW. At the final disposal stage, the evaporation pond, Nitrate nitrogen (NO_3-N) averaged 9 mg/L, and nitrite nitrogen (NO_2-N) remained consistently low, mostly below 0.1 mg/L. Pathogen reduction was also significant, with *E. coli* and Enterococci levels decreasing from 2,30E+05 MPN/100 ml and 2,30E+04 MPN/100 ml at the inlet of the SSF to < 60 MPN/100 ml at the outlet of the CW.

The research was conducted in collaboration with the Federal University of Mato Grosso do Sul (UFMS) in Brazil, where a PhD. candidate evaluated the system's functionality and potential applicability in Brazil and other developing regions. The study aims to generate insights into the robustness of decentralized wastewater treatment solutions and support sustainable sanitation strategies in rural and off-grid areas.

1. Introduction

Rural areas face significant challenges in implementing effective sanitation systems. Traditional centralized wastewater treatment infrastructures are often economically unfeasible and logistically unsuited for sparsely populated regions, especially where topography, distance, or lack of connection to sewer networks complicates access. Additionally, transporting waste to centralized facilities consumes energy and increases greenhouse gas emissions, making such systems unsustainable in the long term (Xiao et al., 2021).

In response, resource-oriented sanitation (ROS) systems offer a more flexible, sustainable alternative. These systems are built on the principles of decentralization, low-energy treatment, and resource recovery, where wastewater streams are treated according to their specific characteristics. The separation of flows at the source -such as urine, feces, and greywater - facilitates targeted treatment and enables the recovery of valuable components like nitrogen, phosphorus, water, and organic matter (Simha & Ganesapillai, 2017). This is particularly relevant in the context of global nutrient scarcity, where phosphorus is a critical, non-renewable resource.

Urine-diverting systems allow for concentrated nutrient recovery from urine, and struvite precipitation ($MgNH_4PO_4 \cdot 6H_2O$) is a widely studied and implemented method for phosphorus recovery. Struvite is a slow-release fertilizer and can be produced with relatively simple technology. However, while most of the phosphorus can be recovered, the resulting nitrogen-rich supernatant, particularly high in ammonium, still requires further treatment to prevent environmental pollution. This has prompted the integration of additional treatment steps such as special soil filters (SSF) containing zeolite-rich lava sand and constructed wetlands (CW) to support nitrification and pathogen reduction (Bruch et al., 2011).

Despite these advances, ROS systems often face operational barriers when implemented in remote communities. Many nutrient recovery technologies are designed for centralized applications and require complex controls, skilled operators, or consistent electricity. Thus, there is a strong need for systems that are robust, low-tech, and operable by non-experts.

To address this need, a self-sufficient, decentralized wastewater treatment and nutrient recovery system was implemented and monitored at Reinighof, Germany. The system integrates urine-diverting dry toilets, a struvite precipitation reactor, a zeolite-based SSF, and a constructed wetland for final polishing—designed to treat wastewater locally while enabling high levels of nutrient recovery and water treatment (Figure 1). A first report addressing the system's implementation and its evaluation during the first years is archived at the DBU library [1]. This document served as a reference framework for the comparative analysis conducted in the current research.

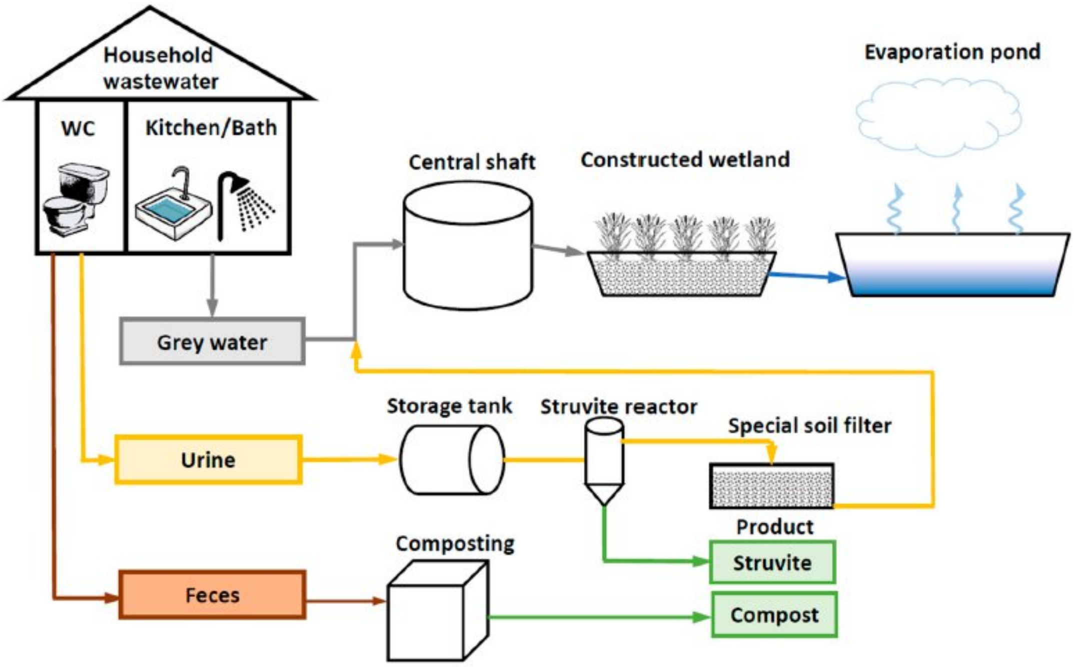

Figure 1: Concept of wastewater treatment and resource recovery system at Reinighof

The following report evaluates the treatment performance, operational stability, and environmental relevance of the system under real-life conditions after six years of operation, with a focus on its applicability to similarly rural and off-grid contexts, such as in rural Brazil.

2. Objectives

The project aimed to develop insights into the system's efficiency, adaptability, and maintenance needs with the following key objectives:

- Evaluation of the overall system operation after more than six years of operation to assess the robustness and stability of treatment performance and soil parameters (pH, conductivity, etc.).
- Evaluation of operation under different loading conditions (with and without shock loading due to campsite occupancy) and at different temperatures (summer, fall/winter).
- Investigation of the former clogged filter media from the specialized soil filter (SSF) after shutdown, as well as to analyze the properties of the newly installed filter media upon refilling of the SSF.

- Assessment of the applicability of the specialized soil filter and the overall system for self-sufficient wastewater disposal in Brazil and other developing emerging countries.
- Development of optimization strategies to enhance system resilience and ensure long-term efficiency.

3. Methodology and Work Program – Concept, Planning, and Design of the Experimental Set-up

The methodology for this research involved an eight-month comprehensive sampling and analysis phase to evaluate the performance and resilience of the decentralized wastewater treatment system; a simplified project timetable is given in Table 1. During the initial month, the project focused on finalizing the sampling methodology while incorporating the predefined analytical parameters and investigation framework. The sampling campaign was structured into three distinct phases. The first phase comprised an intensive five-day sampling during summer, focusing on system performance under high temperatures and peak occupancy conditions. The second phase spanned from July 2024 to January 2025, with biweekly sampling to capture a broader range of operational scenarios and trends. The third phase involved another intensive five-day sampling conducted in winter, assessing the system under reduced usage and more challenging environmental conditions. This phased approach ensured a comprehensive evaluation of the system's performance across diverse seasonal and occupancy contexts.

Systematic data collection will target critical points in the system, including the inflow and outflow of the specialized soil filter, the inflow and outflow of the greywater soil filter, and the evaporation pond.

Table 1: Timetable of the overall project; year of 2024.

	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Jan
Planning of measurement campaigns and experiments								
Summer intensive sampling phase								
Biweekly sampling								
Lab analyses								
Winter intensive sampling phase								
Data processing and evaluation								

Between July 2024 and January 2025, a total of 13 samples were collected from the inlet and outlet of the specialized soil filter (SSF) respectively, while from inlet and outlet of CW and the pond, 21 samples were obtained from each point. The discrepancy in the number of samples is related to the system's varying feeding patterns. The SSF operates intermittently, receiving inflow only when struvite precipitation events take place, which occur at irregular intervals ranging from a minimum of one day to a maximum of approximately 15 days between which feeding. After struvite precipitation, a nitrogen-rich effluent is formed, and a fixed volume of 140 L is introduced into the system in a single feeding event. Urine supernatant floods the filter surface, flows vertically through the SSF and is drained into the control shaft, where the outlet sample is collected. Consequently, the SSF has fewer recorded measurements compared to the CW, which operates under a more continuous flow regime and can be fed a few times a day.

Despite operating under its own regime, the flow within the constructed wetland (CW) is also influenced by the intermittent discharge from the specialized soil filter (SSF). Since the SSF feeds into the CW after each MAP precipitation event, fluctuations in SSF inflow directly impact the hydraulic and treatment dynamics of the CW. From the control shaft, the pre-treated urine supernatant flows into a pump shaft, where it is mixed with greywater from the camping site. This mixture of nitrified urine and greywater is then pumped into a multi-chamber pit, which serves as preliminary treatment stage for greywater. In this multi-chamber pit, greywater from the main house is also mixed and high concentrations from SSF effluent diluted, as it has a volume of 9.3m³. Dissolved Oxygen (OD) measurements indicate that the multi-chamber pit maintains anoxic conditions, which promote denitrification before the effluent reaches the CW.

The CW receives approximately 300 liters of influent per loading event, distributed over an area of 32 m², resulting in a hydraulic loading rate of 9.4 L/m² per event. This influent comprises a mixture of effluent from the SSF treatment unit and greywater generated by the main residence and the camping facility. The system operated under a non-continuous, volume-triggered feeding regime. Specifically, inflow to the CW is initiated when the volume in the multi chamber pit reaches a predefined threshold thereby activating a pump and opening the inlet valve to discharge into the wetland. Consequently, influent loading is irregular and typically increases during the summer months. The treated effluent from the CW then flows by gravity into a second control shaft before being discharged into the pond.

Therefore, the system dynamics are influenced by MAP precipitation events, feeding regimes, climatic factors such as local precipitation and evaporation, distinct loads and the hydraulic retention time between system nodes. As a result, the system exhibits a wide range of values.

On sampling days, pH, turbidity and EC of the five samples were determined onsite by a portable conductivity meter (WTW Cond 3110, Weilheim, Germany) and a portable pH meter (WTW pH 323, Germany). Concentrations of total phosphorus (TP), total Nitrogen (TN), Chemical Oxygen Demand (COD), phosphate-phosphorus (PO_4-P), nitrate-nitrogen (NO_3-N), nitrite-nitrogen (NO_2-N), Chloride (Cl^-) and Sulfate (SO_4^{2-}) were carried out in the lab facilities of the Rheinland-Pfälzische Technische Universität Kaiserslautern-Landau (RPTU) and measured using a Hach DR 3600 photo spectrometer (Germany) with cuvette tests. Magnesium (Mg^{2+}), calcium (Ca^{2+}), potassium (K^+), sodium (Na^+) and ammonium-nitrogen (NH_4-N) were analyzed with ion chromatography (Metrohm 930 Compact IC Flex, with column Metrosep RP 2 Guard, Swiss). Analyses of TN and TP were conducted directly from unfiltered homogeneous samples after chemical-thermal digestion in a thermostat LT200 (Hach Lang GmbH, Düsselford, Germany). Samples for dissolved parameters were filtered with a 0.45m Minisart RC syringe pre-filter (Sartorius AG, G6ottingen, Germany). OD was measured from the multi-chamber pit by a portable Multi-Parameter (w7w Multi 3320, Weilheim, Germany).

Enterococcus, *Escherichia coli* and Coliforms were determined by standard methods (DIN EN ISO 7899-1: 1999-07, DIN EN ISO 9308-3: 1999-07, DIN EN ISO 9308-2: 2014-06), conducted by the partner Laboratory AGROLAB Umwelt GmbH, Dr.-Hell-Str. 6, 24107 Kiel, Germany.

Considering the soil analyses, four samples were analyzed for their composition: (1) the top layer of the former clogged filter media (zeolite-containing lava sand 0-4 mm from Lissingen, Germany), (2) the bottom layer of the former clogged filter media, (3) the solid deposit formed within the supply pipe to the former SSF, and (4) the new filter media (lava sand 0-4 mm from Herchenberg, Germany), with which the system was refurbished in May 2024. BET surface area, a material's total accessible surface area per unit mass via gas adsorption, as well as the particle size distribution of the filter sand samples were analysed by ZetA Partikelanalytik GmbH, D-55129 Mainz. Sample 3 (pipe deposit) consisted of a solid, sintered mass; therefore, to obtain measurable material, it was necessary to partially grind the sample using a kitchen sieve. For sampling (1), (2) and (3), approximately 2 g of the ground material was transferred into a 25 ml Erlenmeyer flask, mixed with 10 ml of water, and treated in an ultrasonic bath for 5 minutes. After sample conditioning, parameters were determined by the Fraunhofer Method, a laser diffraction technique used to measure particle size distribution in suspensions.

The pH ($CaCl_2$), phosphor (P_2O_5), potassium (K_2O) and magnesium (Mg^{2+}) were determined by standard methods (VDLUFA Methodenbuch Band I, A 5.1.1 2016-01 pH ($CaCl_2$), VDLUFA Methodenbuch Band I, A 6.2.1.1 2012-01 Phosphor (P_2O_5) in CAL, potassium (K_2O) in CAL, VDLUFA Methodenbuch Band I, A 6.2.4.1 1991-01 Magnesium (Mg) in $CaCl_2$, DIN ISO 11260 2018-11) by the partner laboratory Agrolab LUFA GmbH, Dr.-Hell-Straße 6, 24107 Kiel, Germany.

4. Operation and Evaluation of the Specialized Soil Filter (SSF)

It is essential to investigate the SSF separately, as this system is the most vulnerable component when compared to the overall treatment system. This vulnerability arises because the SSF receives an influent with an extremely high nitrogen concentration, reaching up to 5.4 g/L TN in the study period. Before, after five years of operation, the filter medium became irreversibly clogged and had to be replaced.

It is important to highlight that the new filter medium, analyzed between July 2024 and January 2025, was only introduced into the system in May 2024. Therefore, it had been in operation for just two months when the monitoring period began. The newly implemented filter material contained a higher proportion of fine particles compared to the previous medium. The former filter media was a Lavasand from Lissingen, Germany, that contained around 10% of natural zeolite and a BET specific surface area of 80-100 m²/g. The current filter media, in turn, is a lava sand from Herchenberg, Germany, with an unknown zeolite content, while the BET specific surface area reaches 40 m²/g (see Table 2 in chapter 6).

Since the system has been reactivated in May 2024 and the analyses began in July 2024, the filter was still undergoing a start-up phase where fine particles were gradually being washed out of the system. This phenomenon was visibly reflected in both the influent and effluent samples, as show in Figure 2; these pictures were taken in July 2024, October 2024 (when the first signs of clear outlet appeared) and January 2025.

In the initial stages of operation, an increased presence of suspended solids in the effluent was observed, originating from the finer particles being carried out of the SSF over time, as these particles were gradually removed, the effluent quality improved, indicating the stabilization of the filter media. This trend was also noticeable in the turbidity analyses, which decreased significantly from 409 NTU in September 2024 to 13.2 NTU in January 2025.

Figure 2: Color differences between Inlet (Zu) and Outlet (Ab) of the SSF from July 2024 to January 2025

Another important factor influencing system performance in the initial months is the gradual development of bacterial biofilms within the filter media. In biological filtration systems, nitrifying bacteria requires time to colonize the surface of the filter material and establish a stable microbial community.

Despite the expectation that nitrification is less stable in colder months due to lower temperatures, low ammonium concentrations in the outlet were also observed during winter. This trend suggests that seasonal variations in the feeding regime may have played a role. During summer, the system was fed much more frequently, with struvite precipitation occurring weekly or even twice a week, corresponding to 80–100 people at the camping site. In contrast, during winter, there were periods of up to two weeks without precipitation, potentially allowing for microbial adaptation to different nutrient availability cycles. Additionally, the microbial community may have had a fully developed by that time, enabling more stable ammonium nitrogen conversion to nitrate nitrogen. These factors collectively highlight the dynamic nature of biological processes within the system and the importance of long-term monitoring to understand performance fluctuations.

The exact number of feeding events during the year is unknown. However, based on community habits, it can be estimated that the system is fed approximately 40 times annually. This estimate considers weekly feeding during summer, biweekly feeding in winter, and feeding approximately every 1.5 weeks during spring and autumn. The hydraulic load per feeding event was therefore always the same. However, when converted to a mean hydraulic load per day, considerable differences arise due to the varying time intervals between feeding events. Based on an annual total of 6400 L distributed across 40 irregular feeding events to a 4m² filter area, the system receives an estimated hydraulic loading rate (HLR) of 4.38 mm/day, well below the 80 mm/d recommended for pretreated wastewater according to DWA A 262.

The SSF consistently received influent with very high ammonium nitrogen (NH_4-N) concentrations, reaching up to 4800 mg/L, and reduced them to between 280 and 1700 mg/L, yielding an average removal efficiency of 83.5%. Total nitrogen (TN) was partially reduced from initial values of

3018–5420 mg/L to 1839–3360 mg/L, while Chemical Oxygen Demand (COD) was lowered by 36–83%, depending on operational conditions. The SSF also exhibited significant nitrification activity, converting NH_4^+ into nitrite (NO_2^-) and nitrate (NO_3^-) — with NO_2^- peaking at 410 mg/L and NO_3^- exceeding 1400 mg/L, indicating intense microbial oxidation processes. In addition to biological transformation, the SSF's performance was influenced by adsorption through the zeolite-containing lava sand filter media, which offered high cation exchange capacity and facilitated ammonium retention. Over time, denitrification likely occurred in downstream components, but some contribution within the SSF can be inferred during longer retention periods or when oxygen levels decreased. The stabilization of effluent quality over the months, alongside the observed decrease in turbidity from 409 NTU to 13.2 NTU, reflected the gradual maturation of microbial biofilms and physical filtration as fine particles were flushed out. This multi-process treatment mechanism—adsorption, nitrification, and partial denitrification—enabled the SSF to effectively reduce nutrient loads before discharging into the downstream constructed wetland (CW), despite variations in hydraulic loading and seasonal operation. Further details and discussions on the system's integrated performance are provided in the following chapter on the overall treatment concept and operation.

5. Operation and Evaluation of the Overall Treatment Concept

This chapter evaluates the performance of both SF and CW together concerning concentrations of Nitrogen, Phosphorus, COD and parameters like pH and Temperature. The values described in this section represent mean concentrations obtained from the analysis.

Initial total nitrogen concentrations are over 5000 mg/L in urine, while the effluent of the CW had an average concentration of TN of 10.7 mg/L at the end of the treatment process. Phosphorus removal was also highly effective, dropped from 11.4 mg/L at the inlet of the SSF to below 0.05 mg/L in the final effluent of the CW. The reduction in turbidity from an initial 67.3 NTU at the SSF influent to 1.73 NTU in the final effluent of the CW, indicating a significant reduction in the suspended soil content. Table 3 in the Appendix presents the average, minimum and maximum values of all parameters for the five analyzed samples.

Considering that the system was originally designed for 16 users but has operated for three years with peak loads of up to 80 people during the summer, it demonstrates a capacity to handle significantly higher usage than intended. The two-step treatment enhances this resilience by directing the high nitrogen load to the SSF, preventing excessive strain on the entire system. Although the volume flow entering the SSF is relatively small compared to the total greywater volume flow, it was crucial to prevent the greywater soil filter from experiencing periodic nitrogen shock loads with extremely high ammonium concentrations, reaching several thousand mg/L.

5.1 Evaluation of the measurement data

5.1.1 – pH

pH values tended to decrease as the wastewater progresses through the system, ultimately stabilizing at the CW outlet. The observed pH variations between the SSF and CW systems suggest differences in the extent and efficiency of nitrification (Figure 3). The SSF influent exhibits a relatively high pH (9.2-10.0) - due to the hydrolysis of urine – which is above the optimal range for nitrification, typically ranging from 7.0 to 8.5 [5]. However, the pH decrease observed at the outlet of the SSF (7.1 – 8.2) suggests that acidifying process, including nitrification, are occurring.

In contrast, the CW inlet shows a more moderate pH (7.6 – 8.0), which falls within the optimal range for denitrification, a process likely occurring in the multi-chamber pit located downstream of the CW, where the CW inlet samples were collected. The further pH reduction at the CW effluent (6.7 – 7.1) suggests increased nitrification and acidification, also through e.g. root exudation, within the CW.

Although the initial pH in the inlet of the SSF was above the optimal range for nitrification, it exhibited a greater pH reduction compared to the CW. This can be explained by nitrification of the higher concentration of ammonium ions (NH_4^+), eliminated to a wide extend within the SSF (see chapter 5.1.3) via nitrification (see chapter 5.1.3 and 5.1.5) consuming significant alkalinity. The pond system maintains a stable pH of 7.4 –8.1.

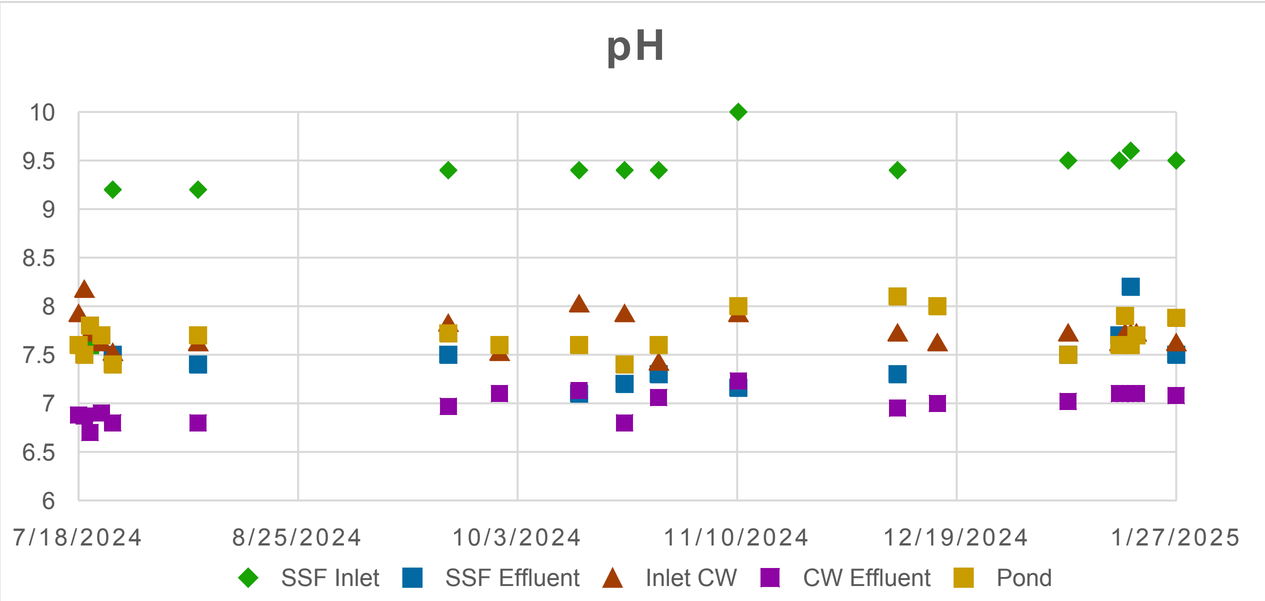

Figure 3: pH values at the five different stages of the treatment system over time.

5.1.2 – Chemical Oxygen Demand (COD)

The COD removal efficiency of the SSF varies between 36% and 83%, with an average above 60%. Efficiency fluctuates due to variations in COD inlet concentrations and operational modifications. The CW demonstrates consistently high COD removal, with efficiencies ranging between 78% and 95% (Figure 4). Even when SSF performance declines, the CW continues to perform well, highlighting its role as a robust secondary treatment stage.

COD concentration in the pond varies between 13.5 mg/L and 51.1 mg/L, with clear seasonal differences trend over time. The highest values were observed in the early sampling dates, between July and September, followed by a decreasing trend. COD values in the pond remain higher than at the outlet of CW, which can be due to weather conditions; during the summer months, higher temperatures and increased sunlight promote algae and plant growth in the pond. This proliferation can lead to higher organic matter production and, consequently, elevated COD values due to the accumulation of biomass and organic compounds.

Chemical Oxygen Demand (COD) levels exhibited a decrease from 5046 ± 802 mg/L at the inlet of the SSF to 17.4 ± 7.0 mg/L in the effluent of the CW, proving the robustness of the overall treatment consisting of struvite precipitation, SSF and CW.

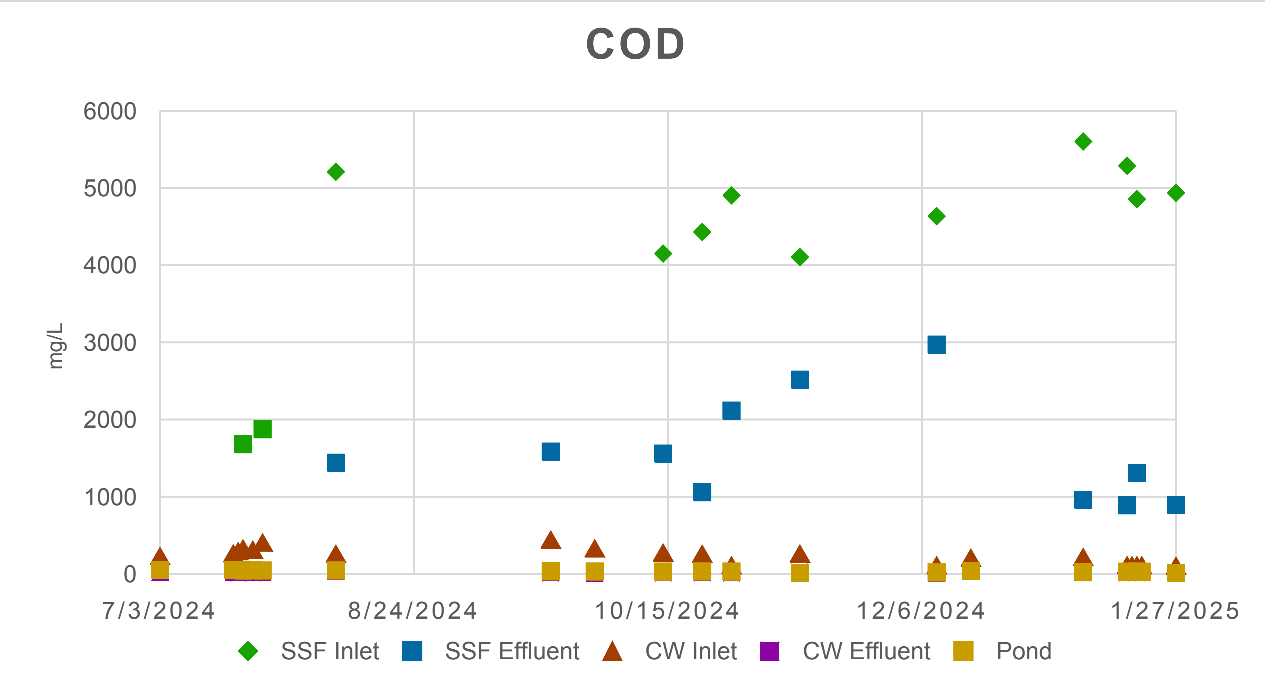

Figure 4: COD values at the five different stages of the treatment system over time.

5.1.3 – Ammonium Nitrogen (NH_4-N)

The NH_4-N concentration in the urine supernatant, and consequently at the SSF inlet, was notably high, ranging between 2.9 and 4.8 g/L, which is depicted in Figure 5, where a base-10 logarithmic scale was applied to the vertical axis due to wide disparity in magnitude between data series.

The ammonium nitrogen concentration corresponds to a specific surface load of approximately 101.5 to 168 g NH_4-N/m² filter surface per feeding event. That gives an approximately ammonium nitrogen load of 0.3 – 0.5 g NH_4-N/(m² d), considering 40 feedings a year.

Despite a substantial reduction of ammonium-nitrogen within the SSF, the effluent from the SSF still contained NH_4-N concentrations ranging from 0.28 to 1.4 g/L.

At the CW inlet, NH_4-N concentrations dropped significantly to 44 ± 23 mg/L, possible due to dilution of the SSF effluent thorough the greywater in the multi chamber pit. Subsequent treatment in the CW further reduces NH4-N levels to 0.7 ± 0.4 mg/L, demonstrating a highly effective NH_4-N removal at this stage.

The removal efficiencies were on average 83.5% at the SSF and 98.1% at the CW, in terms of NH4-N, being highly effective, nearly eliminating ammonium-nitrogen before the CW effluent is conveyed to the pond.

The pond showed NH_4-N concentrations ranging from 0.08 mg/L to 0.74 mg/L, remaining relatively stable over time, with an average of 0.27 ± 0.1 mg/L NH_4-N.

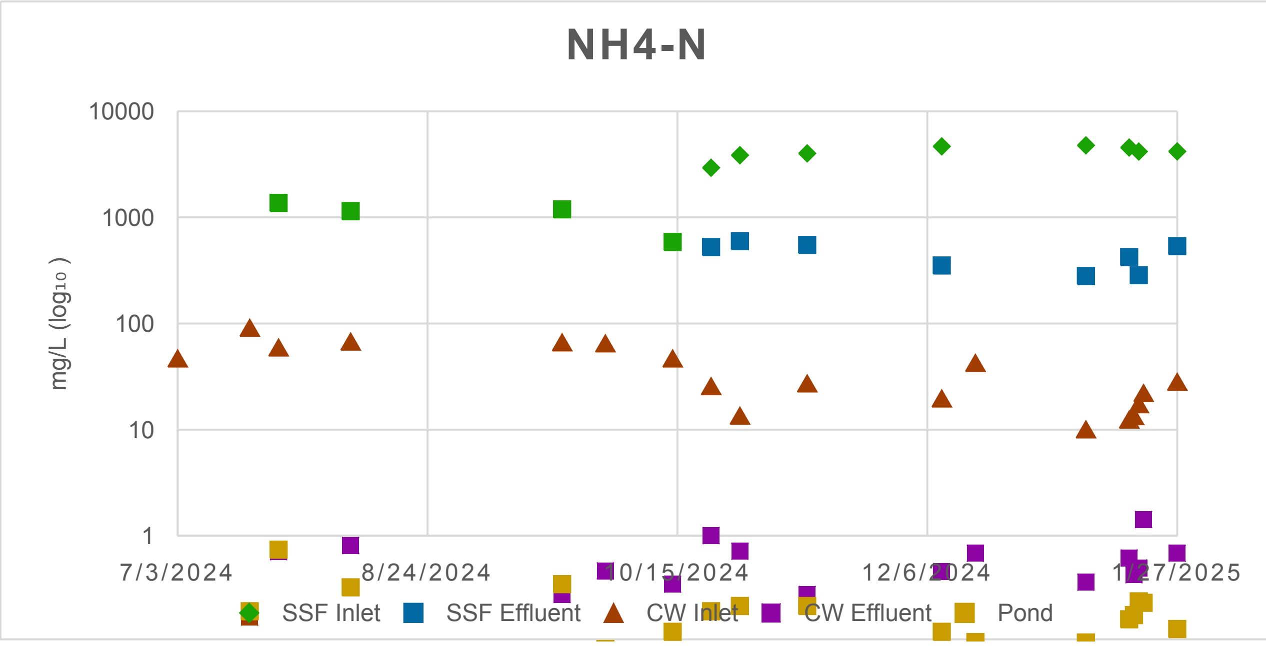

Figure 5: NH4-N values at the five different stages of the treatment system over time, plotted on a base-10 logarithmic scale.

5.1.4 – Nitrite Nitrogen (NO_2-N)

The NO_2-N concentration at the inlet of the SSF was found to be quite stable, with an average of 0.2 mg/L. The SSF outlet, in turn, exhibited high nitrite levels, around 410 mg/L NO_2-N on average, particularly in the summer months (Figure 6). This suggests that the SSF experienced incomplete nitrification or transient system overloads during peak operation. Over time, concentrations at the SSF outlet decreased, especially during winter, with values ranging between 10.4 mg/L and 20 mg/L, although it is known that the activity and growth rates of both Ammonia-Oxidizing Bacteria (responsible for nitritation) and Nitrite-Oxidizing Bacteria (responsible for nitratation) decrease significantly under cold temperature effects. However, the significantly lower nitrite effluent values during colder months align well with improved NH_4-N removal rates in winter, suggesting enhanced system stability and microbial adaptation in winter. This is due to the influence of a significantly lower nitrogen load during this season, as much less urine is produced and subsequently treated in the winter months, relieving the SSF greatly.

However, the downstream constructed wetland (CW) showed a high NO_2-N removal capacity, as demonstrated by the consistently low concentrations at the CW outlet. Most CW effluent values remained below 0.2 mg/L NO_2-N, indicating that any accumulated nitrite from the SSF was efficiently converted to NO_3-N. This can be attributed to the dilution effect of pre-treated urine in greywater, as the SSF supernatant is redirected to the multi-chamber pit, and the favorable anaerobic conditions inside it that promotes denitrification as the aerobic conditions within the CW, promoting nitrification.

The final NO_2-N concentration within the pond remained consistently low, mostly below 0.1 mg/L, indicating no risk of nitrite toxicity within the pond. This confirms the system's ability to effectively prevent nitrite accumulation, ensuring compliance with environmental standards and protecting downstream water quality.

Due to the wide disparity in magnitude between the data series, a base-10 logarithmic scale was applied to the vertical axis of the diagram.

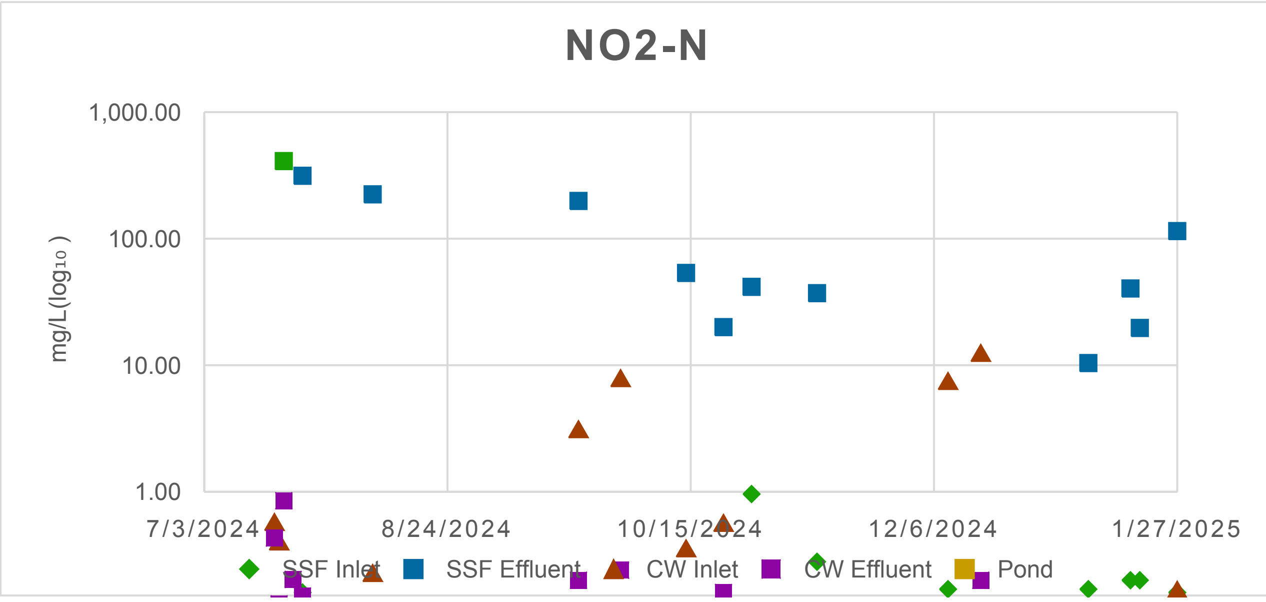

Figure 6: NO2-N values at the five different stages of the treatment system over time, plotted on a base-10 logarithmic scale.

5.1.5 – Nitrate Nitrogen (NO_3-N)

The NO_3-N concentration at the SSF inlet remained, as expected, relatively low, consistently below 11 mg/L. However, the SSF outlet showed significantly higher nitrate concentrations, exceeding 1400 mg/L in multiple instances, particularly between July and October. The observed nitrate accumulation indicates nitrification within the SSF. The peak values in October (2000 mg/L) suggest a high nitrification rate, potentially driven by increased microbial activity during favorable conditions, such as reduced N surface load compared to summer months (Figure 7).

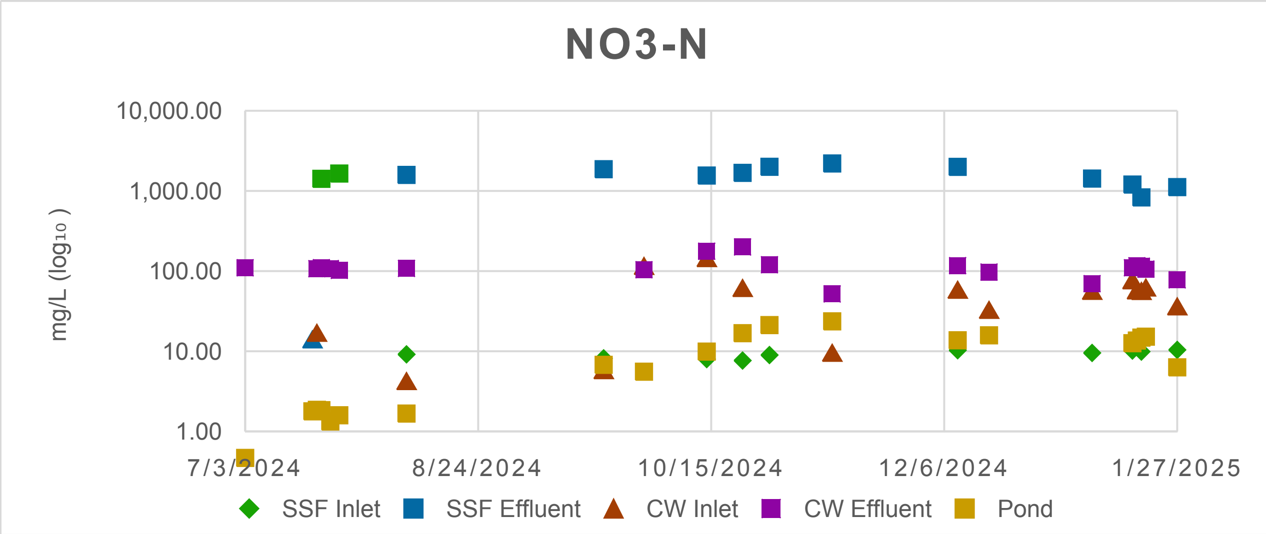

Figure 7: NO3-N values at the five different stages of the treatment system over time, plotted on a base-10 logarithmic scale.

At the inlet of the CW, the multi-chamber pit, NO_3-N concentrations ranged from as low as 3.9 mg/L to over 177 mg/L. This variability results from discontinuous, non-synchronized mixing of nitrate-rich SSF effluent with nitrate-poor greywater in the multi-chamber pit. At the CW outlet, NO_3-N concentrations ranged from 70 mg/L to 200 mg/L. Average NO_3-N concentration in the pond is 9 mg/L

5.1.6 – Total Nitrogen

The SSF inlet TN concentrations were found to be very high and varied significantly, ranging from 3018 mg/L up to 5420 mg/L. The SSF outlet, however, showed partial TN reduction, with average concentration values lying between 1839 mg/L and 3360 mg/L. Those values represent an average Total Nitrogen (TN) removal efficiency of approximately 38.4%. This partial reduction reflects the combined effects of ammonium adsorption onto the zeolite-containing filter media, biological nitrification, and potentially limited denitrification under transient anoxic conditions. While not sufficient for complete nitrogen removal, the SSF effectively reduces nitrogen loading to the downstream constructed wetland, supporting the stability and efficiency of the overall treatment system (Figure 8).

At the inlet of the downstream greywater CW in the multi-chamber pit, TN concentrations were considerably lower compared to the SSF outlet, ranging between 71.2 mg/L and 192 mg/L. This concentration reduction occurs through both (1) dilution by low-TN greywater and (2) denitrification processes within the multi-chamber pit.

Considering that inside the chamber the average Chemical Oxygen Demand (COD) was 194.81 mg/L and the average Total Nitrogen (TN) was 97.02 mg/L, a C/N ratio of approximately 2:1 is achieved. This low C/N ratio indicates a limited availability of readily biodegradable organic carbon relative to nitrogen, which limit complete denitrification. However, the anaerobic conditions prevailing in the pit, combined with moderate summer temperatures around 20 °C and high nitrate concentrations, create a favorable environment for denitrifying microbial communities. Given the presence of nitrate (NO_3^-) in the SSF effluent, the retention time in the multi-chamber pit, and the mixing with greywater providing at least some organic substrate, it is plausible to hypothesize that partial denitrification is occurring within this compartment. This would contribute to the overall nitrogen reduction by converting part of the nitrate to nitrogen gas before the effluent reaches the CW, thereby reducing nitrogen loading and enhancing the efficiency of the subsequent treatment stage.

The CW effluent demonstrated TN concentration values between 87.5 mg/L and 211 mg/L. The slightly higher effluent values than at the inlet can be attributed to water loss through evapotranspiration within the wetland, thus increasing TN concentrations

The final TN concentrations in the pond remained consistently low, mostly below 17.7 mg/L, indicating dilution, effective nitrogen assimilation by plants, and microbial processes within the pond.

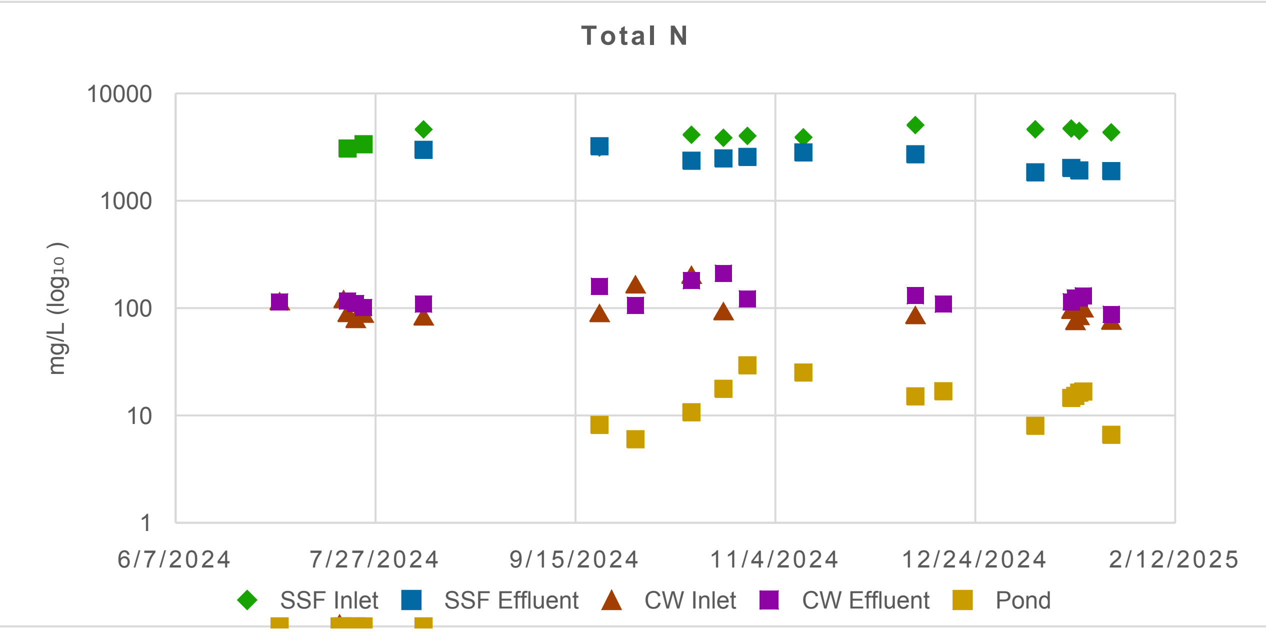

Figure 8: Total N values at the five different stages of the treatment system over time, plotted on a base-10 logarithmic scale.

5.1.7 – Total phosphorus

Phosphorus concentrations within the system were found to be already low due to the recovery of phosphorus through downstream struvite precipitation. Despite these low concentrations, the system further reduces residual phosphorus concentrations. Total phosphorus concentrations decreased from 11.4 ± 9.9 mg/L at the inlet to the SSF to an average value of 0.09 mg/L at the CW outlet, demonstrating the system’s effectiveness in reducing phosphorus concentrations. This can be seen in Figure 9

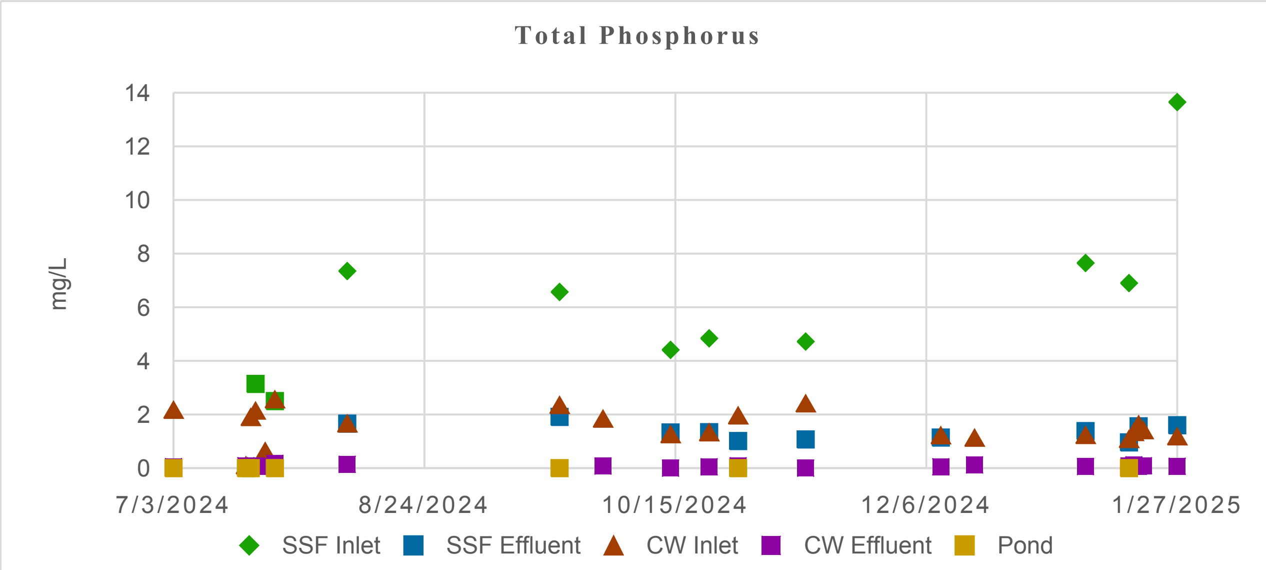

Figure 9: Total Phosphorus values at the five different stages of the treatment system over time.

5.1.8 – Temperature

At Reinighof, despite the decreasing temperatures from July to January (Figure 10), nitrogen transformation processes remained stable, possibly due to microbial adaptation, prolonged retention times, or seasonal changes in organic matter input affecting microbial activity. The biological processes occurring within the system have different optimal temperature ranges. Nitrification operates most efficiently between 25 and 30°C; denitrification has an optimal range between 20 and 30°C, but both can still occur at lower temperatures with reduced efficiency [7].

However, the adaptation of nitrifying bacteria to lower temperatures may explain the continued activity of the nitrifying bacteria within the filter bed even during colder periods. This finding can be corroborated by the results of Gottardo Morandi (2023), who demonstrated that a lava sand constructed wetland treating greywater maintained effective complete nitrification (NH_4-N ≤ 0.5 mg/L, no NO_2-N) even at water temperatures as low as 5°C, due to high cation exchange capacity and the swelling properties of the zeolite-containing lava sand, enhancing contact time, as well as increased BET adsorption surface, which prolonged microbial contact despite cold weather conditions.

If denitrification bacteria are also contributing to nitrogen removal in the system, it is likely occurring within the multi-chamber pit, where anaerobic conditions, supported by the availability of NO_2^- and readily available carbon as well as the high retention time in the pit, favor the process. Moreover, the buried structure may provide a more thermally stable environment, mitigating the effects of lower external temperatures. Further investigations are, however, needed in this matter.

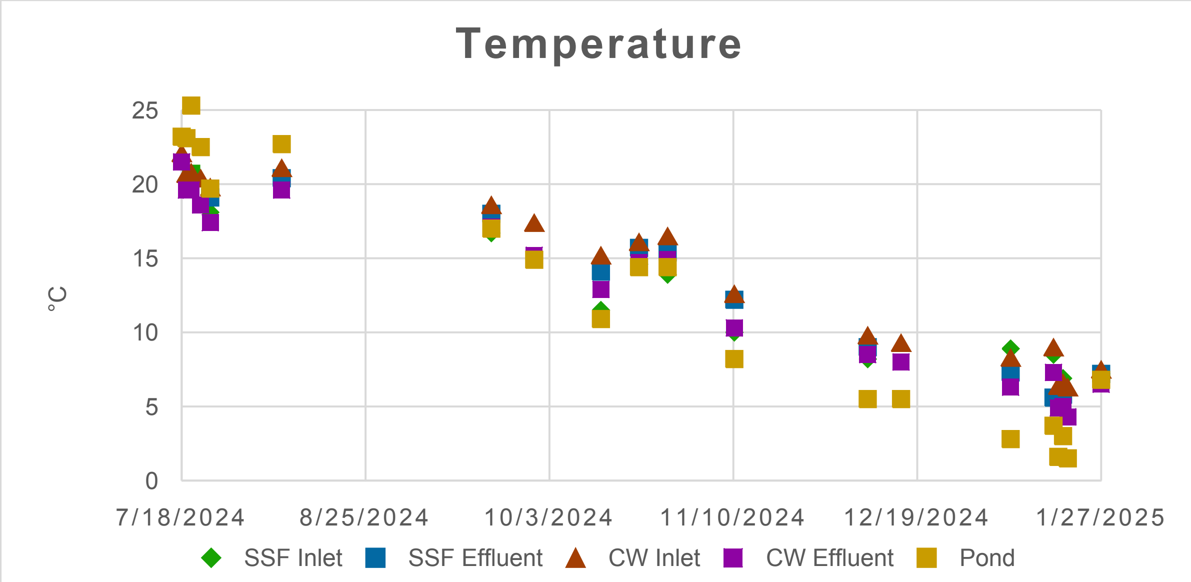

Figure 10: Temperature values at the five different stages of the treatment system over time.

6. Results of the Filter Sand Analyses

Five sand samples were analyzed to assess characteristics such as BET surface, particle size distribution (PSD), pH, cation exchange capacity (CEC) and chemical compounds such as Magnesium, Potassium, Calcium and Phosphorus. These included three samples of the lava sand material sourced from the Lissingen mining site: one from the upper layer and one from the lower layer of the clogged SSF filter bed, as well as one sample of unused Lissingen sand, representing the material prior to clogging. In addition, a sample of a mineral precipitate that had formed inside the outlet piping of the SSF was collected (Figure 11), as this buildup may have contributed to the clogging process. Finally, a sample of the newly installed filter material, a sand sourced from the Herchenberg mining site, was also included.

Figure 11: Precipitate observed within the outlet piping from the SSF before changing the material (Mai 2024).

The BET surface area analysis provides information about the total specific surface area of the soil particles. This is essential because the SSF relies on adsorption and biological processes to efficiently nitrify ammonium nitrogen. A decrease in surface area over time can indicate clogging due to biofilm growth, mineral precipitation, or organic matter accumulation, reducing treatment efficiency. If the BET surface area decreases significantly in clogged soil samples, it suggests pore clogging and reduced reactivity, impacting permeability and microbial activity. Higher surface area is associated with better retention of contaminants. Tracking changes in surface area helps assess the longevity of the filter medium [3].

The particle size distribution (PSD) reveals changes in grain size that can lead to reduced porosity and permeability, causing hydraulic retention time variations and potential system failure. If finer particles accumulate over time, they can block pores, leading to water stagnation and reduced nitrification efficiency. In this regard, d10 and d60 are important parameters, being d10 the grain diameter at which 10% of the filter media's mass is finer than this size and d60 the one where 60% of the material is finer than this size. Those data are obtained from a grain size distribution curve (PSD) and the curve for all the samples analyzed in this report can be found at the appendix. d10 and d60 values allow the calculation of the Uniformity Coefficient and can be determined using Equation (1). This tells how well graded (uniform or diverse) the material is. From the grain size distribution, the coefficient of permeability (Kfa) can also be derived, and can be calculated using equation 2.

$$u = \frac{d60}{d10} \tag{1}$$

$$Kfa = \frac{(d10)^2}{100} \tag{2}$$

Recommended values for the parameters like coefficient of permeability (Kfa), effective grain size (d10), and uniformity coefficient (U), are established in the DWA-A 262 (2017) guidelines. According to these guidelines, the target Kfa for the filter layer in vertical filters with sand (0-2 mm) should ideally amount to 10^{-4} m/s, and with lava sand (0-4mm) it should lay in the range of (0.4 -8).10^{-4} m/s. DWA-A 262 (2017) specifications also set a maximum allowable fine particle content of < 2% for fluviatile sand and < 8% for lava sand and a uniformity coefficient < 5, while d10 according to the guidelines should stay between 0.2 – 0.4mm.

The original Lissingen material exhibited a U of 15.4 and a d10 of 0.11 mm, both outside the recommended limits, although its initial Kfa of 1.2×10^{-4} m/s fell within the acceptable range. In contrast, the current Herchenberg lava sand demonstrates even poorer granulometric characteristics, with a U of 26.9 and a d10 of 0.016 mm, indicating a highly heterogeneous and fine-grained composition. The high proportion of finer grains in the filter media reduces pore space and significantly lowers the percolation velocity, further impairing the hydraulic performance of the system. Its Kfa, at 2.6×10^{-6} m/s, is significantly below the target range, suggesting limited permeability and a high risk of clogging. The BET surface area also differs significantly between the old and new filter media. While the sand from Lissingen exhibits a BET surface area of 81.9 m^2/g, the sand from Herchenberg shows a substantially lower value of 39.1 m^2/g. This indicates that the Herchenberg sand has a smaller specific surface area, which may result in reduced capacity for contaminant adsorption and retention. While the Lissingen material initially performed better in terms of hydraulic conductivity, both materials fail to meet essential DWA-A 262 criteria, with the Herchenberg sand showing greater deviation from optimal values. These results underline the need for a closer assessment and continuous monitoring of the system using the Herchenberg sand, to evaluate its long-term performance and detect potential issues such as clogging or reduced permeability over the coming years.

The results related to the previously used lava sand from the Lissingen site, before and after clogging, revealed significant changes in filter sand characteristics (see Table 2). The unused sand before start-up had a d10 of 0.11 mm. After filter clogging, both the top layer (d10 = 0.023 mm, U = 14.1) and the bottom layer (d10 = 0.0082 mm, U = 16.3) showed significant decreases in particle size. This suggests fine particle accumulation within the whole filter media but with a more significant accumulation of very fine particles at the bottom, likely due to precipitate that is not consistently separated from the supernatant, thus entering the SSF system through the inlet pipe or due to precipitation occurring inside the SSF that can form salt precipitates.

A decrease in hydraulic conductivity (k_{fA}) was also observed by comparing the Lissingen sand before and after operation. The permeability coefficient dropped from 1.2×10^{-4} m/s (before start-up) to 5.3×10^{-6} m/s (top layer) and 6.7×10^{-7} m/s (bottom layer). This suggests that clogging occurred over time, with the most severe reduction in permeability at the bottom of the filter. The finer particles and precipitates likely reduce pore spaces, leading to restricted water flow.

The BET surface area of the Lissingen lava sand also decreased with operation. The initial BET surface area was found to be 81.9 m²/g, but it decreased to 57.05 m²/g (top layer) and 57.15 m²/g (bottom layer) after filter clogging. This reduction indicates that the surface of the grains potentially became partially coated with solid deposits, reducing available adsorption sites.

Table 2 presents the values for Cation Exchange Capacity (CEC) and base saturation—two interrelated parameters that offer key insights into nutrient retention and availability within the analyzed soil and filter media. CEC reflects the total capacity of a material to hold exchangeable cations such as calcium (Ca^{2+}), magnesium (Mg^{2+}), potassium (K^{+}), and sodium (Na^{+}), while base saturation denotes the proportion of the CEC that is occupied specifically by these basic cations. Base exhange is calculated using equation 3.

$$Baseexhange\,(\%) = \frac{[Ca^{2+}+Mg^{2+}+K^{+}+Na^{+}]\cdot 100}{CEC} \qquad (3)$$

Among all materials examined, Lissingen sand exhibited the highest initial CEC (50–70 cmol(+)/kg), indicating a strong capacity for nutrient retention and buffering. However, this capacity declined following system operation, as shown by the reduced values measured in the used filter layers—26.2 cmol(+)/kg in the top layer and 21.5 cmol(+)/kg in the bottom layer. Unfortunately, base saturation data for the Lissingen material prior to use are unavailable, as no measurements were conducted at the time and the original sample is no longer accessible.

The Herchenberg sand displayed the highest measured base saturation at 132.5%, suggesting a high proportion of exchangeable bases. Nonetheless, this elevated percentage is not fully supported by the absolute concentrations of individual cations, indicating a comparatively lower actual ion exchange capacity than Lissingen sand. The base saturation values of the used Lissingen layers were similarly elevated, with 107.4% in the top and 116.0% in the bottom layer, likely reflecting retained basic cations despite CEC reduction.

Notably, the precipitated material collected from the pipe exhibited the highest base saturation at 137.5%. However, this was accompanied by very low concentrations of exchangeable cations, suggesting a limited CEC. These findings imply that, due to its fine texture and low ion exchange capacity, the precipitate acts more as a passive sink for cations rather than an effective exchange medium.

The sample of the precipitated solid mass within the inlet pipe exhibited no significant porosity. The material was found to be distinctly hydrophobic, which can be inferred both from its low C-constant value and from attempts to disperse the material in water. The isotherm showed a pronounced hysteresis across the entire range, indicating the presence of very small, kinetically inhibited micro pores within the size range of nitrogen molecules.

Table 2: Comparison of BET, CEC, Kfa, U and d_{10} from lava sand from Lissingen, Germany, before and after clogging and the current lava sand used in the system, from Herchenberg, Germany before use.

	U	**d10**	**kfA**	**BET**	**CEC**	**Base Saturation**
	(d_{60}/d_{10})	(mm)	($0{,}01\cdot d_{10}^2$ in m/s)	(m²/g)	cmol+/kg	%
Lava sand before start-up (Lissingen)	15.4	0.11	$1.2\cdot10^{-4}$	81.9	50-70*	-
Top layer lava sand after clogging (Lissingen)	14.1	0.02	$5.3\cdot10^{-6}$	57	26.2	107.4
Bottom layer lava sand after clogging (Lissingen)	16.3	0.008	$6.7\cdot10^{-7}$	57.2	21.54	116.0
Current filter material – lava sand from Herchenberg	26.9	0.016	$2.6\cdot10^{-6}$	39.1	15.3	132.5
Sand precipitated in the pipe	15.7	0.0032	$1.0\cdot10^{-7}$	1.4	4.6	137.5

* Value modified from Hasselbach and Bruch (2010). The original value of 500-700 mmol/g is implausibly high, far exceeding typical CECs for cation exchangers. The presented range of 50-70 cmol+/kg is based on an assumed intended value of 500-700 mmol+/kg.

The decline in BET surface area and other changes in physical-chemical properties of the filter material strongly suggest that precipitation processes occur inside the system or precipitates enter the system, potentially altering the filter material's composition and efficiency. To investigate this hypothesis, a chemical analysis of the different filter layers and sand precipitated in the pipes was conducted to determine which compounds were accumulating and possibly affecting the system's performance. The results are depicted in Table 3.

A significant increase in phosphorus (P_2O_5) was observed in the top layer of the former filter material after decommissioning, as shown in Table 3. The presence of 395 mg/100 g of phosphorus in the top layer and 264 mg/100 g in the precipitated solid mass within the inlet pipe supports the hypothesis that phosphorus compounds like calcium phosphates are being deposited within the system, especially in the top centimeters of filter layer. The molar ratio of Magnesium to Phosphorus (1:11.3) in the sand precipitated in the pipe is far from the 1:1 stoichiometric ratio required for struvite, so the presence of struvite cannot be confirmed. A high content of potassium is also observed at the top and bottom layer of the clogged filter media. The molar ratio of Magnesium to Potassium to Phosphorus (1:2.4:11.3) is also not consistent with the 1:1:1 molar ratio for potassium struvite (K-Struvite; $MgKPO_4.6H_2O$), so the presence of K-struvite inside the filter cannot be confirmed either [5]. It is also likely that the presence of NH_4-N within the filter body hinders the formation of K-struvite. On the other hand this can suggest that potassium was absorbed by zeolites present in the filter media or other K salts like K_2SO_4 have been precipitated.

The molar ratios, especially the high P:Mg (11.3:1) and Ca:Mg (7:1) alongside a Ca:P ratio of approximately 0.62 and a pH of 7.9 strongly suggest that calcium phosphates are the most likely major phosphorus-containing precipitates, while magnesium-containing phosphates (like struvite or K-struvite) can only be minor components due to the relative scarcity of magnesium compared to phosphorus. There is, however, an excess of phosphorus even after accounting for all the calcium and magnesium precipitating as common phosphates. This remaining phosphorus might be associated with K/Na, adsorbed, or present in less common forms.

Moreover, the accurate quantification and identification of crystalline precipitates like struvite may be limited using the employed analytical methods targeting plant-available phosphorus and potassium (via CAL) and exchangeable magnesium (via $CaCl_2$) rather than total elemental content, potentially underestimating crystal-bound magnesium. Future work should include more comprehensive techniques such as total elemental analysis and X-Ray Diffraction to determine more precisely the phosphorus compounds present in the precipitate and within the filter material.

Table 3: Chemical properties and cation exchange capacity of soil samples.

Parameter	Unit	Current filter material	Top layer former filter material	Bottom layer former filter material	Sand precipitated in the pipe
pH		6.6	7.7	7.5	7.9
Phosphorus (P2O5) in CAL*	mg/100g	40	395	28	264
Potassium (K2O) in CAL	mg/100g	145	352	289	37
Magnesium (Mg) in CaCl2**	mg/100g	8	38	11	8
Calcium /eff CEC***	cmol+/kg	12.8	2.3	6.5	4.6
Magnesium /eff CEC	cmol+/kg	1.8	7.9	1.9	0.8
Potassium /eff CEC	cmol+/kg	4.3	12.7	11.1	0.3
Sodium /eff CEC	cmol+/kg	1.5	5.2	5.5	0.6

*in CAL – Extractable nutrients using calcium lactate/acetic acid (plant-available P and K).

**in $CaCl_2$ – Values determined using calcium chloride extraction (available Mg and stable pH).

***eff CEC – Effective cation exchange capacity; indicates the soil’s ability to retain nutrient cations.

As shown in Table 4, the TOC content of the former clogged SSF filter media was substantially higher in the top layer (0.64%) compared to both the bottom layer (0.14%) and the current filter media (0.15%). For quality control purposes, each parameter was analyzed in duplicate; the two measurements are denoted as M1 and M2, representing the results of the duplicate analyses conducted on the same sample to determine a mean value. This gradient suggests organic matter accumulation at the surface due to filtration and microbial activity, a typical pattern in vertical flow systems where influent percolates from top to bottom. The precipitated material inside the inlet pipe exhibited the highest TOC value (0.72%), indicating that organic-rich deposits may also form within upstream components and contribute to clogging risks.

The new filter media's low TOC (0.15%, Table 4) reflects its initial state and brief operational time before sampling. The irreversibility of the filter clogging points to inorganic material buildup as the decisive cause of system failure, rather than primarily organic matter, which tends to be decomposable over time. Nevertheless, the TOC data reveals a clear pattern: the top layer of the former SSF (0.64% TOC) and particularly the inlet pipe precipitate (0.72% TOC) showed significant organic accumulation compared to the low TOC in both the bottom layer of the former filter (0.14%) and the new material (0.15%). This stratification suggests concentrated heterotrophic activity at the system's inlet and upper filter regions, likely with a different, possibly more autotrophic, microbial environment (nitrifying bacteria) deeper in the filter. Overall, the TOC results support the idea that while this substantial organic accumulation at the surface and inlet was not the primary cause of the irreversible failure, it likely contributed to an initial reduction in porosity and hydraulic performance, potentially creating conditions that exacerbated the subsequent, decisive inorganic clogging.

Table 4: Total Organic carbon values from soil samples

TOC			
	M1 TOC(%)	M2 TOC (%)	Mean Value
Current filter material SSF	0.15	0.15	0.15
Top layer former SSF filter material	0.63	0.65	0.64
Bottom layer former SSF filter material	0.14	0.14	0.14
Precipitated inside the pipe	0.71	0.73	0.72
MAP	0.33	0.42	0.37

The conditions within the SSF create an environment favorable to the formation of low-solubility precipitates. The high concentrations of Mg^{2+}, K^{+}, Cl^{-}, and other salts originating from urine, combined with an elevated pH of approximately 9.4 at the inlet and favorable temperatures, promote a wide array of precipitation processes. Additionally, struvite precipitates may be carried downstream by the urine supernatant, further contributing to mineral accumulation within the system.

Preventing precipitation entirely within the system presents a significant challenge. However, strategies such as enhanced separation of struvite precipitate before SSF feeding or backwashing the filter bed regularly could potentially help extend the lifespan of the filter media by mitigating clogging and mineral buildup. If the replacement becomes necessary, which is highly probable, the system as a whole is not strongly affected, since the downstream CW ensures operational continuity.

7. Results of the Microbiological Analysis

Initial delays were encountered in securing partnerships with laboratories capable of analyzing pathogens, therefore those analyses started only in October. Also, due to only discontinuous feeding events to the SSF, there are fewer results from SSF compared to the CW and the pond.

Pathogen concentrations were higher in the mix of greywater and urine than in pure urine, as expected based on literature standard values for greywater [8]. The microbiological performance of the treatment system was evaluated by calculating the log reduction of key indicator organisms across two treatment stages: the Soil Filter (SSF) and the Constructed Wetland (CW) (Table 5). The SSF achieved modest reductions, with log reductions < 1 for *Enterococcus*, *E. coli*, and coliforms, indicating limited microbial removal capacity. In contrast, the CW demonstrated higher effectiveness, achieving log reductions of around 2 for *Enterococcus*, 1 for *E. coli*, and < 1 for coliforms. These results highlight the superior pathogen removal performance of the CW compared to the SSF, likely due to longer retention times, greater microbial activity, and physical-chemical interactions within the wetland environment.

In the final pond, Enterococci and *E. coli* concentrations were always < 60 MPN/100ml. Additional disinfection, such as UV treatment, could further enhance water safety if required.

When compared to regulatory benchmarks, these values meet the EU Bathing Water Directive (2006/7/EC) standards for "excellent quality", which require *E. coli* to be ≤250 MPN/100 mL and Enterococci ≤100 MPN/100 mL in inland waters. Additionally, the results comply with the German Water Reuse Ordinance (Wasserwiederverwendungsverordnung), which is based on the EU Regulation 2020/741 for safe water reuse. According to this ordinance, for unrestricted irrigation or urban reuse (Category A), *E. coli* must not exceed 10 MPN/100 mL in 80% of samples and 100 MPN/100 mL in any single sample. While the system's values are slightly above the strictest threshold for Category A, they comfortably meet the requirements for Category B reuse (e.g., for crops not consumed raw or restricted irrigation), where the limit is ≤100 MPN/100 mL. Overall, the system demonstrates a strong capacity for microbial reduction and indicates suitability for both environmental discharge and certain reuse applications, though additional disinfection may be required for stricter water reuse classes.

Table 5: Results of Microbiological Analysis

Sample	Date	Enterococcus MPN/100ml	E. coli MPN/100ml	Coliforms KBE/100ml
Inlet SSF	29.10.24	2.30E+04	2.30E+05	2.40E+04
	27.01.25	1.60E+04	2.30E+04	> 2.40E+05
Outlet SSF	29.10.24	6.00E+02	2.90E+04	8.70E+04
	27.01.25	8.30E+03	4.00E+03	2.90E+03
Inlet CW	29.10.24	2.60E+04	6.60E+05	2.40E+05
	13.11.24	1.70E+04	2.20E+06	2.40E+05
	11.12.24	7.60E+03	8.20E+03	2.40E+05
	16.12.24	2.90E+03	3.20E+03	2.40E+05
	27.01.25	2.50E+04	5.80E+04	> 2.40E+05
Outlet CW	29.10.24	<60	4.80E+02	2.40E+04
	13.11.24	240	1.60E+04	2.40E+05
	11.12.24	<60	6.80E+02	1.00E+05

	16.12.24	<60	58	1.10E+03
	27.01.25	< 60	1.20E+02	2.50E+02
	29.10.24	<60	56	1.70E+03
	13.11.24	<60	58	2.30E+03
Pond	11.12.24	<60	<60	3.90E+02
	16.12.24	<60	<60	1.30E+02
	27.01.25	<60	<60	9.80E+02

8. Transferability of the system to Brazilian conditions

Brazil's tropical and subtropical climate presents significant differences from the temperate conditions of the current system location, the Reinighof. One of the primary factors influencing wastewater treatment is temperature, which directly affects biological processes such as nitrification, denitrification, and anammox processes. Higher average temperatures in Brazil (often above 20°C year-round) could enhance these microbial processes, potentially increasing nitrogen transformation efficiency.

In particular, nitrification typically operates best between 25–30°C but can experience inhibition at excessively high-water temperatures (>35°C). Denitrification, in turn, remains effective at temperatures above 20°C, making it well-suited for Brazilian conditions. Anammox thrives between 20–40°C, suggesting that, if anammox processes are present in the multi-chamber pit, they could function optimally in Brazil, if other prerequisites are met.

Evaporation rates would be significantly higher in Brazil, especially in semi-arid regions, which could lead to increased concentration of dissolved solids and increased clogging risk within the system. Additionally, higher precipitation in some regions may require higher freeboards of the wetland systems.

Beyond climatic aspects, the feasibility of implementing such a system in Brazil depends on economic, social, and cultural factors. Key challenges are implementation and maintenance costs. While Brazil has a mix of wealthy urban areas and lower-income rural communities, many municipalities lack financial resources for infrastructure projects. While technologies such as struvite reactors may require additional operational expertise and maintenance, which can pose challenges in some regions due to cost or limited technical capacity, their implementation remains feasible when supported by decentralized energy sources (e.g., solar panels) or engaged local actors. In contexts where reliable electricity or skilled personnel are unavailable, especially in remote or informal settlements, alternative low-energy solutions—such as water-saturated horizontal subsurface flow constructed wetlands—may offer more appropriate options. However, with proper training, community involvement, and adaptable system design, even more complex technologies can be successfully operated in diverse rural settings.

Public perception and local traditions also play a significant role in adopting new wastewater treatment technologies. In some areas, urine separation and reuse (as in phosphorus recovery via struvite precipitation), especially the implementation of dry toilets, might face resistance due to social taboos or lack of awareness. Successful implementation would require community engagement, educational programs, and policy incentives to promote acceptance and participation.

The Transferability of decentralized wastewater treatment systems extends beyond Brazil to other regions with similar boundary conditions worldwide. Nevertheless, the Reinighof system holds significant potential for adaptation to different contexts. The successful transfer of decentralized wastewater treatment systems requires careful consideration of various factors, including climate conditions, economic feasibility, technical aspects, social acceptance, environmental impact, and regulatory requirements. A thorough assessment is essential to ensure the effective implementation of such a system. A case-by-case assessment is necessary, as each location presents unique challenges that must be addressed for successful adaptation.

10. Modifications to the Project Scope related to Initial Proposal

No significant modifications have been introduced to the original project framework. The sampling schedule, analytical methodologies, and overarching objectives of the project remain consistent with the initial proposal. This consistency ensures that the study remains aligned with its original goals while incorporating adaptive strategies to refine system performance.

11. Presentations and Publications

The research conducted in this project was presented at the IWA Nutrient Removal and Recovery Specialist Conference 2024 (https://uqevents.eventsair.com/nrr24/). This biennial conference took place from November 17 to 21, 2024, in Brisbane, Australia. On the fourth day of the conference, Virginia Ly Pinto (Figure 13) presented her research in a presentation contribution entitled *"Onsite Greywater and Blackwater Management: Nature-Based Solutions for Water and Nutrient Reuse"* during the *"City and Precinct Outcomes"* session. Virginia Ly delivered a presentation on decentralized wastewater management aimed at recovering nutrients from greywater and blackwater. The presentation focused on case studies conducted in Germany and in Brazil. Based on the findings from these studies, the presentation concluded with a discussion on the potential of these systems for urban integration, as well as their social and ecological benefits for different communities. This was supported by practical examples and comparative analyses.

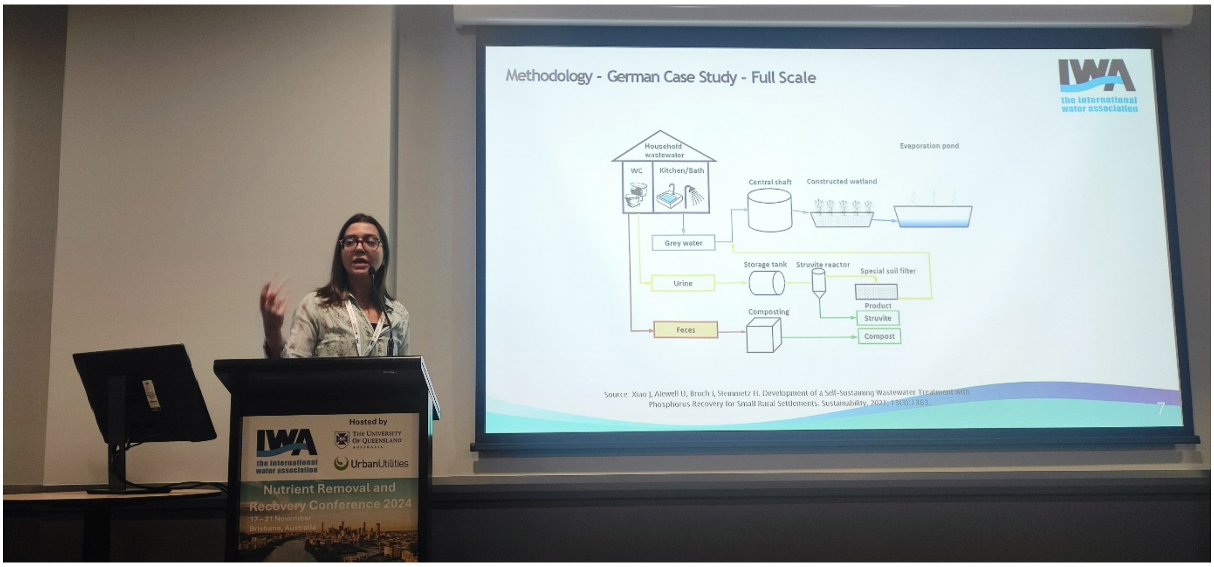

Figure 12: Presentation by Virginia Ly Pinto on the use of nature-based solutions for wastewater treatment in Reinighof, Germany, at IWA Conference in Brisbane.

12. Conclusions and Recommendations for Future Research

The overall treatment system for phosphorus recovery as well as urine and greywater treatment, including struvite precipitation, special soil filter, and constructed wetland, has demonstrated robust performance in phosphorus recovery, nitrification and removal of organic matter over the course of the project, even during peak season. An integrated and complex system comprising technical and nature-based solutions is fundamental to meet treatment targets and operational reliability.

Total nitrogen (TN) was reduced from up to 5420 mg/L in the urine supernatant to 126.4 mg/L at the outlet of the constructed wetland. Ammonium nitrogen (NH_4-N) was reduced from values as high as 4774 mg/L to 0.7 mg/L in constructed wetland effluent. Nitrate Nitrogen (NO_3-N) concentrations increased significantly within the SSF, due to nitrification, peaking on average 2000mg/L, and at the outlet of CW, NO_3-N concentrations were the 70 mg/L to 200 mg/L. Total phosphorus (TP) concentrations, already low due to prior struvite recovery, dropped from 11.4 mg/L to an average value of 0.09 mg/L in the final wetland effluent. Chemical Oxygen Demand (COD) was consistently reduced, with initial concentrations of 5046 ± 802 mg/L at the inlet of the SSF and 17.4 ± 7.0 mg/L in the effluent of the CW. The reduction in turbidity from an initial 67.3 NTU to 1.73 NTU in the final effluent also indicates an efficient removal of suspended solids.

Removal of nitrogen in decentralized systems is a challenge but, when compared to other decentralized nitrogen removal strategies – such as ammonium stripping and absorption –, the current system offers significant cost advantages. Although periodic replacement of the filter media can be required due to clogging and mineral deposition, this approach is more sustainable and economically viable.

At the final disposal stage, the evaporation pond - which receives the treated effluent - exhibited consistently low concentrations across all monitored parameters, indicating effective upstream treatment. Total nitrogen (TN) concentrations in the pond remained below 17.7 mg/L, while ammonium nitrogen (NH_4-N) averaged 0.27 ± 0.1 mg/L, with values ranging from 0.08 to 0.74 mg/L. Nitrate nitrogen (NO_3-N) averaged 9 mg/L, and nitrite nitrogen (NO_2-N) remained consistently low, mostly below 0.1 mg/L. Total phosphorus (TP) concentrations were < 0.05 mg/L, confirming efficient removal throughout the system. COD concentrations varied between 13.5 and 51.1 mg/L, with higher values during the summer, likely due to algal growth. Turbidity decreased significantly from earlier system stages, with an average value of 1.73 NTU. Microbiological analysis showed that *Escherichia coli* and *Enterococcus* levels were consistently < 60 MPN/100 mL, aligning with EU Bathing Water Directive standards for excellent quality. These results demonstrate that the system successfully protects the receiving environment, even under varying seasonal and loading conditions.

Looking forward, several development opportunities could further enhance system resilience and efficiency of the current system. First, modifying the feeding pattern for the SSF to a more intermittent feeding with less volume per feeding event could optimize hydraulic retention times, especially during summer, alleviating the system and potentially contributing to better nitrification rates. In parallel, strategies should be explored to improve solid-liquid phase separation after struvite precipitation, minimizing particle carry-over into the SSF. Finally, exploring the cultivation of plant species capable of thriving in the high-nitrogen environments within the SSF may further contribute to nitrogen removal and overall system stability, because, among others, the plants root movement, induced by wind sway, helps loosen the filter media, reducing clogging risks.

These finds suggest that, with targeted modifications and further research, the system can be adapted to a variety of operational contexts while maintaining both technical and economic feasibility.

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Appendix

Table 6 - Average values of all analyzed parameters for the five points of the system.

Parameter	Unit	SSF Inlet		SSF Effluent		CW Inlet		CW Effluent		Pond	
		Average ± Std. Dev. (Min – Max)	n	Average ± Std. Dev. (Min – Max)	n	Average ± Std. Dev. (Min – Max)	n	Average ± Std. Dev. (Min – Max)	n	Average ± Std. Dev. (Min – Max)	n
pH	-	9.4 ± 0.2 (9.2 – 10.0)	13	7.5 ± 0.3 (7.1 – 8.2)	13	7.7 ± 0.2 (8.15 – 7.4)	20	6.9 ± 0.1 (7.2 – 6.7)	20	7.7 ± 0.2 (8.1 – 7.4)	21
T	ºC	12.8 ± 5.2 (6.9 – 22.0)	13	13.1 ± 5.6 (5.6 – 20.7)	13	14.0 ± 4.6 (22.0 – 6.0)	20	12.6 ± 5.9 (21.5 – 4.3)	20	12.5 ± 8.2 (23.0 – 1.5)	21
turbidity	NTU	67.3 ± 49.9 (21.9 – 175.0)	10	74.3 ± 126 (13.2 - 409)	10	68.3 ± 22.5 (96.9 – 27.1)	14	4.6 ± 6.7 (26.8 – 0.9)	14	1.73 ± 1.9 (8.4 – 0.76)	15
conductivity	µS/cm	28850 ± 2420 (258000 – 34000)	13	20130 ± 3820 (14960 - 26000)	13	1764 ± 469.2 (2700 – 1019)	20	1987 ± 361.8 (2690 – 1441)	20	903 ± 183.4 (1169 – 410)	21
COD	mg/L	5046 ± 802.4 (4105 – 7095)	13	1603 ± 638.3 (2970 – 889)	13	194 ± 88.02 (406 – 73.9)	20	17.4 ± 7.1 (38 – 9.9)	20	33.1 ± 11.3 (51.1 – 13.5)	21
N total	mg/L	4254 ± 689.4 (3018 - 5420)	13	2555 ± 525.9 (3360 – 1888)	13	86.5 ± 45.6 (192 – 22.15)	20	116 ± 41.4 (211 – 16.3)	20	10.7 ± 8.0 (29.3 - <1)	21
P total	mg/L	15 ± 16.0 (6.57 – 57.9)	13	1.58 ± 0.6 (3.14 – 0.96)	13	1.5 ± 0.6 (2.5 – 0.5)	20	0.09 ± 0.03 (0.18 - < 0,05)	20	< 0.05	21
Po43-	mg/L	0.8 ± 0.6 (0.4 – 2.4)	13	0.2 ± 0.2 (0.3 – 0.03)	13	< 0,05	20	< 0.05	20	< 0.05	21
NO3N-	mg/L	8.6 ± 2.7 (0.3 – 10.9)	13	1578 ± 387.6 (2200 – 830)	13	35.7 ± 40.9 (134 – 0.23)	20	105 ± 39.8 (200 – 3.2)	20	9.1 ± 7.1 (23.6 – 0.5)	21
NO2N-	mg/L	0.21 ± 0.2 (0.05 – 0.96)	13	114 ± 131.7 (410.0 – 2.5)	13	2.0 ± 4.0 (7.5 - < 0,015)	20	0.15 ± 0.2 (0.8 – 0.01)	20	0.04 ± 0.03 (0.08 – 0.01)	14
SO42-	mg/L	939 ± 130.2 (813 - 1400)	11	635 ± 202.7 (976 – 404)	11	76.6 ± 14.6 (93.5 – 47.8)	19	75.0 ± 18.3 (116.6 – 49.7)	19	< 40,0	13
Cl-	mg/L	2538 ± 345 (2085 - 3150)	12	1395 ± 378 (2300 – 910)	12	119 ± 75.9 (267 – 52.6)	17	112 ± 47.4 (210 – 62.2)	17	89.7 ± 26.6 (164 – 36.5)	18
Na+	mg/L	1525 ± 191.5 (1100 – 1728)	12	1008 ± 271.0 (1380.6 – 880.8)	12	123 ± 24.9 (152 – 67.9)	17	119 ± 21.1 (155.2 – 88.7)	17	83.4 ± 17.7 (108.2 – 44.7)	18
NH4-N	mg/L	3992 ± 546 (2935.8 - 4774)	12	902 ± 495.5 (1760 – 360)	12	42.0 ± 23 (9.5 – 85.8)	17	0.7 ± 0.4 (1.9 – 0.4)	17	0.27 ± 0.1 (0.9 – 0.10)	18
K	mg/L	1350 ± 414 (864.4 – 2512)	12	1094 ± 525 (1960 – 460.6)	12	73.0 ± 28.6 (113 – 34.5)	17	35.4 ± 7.2 (46.9 – 25.6)	17	29.2 ± 8.2 (41.4 – 12.2)	18
Ca	mg/L	44.2 ± 29.3 (5.26 – 122)	12	1579 ± 684.5 (2567 – 422.3)	12	145 ± 75.9 (276.9 – 49.9)	17	239 ± 56.4 (331.3 – 168.4)	17	72.8 ± 18.2 (95.2 – 31.3)	18
Mg	mg/L	129 ± 46.8 (26.2 – 227.8)	12	205 ± 45.0 (272 – 133)	12	22.8 ± 7.1 (37.6 – 11.1)	17	24.0 ± 5.13 (32.8 – 17.0)	17	13.5 ± 3.0 (16.3 – 6.1)	18