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Sustainable management of coastal organic waste: A pilot study on beach wrack treatment in reed bed system

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ABSTRACT

Beach wrack (BW), a natural accumulation of marine organic material on coastal shorelines, presents significant management challenges due to its high water content and nutrient-rich composition. This study investigates the use of a reed bed system (RBS) for processing BW and its mixture with compost. A pilot facility was established to evaluate dewatering efficiency, stabilization processes, and changes in key parameters such as dry matter, organic matter, nitrogen, and phosphorus content. The results indicate that RBS effectively increase the dry matter content. In case of stabilization of organic matter further studies are necessary. Quarters supplied with a mixture of BW and compost demonstrated enhanced nutrient concentration compared to those fed with BW only. The findings underscore the potential of RBS as a sustainable solution for managing BW, fitting with circular economy principles by transforming waste into valuable resources.

Keywords: stabilization, dewatering, nutrient recovery, reed bed system, beach wrack, organic waste management.

INTRODUCTION

In the past, the term "wrack" was associated with brown macroalgae equipped with air bladders, enabling floating in marine environments. Once these organisms were deposited on shorelines, they were called "beach wrack" (BW) (Chubarenko et al., 2021; Macreadie et al., 2017). Currently, BW is broadly defined as organic material accumulating on beaches, predominantly composed of macroalgae and seagrasses (Orr et al., 2005). Due to intensified human activities, BW often is mixed with anthropogenic waste (Robbe et al., 2021). This material is transported to shorelines by hydrodynamic forces, and is often recirculated back into the marine environment through the same (Barreiro et al., 2011; López et al., 2019).

Worldwide coastal regions observed BW accumulations. The volume of BW deposited along shorelines differs and is shaped by spatial distribution, temporal dynamics, and seasonal variations (Kupczyk et al., 2019; Barreiro et al., 2011). As reported by Barreiero et al. (2011), *Posidonia oceanica* exhibit high primary productivity, but their leaves are weakly consumed by herbivores. Instead, BW undergo natural shedding, contributing to necromass accumulation in coastal waters. Necromass is exported from seagrass meadows. This export can range from 10% to 55% of the total primary production of *Posidonia oceanica,* depending on water depth, hydrodynamic conditions, and geographic location. The studies conducted by Gómez (2013) and Guerrero-Meseguer (2020, 2023) demonstrate that BW accumulation tends to peak during the summer and autumn months. The storm events, play a significant role in the detachment of macroalgae and contribute to the temporal variability of beach wrack deposition, as noted by Song et al. (2015).

The composition of BW is influenced by regional and seasonal factors. The spatial placement of BW within the tidal zone is influenced by the physical properties of the beach (gradient, morphodynamic features, swash zone dynamics, and level of exposure). These factors condition the time of BW deposition at particular locations (Gómez et al., 2013; Ruiz-Delgado et al., 2015). The Eastern coast of Poland is mainly characterized by *Fucus vesiculosus* and *Potamogeton spp*. (Chubarenko et al., 2021).

After the deposition of BW on beaches, it undergoes processes such as fragmentation and decomposition, facilitated by meiofauna, macrofauna, and bacteria (Colombini et al., 2003). The rate of decomposition is influenced by various factors, including the volume of accumulated material, beach morphology, and the physicochemical and nutritional characteristics of the macroalgae (Braeckman et al., 2019). The nutrients released during decomposition enrich coastal waters like fertilizer (Kupczyk et al., 2019; Lastra et al., 2014). The bacterial activity in decomposing BW enhances CO_2 respiration, turning wrack deposits into metabolic hotspots (Lastra et al., 2018). Rising global temperatures are expected to accelerate this process, increasing CO_2 emissions and intensifying nutrient cycling (Coupland et al., 2007). Additionally, BW supports the reproduction of microalgae and the growth of bacteria, providing a critical food source for macrofauna and maintaining biodiversity in marine ecosystems (Gómez et al., 2018; Ruiz-Delgado et al., 2015). However, excessive accumulation of BW can create significant ecological challenges, particularly in eutrophic water bodies, such as the Baltic Sea (Kupczyk et al., 2019).

The accumulation of BW presents both environmental and economic challenges, reinforced by agricultural practices and climate change. Climate-related phenomena, such as raised sea surface temperatures, contribute to nutrient enrichment, encouraging the formation of algal blooms in coastal ecosystems (Alobwede et al., 2019). During BW decomposition, gases such as CO_2, CH_4, and H_2S are released, contributing to climate dynamics and atmospheric pollution (Björk et al., 2023).The removal of BW from beaches is associated with significant costs, as it requires heavy machinery which could damage beach morphology and produces new pollution like noise (Boudouresque et al., 2016). Managing BW as waste involves hundreds to millions of euros annually across various coastal regions (Robbe et al., 2021; Macreadie et al., 2017).

Recognizing BW as a valuable resource usable in applications such as organic fertilizers and biofuels presents an opportunity to reduce its environmental impact. BW presents significant potential beyond traditional applications, particularly as a source of soil fertilizers and plant biostimulants (Villares et al., 2016; Illera-Vives et al., 2013). Research demonstrates that both raw and composted macroalgae can positively influence the physical and chemical properties of soil (Illera-Vives et al., 2015), increase microbial activity (Wang et al., 2016, 2018), and provide nutrients essential for beneficial soil organisms (Butt et al., 2020).

BW poses a dual challenge: it impacts eutrophic water bodies and contributes to climate change (environmental issues) while also creating social and economic problems for local authorities. Under Directive 2006/7/EC of the European Parliament and the Council of 15 February 2006, concerning bathing water quality management, coastal municipalities are obligated to manage the accumulation of macroalgae on shorelines. Furthermore, compliance with Directive (EU) 2018/850 prohibits the landfilling of biodegradable waste, mandating the utilization of materials such as BW (Kupczyk et al., 2019). As an underutilized organic biomass, BW has significant potential for various applications within the circular economy framework, including coastal protection, biogas generation, and bioresource recovery (Chubarenko et al., 2021; Villares et al., 2016).

Polish waste management framework is still in its developmental stages, where BW is primarily classified as waste with minimal efforts directed towards its recycling or reuse. Beach management in Poland is overseen by local government entities, such as the Gdynia and Gdańsk Sports Centres, which contract private companies for beach cleaning and maintenance. However, these contracts frequently lack explicit provisions regarding the treatment or management of BW, leading to its general disposal instead of exploring recycling. These practices reflect the need for more comprehensive policies and strategies to address the dual goals of sustainable beach management and resource reuse.

One of the Gdańsk University of Technology concepts fitting in use BW as a source is exploring a reed bed system (RBS) processing BW material. Since the end of the 1980s, this method has been used in many European countries (France, Denmark, Germany, Greece, Spain, Belgium, Italy) as well as in Great Britain. In Poland, such facilities can be found in the conventional wastewater treatment plants in Gniewino and Zamborów. In smaller wastewater treatment plants, this method was used in Darżlubie and Nadole. Until now, this technology has been used to dewater and stabilize sewage sludge from various sources: from aerobic stabilization, from anaerobic digestion, from conventional activated sludge chambers, from chambers with prolonged aeration, from septic tanks or Imhoff tanks. The RBS method is based on multi-layer floods of concrete above-ground objects or basins inhabited by reed with sewage sludge (characterized by a high water content of about 99–99.5%). The exploitation of RBS is on average from 8 to 12 years, but there are also facilities where treatment time of sewage sludge is longer and is to 15 or even 20 years. RBS usually consists of several quarters and works in an alternating cycle: this means that only one quarter is flooded at a time (the irrigation phase) and the others are in the rest phase – they are not flooded (Kołecka, 2019).

RBS is related to the simulation of processes occurring in natural wetland ecosystems. The dewatering process takes place mainly by water infiltration through the bed and outflow in the form of reject water and the evapotranspiration of water from above-ground parts of the reeds (Nielsen and Cooper, 2011). This process leads to a reduction in both the volume and mass of the raw material, as well as an increase in the content of dry matter on the level from 14% dm to 32% dm. Stabilization occurs through differences in oxidation-reducing potentials, resulting in mineralization of organic matter as well as nitrogen transformations. The effect of stabilization is to lower the content of organic compounds to the range from 41% dm to 60% dm and the elimination of unpleasant odors (Kołecka, 2019).

The advantages this method also includes:

- the simple construction of RBS (Kołecka and Rohde, 2018) – the simple technical solutions influence its easier availability to laypeople, an easier understanding of the technology, and also a lower risk of expensive repairs during the exploitation period;
- low investment and exploitation costs (Kołecka and Rohde, 2018) – financing is a very important issue for the local authorities responsible for BW management, so the low costs of the construction and maintenance of RBS are a factor favoring the choice of this method, and it is intended to produce a fertilizer that can have an extra, positive influence on the budget of the institution which will decide on such a solution;
- a natural look, which will enable an easy fit into existing landscapes (Sobczyk and Sypuła, 2011) – the natural and easily fitting appearance of RBS in the case of the application BW is a factor increasing its potential. The profitability of this type of method depends on the distance of the raw material collecting site to the processing site, and due to its eye- pleasing appearance, the facility can be built in close proximity to the beach, because it will be a great complement to the seaside landscape;
- lack of the necessity to use chemicals, low emissions and low energy consumption (Obarska-Pempkowiak et. al., 2015) – the processes occurring in RBS are based on natural mechanisms, which do not need, as in the case of mechanical dewatering, the addition of polyelectrolytes to improve the efficiency of the process. The lack of additional foreign substances introduced into the environment provides an eco-friendly solution; this method also consumes energy only for the supply of BW to the beds by pump, and low emissions are associated with a lower production of carbon dioxide than in the case of traditional methods.

Moreover, RBS are a sustainable solution fitting in mitigation trends, i.e. reducing CO_2 emissions. Additionally, they provide many ecosystem functions (Shah et al. (2023), Rosil et al. (2017)).

Thus the article aimed to evaluate the potential of using RBS for processing BW, including its mixture with compost, focusing on dewatering efficiency, organic matter stabilization, and nutrient content. The research explored the system's performance under different feed material conditions to assess its feasibility as a sustainable management approach for coastal organic waste.

MATERIALS AND METHODS

Research facility

The application of RBS technology for the processing of BW represents an innovative approach, evaluated as part of the CONTRA project's "FERTIWRACK" case study (Swarzewo, Puck Bay, Poland). The pilot RBS was built in the autumn of 2019. The facility comprises of two modular units constructed on wooden pallets, with a substrate composed of aggregate, gravel, and sand. The reeds were planted in the upper sand layer (Figure 1). Each modular unit was subdivided into four quarters (cube A-D with quarters: A, B, C, D; cube-H with quarters: E, F, G, H) (Figure 2) with drainage pipes installed at the base of each section. These pipes were responsible for the removal of reject water while simultaneously providing aeration to RBS.

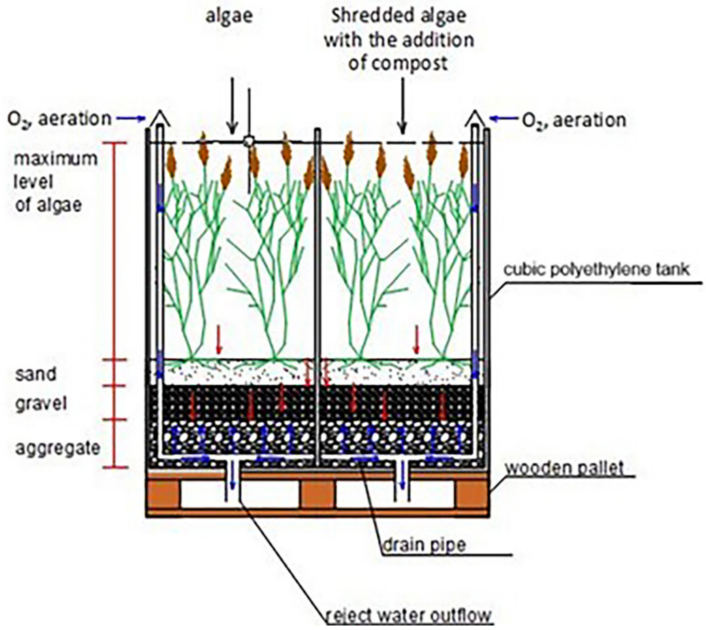

Figure 1. Scheme of one module cube of pilot RBS (Kupczyk's study)

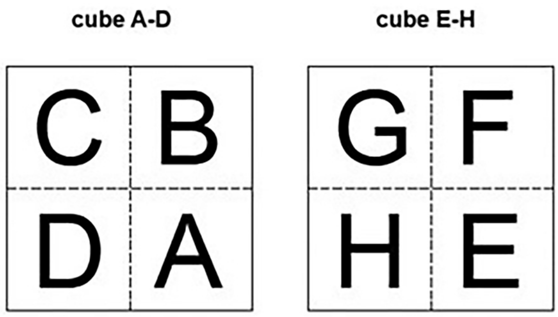

Figure 2. Cubes of pilot RBS subdivided on quarters (Kupczyk's study)

Research material

The research period covers the years 2019–2021. During this time, a different number of supplies with a different type of material were delivered to each quarter of deposits (Table 1). The sections were supplied with either BW (quarters: A, B, D, E, G) or a mixture of BW and compost (quarters: C, F, H). Table 1 presents the details of the supplies performed for each section.

The BW material, intended for evaluating the potential application of RBS in the treatment of BW, was collected from the coastal waters Rzucewo's beach. The collection process was conducted manually using a landing net. The compost that was mixed with algae and used to feed individual quarters came from the aerobic stabilization of sewage sludge from a multi-stage wastewater treatment plant.

Throughout the study, analyses were conducted on both raw beach wrack (RBW) and the material treated in the pilot RBS: processed beach wrack (PBW) was examined in quarters A, B, D, E, and G, while the processed beach wrack mixed with compost (PBW + compost) was analyzed in quarters C, F, and H.

Research methodology

During the study period, measurements were performed for both raw beach wrack (RBW), collected directly from the beach (on a monthly basis during the period of material occurrence), and the material processed in the pilot RBS (on a monthly basis during the feeding period, one month after the feeding of particular quarter): processed beach wrack (PBW) in quarters A, B, D, E, and G, as well as processed beach wrack mixed with compost (PBW + compost) in quarters C, F, and H.

Basic parameters were studied to assess dewatering- determination of dry matter content, and stabilization- determination of the organic matter content of the material. Additionally, the nutrient content was evaluated by measuring Kjeldahl nitrogen and total phosphorus concentrations. The determinations were performed in accordance with the applicable standards presented in Table 2.

The dry matter content in the samples was determined following the PN-EN 12880:2004 standard, which outlines the methodology for

Table 1. Scheme of supplies of raw material for individual sections of cubes A-D and E-H

Quarters	Feeding period	Total number of feedings	Material delivered to quarters	Material delivered to quarters in total	Comments
A	18.11.2019	1	10l BW	40l BW	-
	23.06–30.08.2021	3			
B	27.04–21.09.2020	6	10l BW	100l BW	*23.06.2021, 30.08.2021- supply of 5l BW
	29.04–30.08.2021	5	10l BW*		
C	27.04–21.09.2020	6	10l BW + 10l compost**	65l BW + 65l compost	**21.09.2020- supply of 5l BW + 5l compost
	29.04–17.05.2021	2	5l BW + 5l compost		
D	27.04–21.09.2020	6	15l BW	145l BW + 145l compost	***23.06.2021, 30.08.2021- supply of 5l BW
	29.04–30.08.2021	5	15l BW***		
E	23.06–30.08.2021	3	5l BW	15l BW	-
F	27.04–21.09.2020	6	5l BW + 5l compost	42l BW + 45l compost	****23.06.2021-supply of 2l BW + 5l compost
	29.04–23.06.2021	3	5l BW + 5l compost****		
G	20.08–21.09.2020	2	5l BW	30l BW	-
	27.07–30.08.2021	2	10l BW		
H	20.08–21.09.2020	2	5l BW + 5l compost	20l BW + 20l compost	-
	29.04–27.05.2021	2			

Table 2. Determinations with their standards

Determination	Standard
Dry matter	PN-EN 12880:2004
Organic matter	PN-EN 15935:2022-01
Kjeldahl nitrogen	PN-EN 16169:2012
Total phosphorus	PN-EN 14672:2006

determining the dry matter in sewage sludge. The method for determining dry matter involves a precise drying process of the samples in an oven at a specified temperature and time.

The organic matter was determined following the PN-EN 15935:2022-01 standard, which pertains to the determination of mass loss during the ignition of materials. The mass loss determination process involves measuring the mass of the sample before and after ignition at 550°C. During the ignition process, organic substances are burned off, leaving behind the mineral material.

The Kjeldahl nitrogen determination was carried out according to the PN-EN 16169:2012 standard. The method includes the following steps: sample mineralization in an acidic environment with copper in the presence of sodium sulfate to raise the temperature of mineralization, distillation of the mineralized sample in an alkaline environment, titration: ammonia is quantified in the obtained ammonium sulfate solution. The sum of ammonium and organic (non-nitrate) nitrogen is determined as Kjeldahl nitrogen. The determination of total phosphorus was carried out in accordance with the PN-EN 14672:2006 standard, which pertains to the analysis of phosphorus in water and sediment samples. The method includes the following steps: sample preparation, phosphorus digestion, reaction with ammonium molybdate, color intensity measurement. The phosphorus concentration is proportional to the color intensity.

RESULTS AND DISCUSSION

Load of material

Table 3 presents the data on the load of material: PBW and PBW + compost in the years 2019–2021 for pilot RBS (cube A-D and cube E-H) and individual quarters (A, B, C, D, E, F, G, H). The material load values in individual quarters vary and depend on the feed material: either BW material or a mixture of BW and compost, as well as the number of feedings applied to a particular part of the deposits. The highest total load of material per year was recorded in quarter C (139.2 kg dm/m²·year) in bed A-D. In bed E-H, the highest value was observed in part F (95.2 kg dm/m²·year). The lowest total load of material per year was recorded in quarter A (15.5 kg dm/m²·year) in bed A-D and in bed E-H, the lowest value was observed in parts E and G (11.6

Table 3. Load of material of tested quarters of pilot RBS

Load of material		A	B	C	D	E	F	G	H
2019	[kg dm/m^2]	7.7	0.0	0.0	0.0	0.0	0.0	0.0	0.0
2020	[kg dm/m^2]	0.0	46.4	235.6	69.6	0.0	128.5	7.7	42.8
2021	[kg dm/m^2]	23.2	30.9	42.8	42.5	11.6	61.9	15.5	42.8
Total	[kg dm/m^2]	30.9	77.3	278.4	112.1	11.6	190.4	23.2	85.7
Total per year	[kg dm/m^2 * year]	15.5	38.7	139.2	56.1	11.6	95.2	11.6	42.8

kg dm/m²·year). Higher loads are associated with quarters discharged with the addition of compost, while lower loads are observed in parts of deposits supplied with BW only.

For RBS used in the treatment of sewage sludge, the maximum loading rate for secondary sludge should not exceed 60 kg d.m./m²·year. In the case of sludge with a high fat content or a low sludge age, it is recommended to reduce the loading rate, which should not exceed 50 kg dm/m²·year (Obarska-Pempkowiak et al., 2015). Nielsen (2023) also emphasized that during the start-up phase (approximately 2 years), it is advisable to apply lower doses than the designed ones. This approach allows the plants sufficient time to establish their root systems and achieve better development.

The pilot RBS, designed for processing BW and its mixture with compost, had a short start-up period, as it was established in November 2019, with charging started in April 2020. Until September 2020, quarter C had been supplied with load of material, which amount was 235.6 kg dm/m²·year while maintaining its functionality. This demonstrates that, in the case of BW treatment in RBS, the load of material can be significantly higher than that for sewage sludge processing. Consequently, the total area required for this type of system to process BW will be smaller.

Raw beach wrack

Table 4 presents the minimum, maximum, average, and median values of basic parameters characterizing RBW for years 2020 and 2021. The parameters include dry matter content, organic matter content, Kjeldahl nitrogen content, and total phosphorus content. The table also provides the average, minimum and maximum values for the combined data from analyzed years as well as average together for 2020 and 2021.

In terms of dry matter content, the values ranged between 4.5% and 7.9% in 2020, with an average of 5.6%. In 2021, the dry matter content increased significantly, ranging from 9.5% to 17.0%, with an average of 11.7%. The combined average minimum for 2020–2021 was 7.0%, while the combined average maximum was 12.4%, and the overall average was 8.6%. In the case of the sewage sludge originating from preliminary wastewater treatment, the dry matter content ranges from 5% to 10%, while for secondary sludge after biological deposits, the dry matter content falls within the range of 4% to 8% and after activated sludge the range is between 0.5–3.0% dm (Podedworna and Umiejewska, 2008). RBW exhibits a higher range of dry matter values, however, it still qualifies as a material with high water content, similar to sewage sludge.

Table 4. Values of minimum, maximum, average and median of basic parameters in RBW collected in occurrence period in 2020–2021

RBW											
Year	2020				2021				2020–2021*		
	Min	Max	Average	Median	Min	Max	Average	Median	Average min	Average max	Average
Dry matter [%]	4.5	7.9	5.6	5.3	9.5	17.0	11.7	10.8	7.0	12.4	8.6
Organic matter\ [% dm]	36.1	59.4	51.6	53.7	26.9	75.1	50.6	52.4	31.5	67.3	51.1
Kjeldahl nitrogen [% dm]	1.12	3.05	2.19	2.40	1.53	2.02	1.72	1.74	1.32	2.54	1.96
Total phosphorus [% dm]	0.98	2.34	1.60	1.61	1.57	7.14	3.04	2.14	1.28	4.74	2.32

Note: *average values for 2020 and 2021.

For organic matter content, the 2020 data indicate a range from 36.1% dm to 59.4% dm, with an average of 51.6% dm. In 2021, the range was broader, from 26.9% dm to 75.1% dm, with an average of 50.6% dm. The combined average minimum and maximum values for 2020–2021 were 31.5% dm and 67.3% dm, respectively, with an overall average of 51.1% dm. For sewage sludge, the organic matter content ranges from 60% dm to 70% dm for primary sludge and from 55% dm to 80% dm for secondary sludge (Podedworna and Umiejewska, 2008). The values observed in the analyzed RBW exhibit a lower range compared to those found in sewage sludge. However, BW typically has high variability in dry matter content depending on species composition and environmental conditions,

The Kjeldahl nitrogen content varied between 1.12% dm and 3.05% dm in 2020, with an average of 2.40% dm. In 2021, the range narrowed slightly, from 1.53% dm to 2.02% dm, with an average of 1.74% dm. The combined average minimum and maximum values for 2020–2021 were 1.32% dm and 2.54% dm, respectively, with an overall average of 1.96% dm. The nitrogen content in primary sludge ranges between 2–7% dm, while in secondary sludge, it varies between 1.5–5.0% dm (after biological deposits) and 3–10% dm (after activated sludge) (Podedworna and Umiejewska, 2008). The nitrogen content in the analyzed RBW is significantly lower, placing it at the lower end of the range observed for sewage sludge.

For total phosphorus content, the 2020 data show a range from 0.98% dm to 2.34% dm, with an average of 1.60 dm. In 2021, the range was from 1.57% dm to 7.14% dm, with an average of 3.04% dm. The combined average minimum and maximum values for 2020–2021 were 1.28% dm and 4.74% dm, respectively, with an overall average of 2.32% dm.

For sewage sludge, the phosphorus content ranges from 0.4% to 3.0% dm for primary sludge and from 0.9% to 1.5% dm for secondary sludge (Podedworna and Umiejewska, 2008). The values observed in the RBW demonstrate a higher phosphorus range compared to sewage sludge. The data concerning RBW highlight notable year to year variations in all parameters, with generally higher values for dry matter and total phosphorus content in 2021. This may suggest differing environmental conditions affecting the composition of RBW between the two years. RBW demonstrates higher phosphorus levels compared to sewage sludge. Conversely, the nitrogen content in RBW is lower in comparison to sewage sludge. The dry matter content, although variable, aligns with that of sewage sludge and organic matter is lower than in sewage sludge and it could potentially influence on shorter time of stabilization in RBS.

Processed beach wrack and processed beach wrack mixed with compost in pilot reed bed system

Table 5 presents the dry matter content in PBW and PBW + compost in the pilot RBS across different quarters (A to H) during 2020 and 2021. Quarters C, F, and H were fed with a mixture of BW and compost, while quarters A, B, D, E, and G were supplied with only BW. The data include

Table 5. Dry matter content in material processed in pilot RBS

Dry matter [%]											
Year	2020				2021				2020–2021**		
Quarters	Min	Max	Average	Median	Min	Max	Average	Median	Average min	Average max	Average
A	61.9	98.8	86.8	92.4	21.3	35.3	30.6	35.1	41.6	67.1	58.7
B	26.2	89.3	59.0	57.7	16.8	72.9	42.6	36.5	21.5	81.1	50.8
C	21.4	46.3	33.8	35.5	29.9	78.2	47.7	46.4	25.7	62.3	40.7
D	19.0	90.2	56.2	52.4	23.0	85.6	47.3	31.3	21.0	87.9	51.8
E	l.m	l.m	l.m	l.m	22.2	84.2	45.3	29.5	22.2	84.2	45.3
F	35.0	71.9	53.5	45.9	27.5	69.2	44.0	43.6	31.3	70.6	48.8
G	24.0	84.8	54.4	54.4	22.3	35.4	27.1	23.5	23.2	60.1	40.7
H	25.9	48.1	37.0	48.1	25.4	83.4	58.5	61.4	25.7	65.8	47.8

Note: *l.m.- lack of material, **average values for 2020 and 2021.

minimum, maximum, average, and median values for each year, as well as combined averages for 2020–2021.

Quarter A fed with BW, revealed the highest dry matter content among all quarters, with a two-year average of 58.7%. In 2020, values ranged from 61.9% to 98.8%, with an average of 86.8%. In 2021, the dry matter content decreased significantly, ranging from 21.3% to 35.3%, with an average of 30.6%. Despite the drop, quarter A still outperformed most other quarters, indicating effective dewatering.

Quarter B also fed with BW, demonstrated dry matter content, with a two-year average of 50.8%. In 2020, values ranged from 26.2% to 89.3%, averaging 59.0%. In 2021, values ranged from 16.8% to 72.9%, with an average of 42.6%. Quarter B showed greater variability compared to quarter A.

Quarter C fed with mixture of BW and compost, demonstrated a two-year average dry matter content of 40.7%, slightly lower than most BW-fed quarters. In 2020, values ranged from 21.4% to 46.3%, with an average of 33.8%. In 2021, performance improved, values ranged from 29.9% to 78.2%, with an average of 47.7%.

Quarter D fed with only BW, had a two-year average dry matter content of 51.8%. In 2020, values ranged from 19.0% to 90.2%, averaging 56.2%. In 2021, values ranged from 23.0% to 85.6%, with an average of 47.3%.

Quarter E also fed only with BW, had no data in 2020 due to lack of delivers to this quarter. In 2021, the dry matter content ranged from 22.2% to 84.2%, with an average of 45.3%.

Quarter F fed with BW with addition of compost, exhibited a two-year average dry matter content of 48.8%. In 2020, values ranged from 35.0% to 71.9%, averaging 53.5%. In 2021, values ranged from 27.5% to 69.2%, with an average of 44,0%. Quarter F performed well, with dry matter content comparable to some BW-fed quarters, highlighting the positive impact of compost addition.

Quarter G fed with BW, consistently exhibited the lowest dry matter content among the BW-fed quarters, with a two-year average of 40.7%. In 2020, values ranged from 24.0% to 84.8%, averaging 54.4%. In 2021, values ranged from 22.3% to 35.4%, with an average of 27.1%. The poor performance of Quarter G suggests lower dewatering efficiency compared to other quarters.

Quarter H fed with mix of BW with compost, consistently outperformed other compost-treated quarters, with a two-year average dry matter content of 47.8%. In 2020, values ranged from 25.9% to 48.1%, averaging 37.0%. In 2021, values ranged from 25.4% to 83.4%, with an average of 58,5%.

Sludge dewatered in RBS exhibits varying dry matter content. For instance, the dry matter content achieved in excess activated sludge ranges from 18% to 24% (Uggetti et al., 2009). In the case of excess activated sludge and secondary sludge from sedimentation tanks, the values range from 20% (Nielsen, 2007) to 29% (Kołecka and Obarska-Pempkowiak, 2013). Additionally, the sludge from Imhoff tanks reaches a dry matter content of 42% (Kołecka, 2019). The highest average dry matter content over the two-year dewatering period was observed in the material processed in quarter A (58.7%), while the lowest was recorded in the material from quarters C and G (40.7%). This range indicates a higher dewatering efficiency for the tested material compared to the referenced sewage sludge.

Figure 3 illustrates the changes in dry matter content over time in PBW and PBW +compost in quarters of the pilot RBS and in collected RBW.

The dry matter content of RBW shows variability, with values ranging from a minimum of 4.5% (May and June 2020) to a maximum of 17.0% (June 2021). This low dry matter content highlights the significant water content in the BW. The graph also illustrates that the dry matter content of RBW was lower than that observed in PBW and PBW +compost in any of quarters, which show that effectiveness of the pilot RBS in enhancing dry matter content.

When comparing the quarters supplied with only BW and those supplied with addition of compost, clear differences emerge.

Quarters A, B, D, E, and G, which were supplied with BW, demonstrate greater variability in dry matter content over time. For example, quarter A achieves some of the highest dry matter values, reaching nearly 100% on most sampling dates in 2020, indicating effective dewatering. However, quarter A was fed only once in 2019 and next supply of raw material was in June 2021. In connection with such supply scheme, values of dry matter content noticed in 2020 could be higher than in another quarters. This shows also that longer breaks in the supply of raw material improve dewatering efficiency. However, quarters like B and G exhibit more moderate values, with dry matter content often falling below 60%.

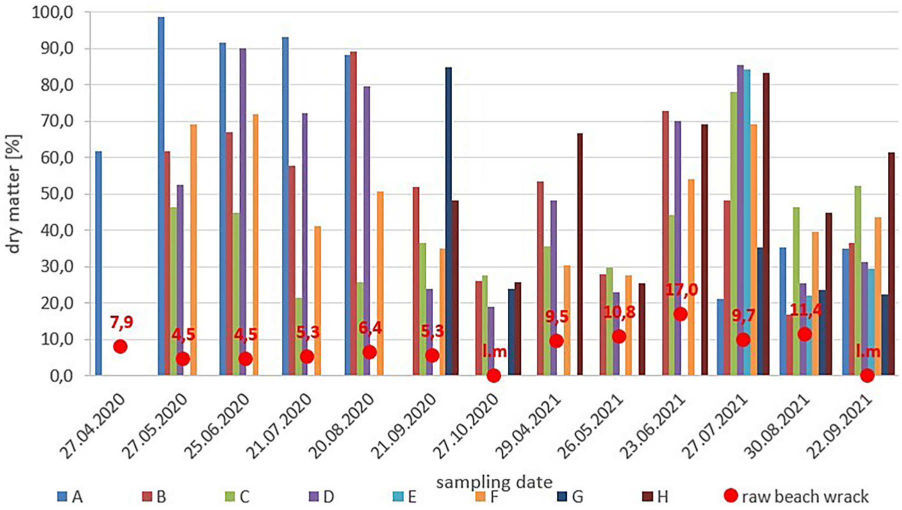

Figure 3. Dry matter content in PBW and PBW + compost vs RBW, *l.m.- lack of material

In contrast, quarters C, F, and H, fed with the mixture of BW and compost, show more balanced values of dry matter. Quarter H, in particular, achieves notably high dry matter content, exceeding 80% on several occasions, which suggests that the addition of compost enhances dewatering efficiency. The material from quarter C, while exhibiting slightly lower dry matter content compared to H, maintains steady values throughout the observation period. These stable values of dry matter across quarters supplied with mixture highlights the improvement of dewatering effect with compost addition.

Overall, the comparison indicates that while quarters fed with BW can achieve high dry matter values, their performance is more variable and less predictable. In contrast, the addition of compost results in more consistent dry matter content. Table 6 presents organic matter content in PBW and PBW + compost in pilot RBS for years 2020–2021.

For quarter A, fed with BW, the organic matter content in 2020 ranged from a minimum of 29.4% dm to a maximum of 54.7% dm, with an average of 42.9% dm. In 2021, the organic matter content decreased, ranging between 23.7% dm and 60,5% dm, with an average of 38.0% dm. Over the combined period (2020–2021), quarter A showed an average minimum organic matter content of 26,6% dm and an average maximum of 57.6% dm, resulting in an overall average of 40.5% dm.

Quarter B, also supplied with only BW, organic matter content in 2020, ranging from 9.7% dm to 59.0% dm, with an average of 35.0% dm. In 2021, the values were higher, ranged between 32.4% dm and 67.3% dm, with an average of 50.4% dm. Over the two years, quarter B demonstrated an average minimum organic matter of 21.1% dm, an average maximum of 63.2% dm, and an overall average of 42.7% dm.

Table 6. Organic matter content in PBW and PBW + compost in pilot RBS

Organic matter [% dm]											
Year	2020				2021				2020–2021**		
Quarters	Min	Max	Average	Median	Min	Max	Average	Median	Average min	Average max	Average
A	29.4	54.7	42.9	43.9	23.7	60.5	38.0	29.9	26.6	57.6	40.5
B	9.7	59.0	35.0	30.7	32.4	67.3	50.4	46.9	21.1	63.2	42.7
C	28.4	60.6	49.6	55.4	33.2	56.7	49.6	52.0	30.8	58.7	49.6
D	13.5	63.5	36.7	37.0	35.2	78.5	55.2	47.6	24.4	71.0	45.9
E	l.m	l.m	l.m	l.m	41.4	65.1	54.8	57.8	41.4	65.1	54.8
F	23.6	62.5	47.6	51.8	36.5	54.9	46.6	47.0	30.1	58.7	47.1
G	15.2	56.2	35.7	35.7	33.3	56.0	46.6	50.6	24.3	56.1	41.2
H	55.0	57.7	56.4	56.4	29.8	64.9	49.9	53.6	42.4	61.3	53.1

Note: *l.m.- lack of material, **average values for 2020 and 2021.

Quarter C, fed with a mixture of BW and compost, in 2020 exhibited the organic matter content with values ranging from 28.4% dm to 60,6% dm, an average of 49.6% dm. In 2021, the range shifted slightly, with a minimum of 33.2% dm and a maximum of 56.7% dm and an average of 49.6% dm. Across 2020–2021, quarter C recorded an average minimum of 30.8% dm, an average maximum of 58.7% dm, and an overall average of 49.6% dm.

Quarter D, supplied with only BW, in 2020 showed the organic matter content with values ranging from 13.5% dm to 63.5% dm and an average of 36.7% dm. In 2021, the organic matter content increased, ranging between 35.2% dm and 78.5% dm, with an average of 55.2% dm. Over the two years, quarter D showed an average minimum organic matter of 24.4% dm, an average maximum of 71.0%, dm and an overall average of 45.9% dm.

Quarter E, also fed with BW, in 2020 shows no data due to lack of supplies of this part of deposit. In 2021, organic matter content ranged between 41.4% dm and 65.1% dm, with an average of 54.8% dm.

Quarter F, supplied with the mixture of BW and compost, had the lowest organic matter content in 2020, ranging from 23.6% dm to 62.5% dm, with an average of 47.6% dm. In 2021, the organic matter ranged from 36.5% dm to 54.9% dm, with an average of 46.6% dm. Across the two years, quarter F recorded an average minimum of 30.1% dm, an average maximum of 58.7% dm, and an overall average of 47.1% dm.

Quarter G, supplied with BW, demonstrated organic matter content in 2020, ranging from 15.2% dm to 56.2% dm, with an average of 35.7% dm. In 2021, the values increased, ranging between 33.3% dm and 56.0% dm, with an average of 46.6% dm. Over 2020–2021, quarter G exhibited an average minimum organic matter of 24.3% dm, an average maximum of 56.1% dm, and an overall average of 41.2% dm.

Quarter H, also fed with a mixture of BW and compost, showed organic matter content in 2020, ranging between 55.0% dm and 57.7% dm, with an average of 56.4% dm. In 2021, the range was wider to 29.5% dm to 64.9% dm, with an average of 49.9% dm. Over the two years, quarter H recorded an average minimum of 42.4% dm, an average maximum of 61.3% dm, and an overall average of 53.1% dm.

In RBS for processing sewage sludge, a reduction in organic matter concentration, indicative of ongoing stabilization, is observed at a level of 25–30% over a period of 10–12 years. The final concentration of organic matter in the accumulated sludge typically ranges between 40–50% dm. Values above this range indicate either an insufficient retention time of the sludge within the system or improper system operation. For example, the organic matter content in primary sludge and excess sludge from biological deposits is 52% dm (Kołecka, 2019). In excess activated sludge from chambers and secondary settling tanks, the content ranges from 41% dm (Nielsen, 2007) to 42% dm (Kołecka and Obarska-Pempkowiak, 2013). In excess activated sludge, the organic matter content varies between 41% dm and 48% dm (Uggetti et al., 2009), while in sludge from the Imhoff tanks, it reaches 45% dm (Kołecka, 2019). The two-year average organic matter content for the PBW and PBW + compost in the pilot RBS ranges from 40.5% dm (quarter A) to 54.8% dm (quarter E). It is important to note that the processing period in the pilot RBS is relatively short compared to facilities used for sewage sludge treatment. The stabilization efficiency increases with longer retention times of the material in the bed.

The graph in Figure 4 illustrates the changes in organic matter content in PBW and PBW + compost from individual quarters (A, B, C, D, E, F, G, H) and RBW over the sampling period. RBW, marked with red dots, demonstrates substantial variability in organic matter content across the sampling dates, with values ranging from 26.9% dm to 75.1% dm. The highest organic matter content of 75.1% dm was recorded in April 2021, while the lowest value of 26.9% dm was observed in May 2021. These fluctuations underline the variability in the composition of the raw material, which influenced the stabilization process in treatment quarters. For quarters supplied with BW, the organic matter content exhibited varying trends depending on the quarter. Quarters B and D showed stable levels of organic matter, with occasional peaks, such as in quarter D around May 2021. This increase may be related to the addition of RBW, which contained a high organic matter content (above 75% dm). Quarter E, where data only for 2021 were available, had similar situation. When RBW with organic matter was 62.4% dm, was added to quarter, the organic matter in material after month processing (August 2021) achieve

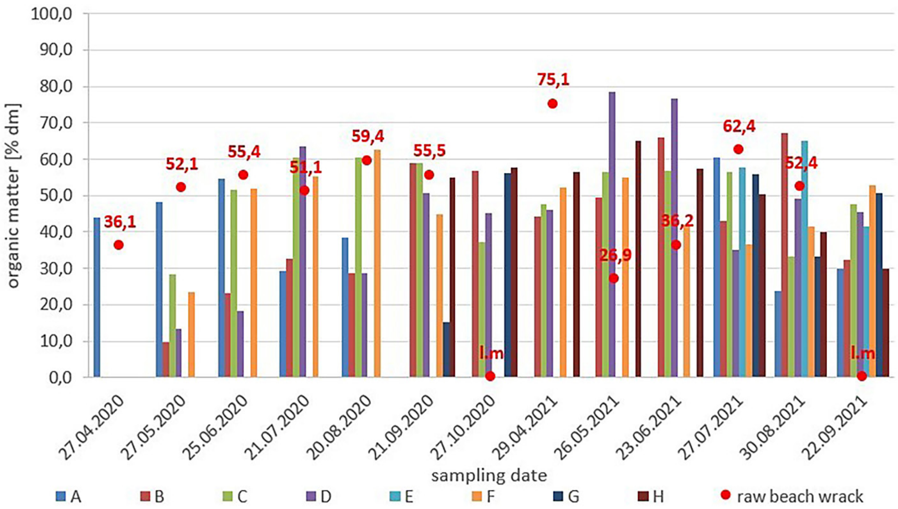

Figure 4. Organic matter content in PBW and PBW + compost vs RBW, *l.m.- lack of material

higher content of this parameter than in the previous month. This shows that stabilization becomes more effective after longer time.

In contrast, the quarters fed with the mixture of BW and compost (C, F, H) demonstrated generally higher organic matter content. The quarters fed with mixture show the compost addition is promoting organic matter retention. In summary, the addition of compost to BW in quarters C, F, and H increases the organic matter content compared to the quarters supplied with BW, thus reducing the stabilization of the material in the quarters with addition of compost. The variability in RBW composition also had influence on processing outcomes. It should be emphasized that stabilization requires appropriate time. Table 7 presents the Kjeldahl nitrogen content in the PBW and PBW + compost in the pilot RBS for each quarter (A, B, C, D, E, F, G, H) during 2020 and 2021.

In quarter A, supplied with BW, the average Kjeldahl nitrogen content in 2020 was 2.19% dm and in 2021 was 1.61% dm, resulting in an overall average of 1.90% dm for 2020–2021. The nitrogen content in this quarter showed limited variability, with average minimum and maximum values of 1.77% dm and 2.08% dm, over the two years.

Quarter B, also fed with BW, exhibited slightly higher nitrogen levels than quarter A, with a two-year average of 2.01% dm. The nitrogen content ranging from an average minimum of 1.19% dm to a maximum of 2.59% dm.

Quarter C, supplied with a mixture of BW and compost, consistently displayed higher Kjeldahl nitrogen levels compared to quarters A and B. The two-year average nitrogen content was 2.53% dm, with range of average minimum (1.80% dm) and maximum (2.94% dm) values. The higher nitrogen levels were connected with addition of compost.

Table 7. Content of Kjeldahl nitrogen in PBW and PBW + compost in pilot RBS

Kjeldah nitrogen [% dm]											
Year	2020				2021				2020–2021**		
Quarters	Min	Max	Average	Median	Min	Max	Average	Median	Average min	Average max	Average
A	2.05	2.48	2.19	2.18	1.49	1.69	1.61	1.65	1.77	2.08	1.90
B	0.64	2.80	1.85	1.93	1.73	2.38	2.17	2.21	1.19	2.59	2.01
C	2.05	2.88	2.66	2.82	1.56	2.99	2.39	2.40	1.80	2.94	2.53
D	0.66	2.44	1.80	2.02	1.66	2.40	2.08	2.09	1.16	2.42	1.94
E	l.m	l.m	l.m	l.m	1.97	5.62	3.23	2.10	1.97	5.62	3.23
F	1.19	2.81	2.13	2.67	2.17	2.78	2.51	2.54	1.68	2.79	2.32
G	0.64	2.86	1.75	1.75	1.91	3.73	2.62	2.21	1.28	3.30	2.19
H	2.62	2.70	2.66	2.66	1.99	3.77	2.77	2.61	2.30	3.23	2.71

Note: *l.m.- lack of material, **average values for 2020 and 2021.

Quarter D, another quarter fed with BW, showed a two-year average nitrogen content of 1.94% dm. The nitrogen levels exhibited variability, with the lowest average minimum (1.16% dm) among all quarters and an average maximum of 2.42% dm. The results were comparable to other quarters supplied with BW.

Quarter E, demonstrated high nitrogen levels in 2021, with an average of 3.23% dm. This was the highest recorded value for all quarters, suggesting a different behavior as in quarters fed with only BW.

Quarter F, fed with the mixture of BW and compost, recorded high nitrogen content, with a two-year average of 2.32% dm. The average minimum (1.68% dm) and maximum (2.79% dm) values were slightly lower than quarter C but higher than most quarters supplied with BW. This underscores the positive impact of compost addition on nitrogen content.

Quarter G, supplied with BW, showed two-year average of 2.19% dm. The nitrogen content ranged from an average minimum of 1.28% dm to a maximum of 3.30% dm.

Quarter H, another quarter fed with the mixture of BW and compost, exhibited the two-year average of 2.71% dm. The average minimum and maximum values were 2.30% dm and 3.23% dm, respectively, indicating effective nitrogen retention due to the compost addition.

In summary, the quarters supplied with a mixture of BW and compost (C, F, H) consistently demonstrated higher Kjeldahl nitrogen content compared to those fed with BW (A, B, D, E, G). Among the quarters with addition of compost, quarter H showed the highest nitrogen levels, highlighting the positive impact of compost addition on nitrogen levels in the RBS. Conversely, quarter A exhibited the lowest average nitrogen for 2020–2021, confirming the lower nitrogen content in the BW material itself than in the mixture with compost. In the case of sewage sludge, the nutrient content depends on the type of wastewater and the applied treatment processes. Exemplary nitrogen concentrations observed in RBS used for processing sewage sludge are as follows: Denmark: 2.6–3.6% dm (Nielsen and Bruun, 2015), Spain: 3.0–3.3% dm (Uggetti et al., 2009) and Poland: 1–10% dm (Kołecka et al., 2018). Among the analyzed quarters, the highest two-year average nitrogen content was observed in the material processed in quarter E (3.23% dm), while the lowest was recorded in quarter A (1.90% dm). These values fit with the nitrogen concentrations typically found in stabilized sewage sludge processed in RBS.

Figure 5 illustrates changes in Kjeldahl nitrogen content in RBW and PBW and PBW + compost in particular quarters of the pilot RBS. The RBW, represented by red markers, displayed fluctuating Kjeldahl nitrogen levels across sampling dates, with values ranging from a low of 1.12% in April 2020 to a peak of 3.05% in July 2020.

For the quarters fed with BW, variability in Kjeldahl nitrogen content is noticeable. Quarter A demonstrated relatively low nitrogen levels throughout the study period, with values generally below 2.5% dm. Quarter B followed a similar pattern, with slightly higher nitrogen content in some sampling periods. Quarter E, with limited data, showed one pick in nitrogen level in August

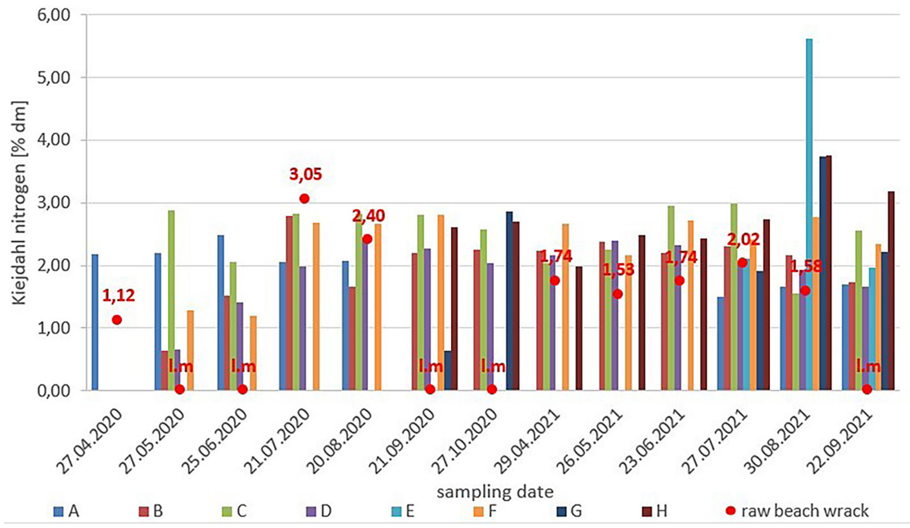

Figure 5. Kjeldahl nitrogen content in PBW and PBW + compost vs RBW, *l.m.- lack of material

2021, it could be connected with to the low number of feedings in the quarter- low load of material, which may promote nitrogen accumulation. A high hydraulic load enhanced the total nitrogen removal efficiency (Tan et al., 2024). Low load of material could resulted in slower development of nitrogen-removing microorganisms as well as weaker growth of reeds, leading to reduced nitrogen uptake. In contrast, the quarters supplied with the mixture of BW and compost (C, F, H) demonstrated higher nitrogen levels compared to those fed with BW. Quarter C showed relatively stable nitrogen content. Quarter F maintained moderate nitrogen levels, which were generally higher than in the quarters fed with only BW. Quarter H recorded the highest nitrogen values among all quarters.

Overall, the graph underscores the influence of compost addition higher content of nitrogen in quarters of the RBS. These findings support the use of compost as an effective amendment for enhancing nitrogen levels in PBW and PBW + compost. Table 8 presents the total phosphorus content in PBW and PBW + compost of the RBS for the years 2020 and 2021.

The phosphorus levels in quarter A remained low and consistent across both years, with a two-year average of 1.23% dm. In 2020, phosphorus ranged from 0.94% dm to 1.43% dm (average 1.5% dm), increasing slightly in 2021 to a range of 1.26% dm–1.38% dm (average 1.32% dm). This quarter exhibited limited phosphorus retention compared to others.

Quarter B showed higher phosphorus levels than quarter A, with a two-year average of 2.26% dm. In 2020, phosphorus content ranged from 0.89% dm to 2.29% dm (average 1.42% dm). In 2021, the range increased slightly to 1.04–10.5% dm (average 3.10% dm).

The phosphorus levels in quarter C were substantially higher than in quarters fed with BW, with a two-year average of.10% dm. In 2020, values ranged from 0.39% dm to 3.06% dm (average 1.99% dm), while in 2021, the range increased dramatically to 1.76–47.24% dm (average 12.22% dm). The addition of compost significantly enhanced the phosphorus content. In 2020, the reed took up phosphorus for its growth needs, and in 2021 there was an accumulation of phosphorus in the PBW + compost.

Quarter D demonstrated with a two-year average of 2.66% dm. In 2020, values ranged from 0.51% dm to 2.33% dm (average 1.54% dm), increasing significantly in 2021 to 1.61–13.13% dm (average 3.78% dm). Quarter D showed improved performance compared to quarters A and B.

Quarter E in 2021, phosphorus levels ranged from 1.31% dm to 1.73% dm (average 1.58% dm).

Quarter F exhibited consistent and elevated phosphorus levels, with a two-year average of 12.8% dm. In 2020, phosphorus ranged from 0.95% dm to 2.65% dm (average 1.88% dm), while in 2021, levels increased to 20.43–24.92% dm (average 22.67% dm). This indicates high efficiency in phosphorus retention due to the compost addition.

Quarter G exhibited the lowest phosphorus levels among quarters supplied with only BW with a two-year average of 1.00% dm. In 2020, values ranged from 0.39% dm to 2.58% dm (average 1.48% dm), while in 2021, the range decreased to 0.41–0.61% dm (average 0.52%). This quarter showed poor phosphorus retention.

Quarter H recorded a two-year average of 12.26% dm. In 2020, values ranged from 1.29% dm to 3.76% dm (average 2.53% dm), while in 2021, the range increased significantly to 17.55–37.33% dm (average 21.99% dm).

Table 8. Content of total phosphorus in PBW and PBW + compost in pilot RBS

Total phosphorus [% dm]											
Year	2020				2021				2020–2021**		
Quarters	Min	Max	Average	Median	Min	Max	Average	Median	Average min	Average max	Average
A	0.94	1.43	1.15	1.13	1.26	1.38	1.32	1.32	1.10	1.41	1.23
B	0.89	2.29	1.42	1.28	1.04	10.05	3.10	1.92	0.96	6.17	2.26
C	0.39	3.06	1.99	2.02	1.76	47.24	12.22	2.23	1.07	25.15	7.10
D	0.51	2.33	1.54	1.69	1.61	13.13	3.78	2.00	1.06	7.73	2.66
E	l.m	l.m	l.m	l.m	1.31	1.73	1.58	1.71	1.31	1.73	1.58
F	0.95	2.65	1.88	1.69	20.43	24.92	22.67	22.67	10.69	13.78	12.28
G	0.39	2.58	1.48	1.48	0.41	0.61	0.52	0.54	0.40	1.59	1.00
H	1.29	3.76	2.53	2.53	17.55	37.33	21.99	19.25	9.42	20.54	12.26

Note: *l.m.- lack of material, **average values for 2020 and 2021.

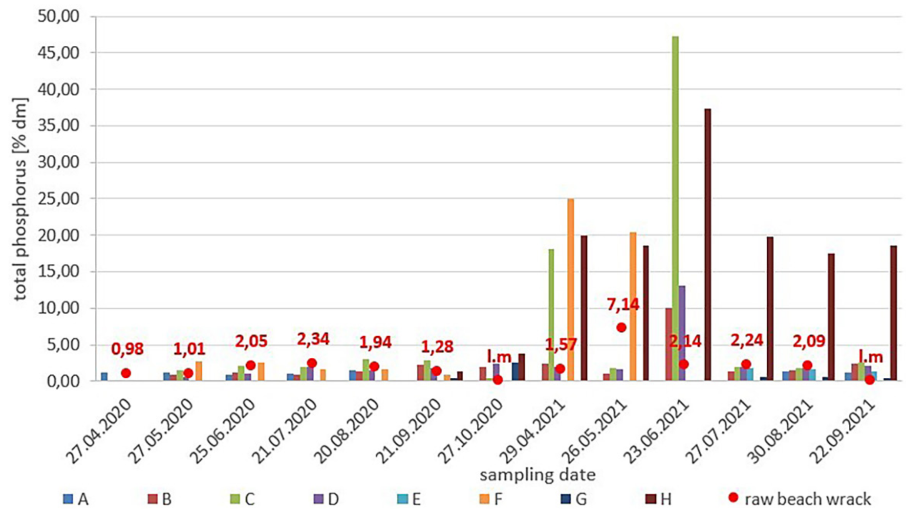

Figure 6. Total phosphorus content in PBW and PBW + compost vs RBW, *l.m.- lack of material

The addition of compost resulted in outstanding phosphorus retention.

The quarters fed with the mixture of BW and compost (C, F, H) showed higher phosphorus content compared to quarters fed with BW (A, B, D, E, G). Compost addition appears to enhance the phosphorus content in PBW + compost. The phosphorus content in stabilized sewage sludge processed in RBS varies by country, with reported ranges of 3.4–4.1% dm in Denmark (Nielsen and Bruun, 2015), 0.07–0.14% dm in Spain (Uggetti et al., 2009), and 0.2–1% dm in Poland (Kołecka et al., 2018). In comparison, the two-year average phosphorus content in the PBW and PBW + compost was significantly higher, ranging from 1.0% dm (quarter G) to 12.28% dm (quarter F). The higher phosphorus concentrations observed in the pilot RBS can be attributed to the addition of compost to the BW material.

Figure 6 illustrates changes in total phosphorus content in PBW and PBW + compost from different quarters of the pilot RBS and in RBW over the study period.

The total phosphorus content in RBW, represented by red marks, varied across sampling dates. The lowest value of 0.98% dm was observed in April 2020, while the highest value of 7.14% was recorded in May 2021. These fluctuations reflect the natural variability in phosphorus levels in the input material, which could influence the concentration of phosphorus in PBW and PBW + compost in pilot RBS.

In the quarters supplied with BW, the phosphorus levels were generally lower compared to those treated with the addition of compost. For instance, in quarter A, the phosphorus content remained relatively stable and low, consistently below 1.5% dm. Similar trends were observed in quarter B, where the phosphorus levels occasionally peaked but remained below 2.5% dm. Quarter D exhibited slightly higher variability, with occasional peaks in the phosphorus content, but overall, its values were comparable to those of quarters A and B. Quarter G consistently recorded some of the lowest phosphorus levels. The quarters treated with a mixture of BW and compost (C, F, H) showed significantly higher phosphorus content. Quarter C recorded the highest phosphorus levels among all quarters, peaking at 47.24% dm in June 2021. Quarter F showed elevated phosphorus levels compared to the only BW quarters, with notable peaks on certain sampling dates. Quarter H displayed a similar trend, maintaining moderate to high phosphorus content throughout the study period.

In summary, the addition of compost to BW in quarters C, F, and H substantially enhanced the phosphorus content compared to the quarters treated with BW.

CONCLUSIONS

Many of the research results obtained indicate potential advantages of using RBS for BW processing. The pilot RBS demonstrated high efficiency in processing BW and BW + compost, significantly increasing dry matter content, which directly leads to a reduction in the volume and mass of PBW and PBW + compost, facilitating further management of this material.

In the case of stabilizing organic matter further studies are necessary. However, as results from other research, over a prolonged operational period, RBS effectively stabilizes the processed material by reducing its organic matter content, thereby creating the potential for its utilization, for example, in agriculture.

The quarters supplied with a mixture of BW and compost exhibited higher concentration of nutrients, particularly for phosphorus, compared to quarters fed only with BW. However, the addition of compost enhanced the organic matter in PBW + compost, reducing the effectiveness of material stabilization. Given its nitrogen and phosphorus content, as well as its possible agricultural applications, the processed material holds promise as a valuable resource, that could generate not only fertilization benefits related to the reuse of nutrients, but also financial benefits related to the sale of fertilizer.

The variability in RBW composition influenced processing results. Preliminary studies have shown that the RBS used for processing BW and its mixture with compost, compared to the RBS processing sewage sludge, can operate effectively under significantly higher loads, i.e. 139.2 kg dm/m²·year (quarter C), therefore, the system area can be smaller than when using RBS for sewage sludge processing, which reduces area requirements and investment costs.

RBS offers an environmentally friendly, cost-effective, and sustainable approach to BW management, contributing to nutrient recycling and providing alternative to landfill disposal.

The application of RBS fits in circular economy goals by transforming organic waste into valuable resource, supporting sustainable coastal waste management strategies.

However, use of RBS also has some disadvantages that should be taken into account when choosing the BW processing technology. Firstly, RBS requires adequate space for proper operation, which may be a limitation for large-scale implementation in areas with limited available land. Secondly, newly established RBS requires time for reed growth and microbial colonization, delaying full operational efficiency. Thirdly, for proper functioning of RBS, appropriate management of the system is needed, including BW loads, which requires appropriate knowledge of the facility staff.

To sum up, RBS presents a sustainable and cost-effective alternative to traditional processing methods, with significant advantages in nutrient recovery, energy efficiency, and long-term operational stability. However, it requires sufficient land, proper loading BW management and time to achieve fully operational period to maintain optimal performance. Understanding these limitations allows for improved system design and operational strategies to maximize the benefits of RBS in BW processing.

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REFERENCES

1. Alobwede, E., Leake, J. R., & Pandhal, J. (2019). Circular economy fertilization: Testing micro and macro algal species as soil improvers and nutrient sources for crop production in greenhouse and field conditions. *Geoderma, 334*, 113–123. https://doi.org/10.1016/j.geoderma.2018.07.049
2. Barreiro, F., Gómez, M., Lastra, M., López, J., & de la H, R. (2011). Annual cycle of wrack supply to sandy beaches: Effect of the physical environment. *Marine Ecology Progress Series, 433*, 65–74. https://doi.org/10.3354/meps09130
3. Björk, M., Rosenqvist, G., & Gröndahl, F. (2023). Methane emissions from macrophyte beach wrack on Baltic seashores. *Ambio, 52*, 171–181. https://doi.org/10.1007/s13280-022-01774-4
4. Boudouresque, C. F., Pergent, G., Pergent-Martini, C., Ruitton, S., Thibaut, T., & Verlaque, M. (2016). The necromass of the Posidonia oceanica seagrass meadow: fate, role, ecosystem services and vulnerability. *Hydrobiologia, 781*, 25–42. https://doi.org/10.1007/s10750-015-2333-y
5. Braeckman, U., Pasotti, F., Vázquez, S., Zacher, K., Hoffmann, R., Elvert, M., Marchant, H., Buckner, C., Quartino, M. L., Mác Cormack, W., Soetaert, K., Wenzhöfer, F., & Vanreusel, A. (2019). Degradation of macroalgal detritus in shallow coastal Antarctic sediments. *Limnology and Oceanography, 64*, 1423–1441. https://doi.org/10.1002/lno.11125
6. Butt, K. R., Méline, C., & Pérès, G. (2020). Marine

macroalgae as food for earthworms: growth and selection experiments across ecotypes. *Environmental Science and Pollution Research, 27*, 33493–33499. https://doi.org/10.1007/s11356-020-07666-y

7. Chubarenko, B., Woelfel, J., Hofmann, J., Aldag, S., Beldowski, J., Burlakovs, J., Garrels, T., Gorbunova, J., Guizani, S., Kupczyk, A., Kotwicki, L., Domnin, D., Gajewska, M., Hogland, W., Kołecka, K., Nielsen, J., & Schubert, H. (2021). Converting beach wrack into a resource as a challenge for the Baltic Sea (an overview). *Ocean & Coastal Management, 200*, 105413. https://doi.org/10.1016/j.ocecoaman.2020.105413
8. Colombini, I., Chelazzi, L., Gibson, R. N., & Atkinson, R. J. A. (2003). Influence of marine allochthonous input on sandy beach communities. *Oceanography and Marine Biology Annual Review, 41*, 115–159.
9. Coupland, G. T., Duarte, C. M., & Walker, D. I. (2007). High metabolic rates in beach cast communities. *Ecosystems, 10*, 1341–1350. https://doi.org/10.1007/s10021-007-9102-3
10. European Union. (2006). Directive 2006/7/EC of the European Parliament and of the Council of 15 February 2006 concerning the management of bathing water quality. https://eur-lex.europa.eu/legal-content/PL/TXT/?uri=CELEX%3A02006L0007-20140101
11. European Union. (2018). Directive (EU) 2018/850 of the European Parliament and of the Council of 30 May 2018. https://eur-lex.europa.eu/eli/dir/2018/850/oj?locale=pl
12. Gómez, M., Barreiro, F., López, J., Lastra, M., & de la Huz, R. (2013). Deposition patterns of algal wrack species on estuarine beaches. *Aquatic Botany, 105*, 25–33. https://doi.org/10.1016/j.aquabot.2012.12.001
13. Guerrero-Meseguer, L., Veiga, P., & Rubal, M. (2020). Spatio-temporal variability of anthropogenic and natural wrack accumulations along the driftline: marine litter overcomes wrack in the northern sandy beaches of Portugal. *Journal of Marine Science and Engineering, 8*, 966. https://doi.org/10.3390/jmse8120966
14. Guerrero-Meseguer, L., Veiga, P., & Rubal, M. (2023). Spatio-temporal variability of wrack along the northern Portuguese sandy beaches. *Estuaries and Coasts, 46*, 818–828. https://doi.org/10.1007/s12237-023-01176-z
15. Illera-Vives, M., López-Fabal, A., López-Mosquera, M. E., & Ribeiro, H. M. (2015). Mineralization dynamics in soil fertilized with seaweed–fish waste compost. *Journal of the Science of Food and Agriculture, 95*, 3047–3054. https://doi.org/10.1002/jsfa.7207
16. Illera-Vives, M., Seoane Labandeira, S., & López-Mosquera, M. E. (2013). Production of compost from marine waste: Evaluation of the product for use in ecological agriculture. *Journal of Applied Phycology, 25*, 1395–1403. https://doi.org/10.1007/s10811-013-9997-3
17. Kołecka, K. (2019). *Usuwanie zanieczyszczeń i stabilizacja osadów ściekowych w systemach trzcinowych*. [Removal of pollutants and stabilization of sewage sludge in reed systems.] Wydawnictwo Polskiej Akademii Nauk, Monografie Komitetu Inżynierii Środowiska 149.
18. Kołecka, K., Obarska-Pempkowiak, H., & Gajewska, M. (2018). Polish experience in operation of sludge treatment reed beds. *Ecological Engineering, 120*, 405–410. https://doi.org/10.1016/j.ecoleng.2018.06.022
19. Kołecka, K., & Obarska-Pempkowiak, H. (2013). Potential fertilizing properties of sewage sludge treated in the Sludge Treatment Reed Beds (STRB). *Water Science and Technology, 68*(6), 1412–1418. https://doi.org/10.2166/wst.2013.393
20. Kołecka, K., & Rohde, D. (2018). Zalety i problemy związane z zagospodarowaniem osadów ściekowych metodą trzcinową [Advantages and problems related to sewage sludge management with the reed bed system method]. *Rynek Instalacyjny, 5*, 20–25.
21. Kupczyk, A., Kołecka, K., & Gajewska, M. H. (2019). Solving the beach wrack problems by on-site treatment with reed beds towards fertilizer amendments. *Journal of Ecological Engineering, 20*(8), 252–261. https://doi.org/10.12911/22998993/111717
22. Lastra, M., López, J., & Rodil, I. F. (2018). Warming intensifies CO_2 flux and nutrient release from algal wrack subsidies on sandy beaches. *Global Change Biology, 24*, 3766–3779. https://doi.org/10.1111/gcb.14278
23. Lastra, M., Rodil, I. F., Sánchez-Mata, A., García-Gallego, M., & Mora, J. (2014). Fate and processing of macroalgal wrack subsidies in beaches of Deception Island, Antarctic Peninsula. *Journal of Sea Research, 88*, 1–10. https://doi.org/10.1016/j.seares.2013.12.011
24. López, B. A., Macaya, E. C., Jeldres, R., Valdivia, N., Bonta, C. C., Tala, F., & Thiel, M. (2019). Spatio-temporal variability of strandings of the southern bull kelp Durvillaea antarctica (Fucales, Phaeophyceae) on beaches along the coast of Chile—Linked to local storms. *Journal of Applied Phycology, 31*, 2159–2173. https://doi.org/10.1007/s10811-018-1705-x
25. Macreadie, P. I., Trevathan-Tackett, S. M., Baldock, J. A., & Kelleway, J. J. (2017). Converting beachcast seagrass wrack into biochar: a climate-friendly solution to a coastal problem. *Science of the Total Environment, 574*, 90–94. https://doi.org/10.1016/j.scitotenv.2016.09.021
26. Nielsen, S. (2007). Helsinge sludge reed bed

systems: Reduction of pathogenic microorganisms. *Water Science and Technology, 56*(3), 175–182. https://doi.org/10.2166/wst.2007.491

27. Nielsen, S., & Bruun, E. W. (2015). Sludge quality after 10–20 years of treatment in reed bed systems. *Environmental Science and Pollution Research, 22*(17), 12885–12891. https://doi.org/10.1007/s11356-014-3815-6
28. Nielsen, S., & Cooper, D. J. (2011). Dewatering sludge originating in water treatment works in reed bed systems. *Water Science & Technology, 64*(2), 361–366. https://doi.org/10.2166/wst.2011.581
29. Obarska-Pempkowiak, H., Kołecka, K., Buchholtz, K., & Gajewska, M. (2015). Ekoinżynieria w zintegrowanym odwadnianiu i stabilizacji osadów ściekowych w systemach trzcinowych [Eco-engineering in integrated dewatering and stabilization of sewage sludge in the reed bed systems]. *Przemysł Chemiczny, 94*(12), 2299–2303.
30. Orr, M., Zimmer, M., Jelinski, D. E., & Mews, M. (2005). Wrack deposition on different beach types: spatial and temporal variation in the pattern of subsidy. *Ecology, 86*(6), 1496–1507. https://doi.org/10.1890/04-1486
31. Podedworna, J., & Umiejewska, K. (2008). Technologia osadów ściekowych [Sewage sludge technology]. *Oficyna Wydawnicza Politechniki Warszawskiej, 228*.
32. Robbe, E., Woelfel, J., Balčiūnas, A., & Schernewski, G. (2021). An impact assessment of beach wrack and litter on beach ecosystem services to support coastal management at the Baltic Sea. *Environmental Management, 68*, 835–859. https://doi.org/10.1007/s00267-021-01533-3
33. Rosli, F. A., Lee, K. E., Goh, C. T., Mokhtar, M., Latif, M. T., Goh, T. L., & Simon, N. (2017). The use of constructed wetlands in sequestrating carbon: An overview. *Nature Environment & Pollution Technology, 16*(3), 893–899.
34. Ruiz-Delgado, M. C., Reyes-Martínez, M. J., Sánchez-Moyano, J. E., López-Pérez, J., & García-García, F. J. (2015). Distribution patterns of supralittoral arthropods: Wrack deposits as a source of food and refuge on exposed sandy beaches (SW Spain). *Hydrobiologia, 742*, 205–219. https://doi.org/10.1007/s10750-014-1986-2
35. Shah, A. M., Liu, G., & Chen, Y. (2023). Urban constructed wetlands: Assessing ecosystem services and disservices for safe, resilient, and sustainable cities. *Frontiers of Engineering Management, 10*, 582–596. https://doi.org/10.1007/s42524-023-0268-y
36. Sobczyk, R., & Sypuła, M. (2011). Wykorzystanie makrofitów do przetwarzania osadów ściekowych na mursz [The use of macrophytes for processing sewage sludge into muck]. *Forum Instalatora, 4*, 46–48.
37. Song, Z., Shu, Q., Bao, Y., Yin, X., & Qiao, F. (2015). The prediction on the 2015/16 El Niño event from the perspective of FIO-ESM. *Acta Oceanologica Sinica, 34*(6), 67–71. https://doi.org/10.1007/s13131-015-0734-1
38. Tan, Y. Y., Huong, Y. Z., Tang, F. E., & Saptoro, A. (2024). A review of sewage sludge dewatering and stabilisation in reed bed system: Towards the process-based modeling. International *Journal of Environmental Science and Technology, 21*, 997–1020. https://doi.org/10.1007/s13762-023-05063-9
39. Uggetti, E., Llorens, E., Pedescoll, A., Ferrer, I., Castellnou, R., & García, J. (2009). Sludge dewatering and stabilization in drying reed beds: Characterization of three full-scale systems in Catalonia, Spain. *Bioresource Technology, 100*, 3882–3890. https://doi.org/10.1016/j.biortech.2009.03.047
40. Villares, R., Fernández-Lema, E., & López-Mosquera, M. E. (2016). Evaluation of beach wrack for use as an organic fertilizer: Temporal survey in different areas. *Thalassas: International Journal of Marine Sciences, 32*, 19–36. https://doi.org/10.1007/s41208-015-0003-5
41. Wang, M., Chen, L., Li, Y., Chen, L., Liu, Z., Wang, X., Yan, P., Qin, S. (2018). Responses of soil microbial communities to a short-term application of seaweed fertilizer revealed by deep amplicon sequencing. *Applied Soil Ecology, 125*, 288–296. https://doi.org/10.1016/j.apsoil.2018.02.013
42. Wang, Y., Fu, F., Li, J., Wang, G., Wu, M., Zhan, J., Chen, X., & Mao, Z. (2016). Effects of seaweed fertilizer on the growth of Malus hupehensis Rehd. seedlings, soil enzyme activities, and fungal communities under replant condition. *European Journal of Soil Biology, 75*, 1–7. https://doi.org/10.1016/j.ejsobi.2016.04.003