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Biodegradation Kinetics of Organic Matter in Water from Sludge Dewatering after Autothermal Thermophilic Aerobic Digestion

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Abstract: The study presents the research results on the rejected water generated in dewatering sludge stabilised in Autothermal Thermophilic Aerobic Digestion (ATAD) technology. The research was carried out in three municipal wastewater treatment plants (WWTPs), with a capacity of 1500 to 3260 m^3 d^{-1} and a sludge node capacity of 835 to 2000 kg DM d^{-1}. The mean content of Kjeldahl nitrogen (TKN) in the rejected water samples taken from each object ranged from 485 to 1573 mg N L^{-1}, ammonium nitrogen 318 to 736 mg N L^{-1}, and the average concentration of total phosphorus ranged from 96 to 281 mg P L^{-1}. The average content of organic matter expressed as five-day biological oxygen demand (BOD_5) ranged from 205 to 730 mg O_2 L^{-1}, while chemical oxygen demand (COD) ranged from 767 to 4884 mg O_2 L^{-1}. The study determined the kinetics of the biochemical decomposition of organic matter, assuming that it follows the first-order equation. The average reaction rate constant k in subsequent treatment plants was estimated at 0.424, 0.513 and 0.782 d^{-1}. The R^2 coefficient determining the model's adjustment to empirical values was not lower than 0.952. The organic matter biodegradability index average values ranged from 0.17 to 0.26.

Keywords: rejected water; autothermal thermophilic aerobic digestion; ATAD; dewatering; biodegradation kinetics

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Citation: Biedka, P. Biodegradation Kinetics of Organic Matter in Water from Sludge Dewatering after Autothermal Thermophilic Aerobic Digestion. *Energies* **2023**, *16*, 203. https://doi.org/10.3390/en16010203

Academic Editors: Marcin Dębowski, Marcin Zieliński and Joanna Kazimierowicz

Received: 13 November 2022
Revised: 18 December 2022
Accepted: 22 December 2022
Published: 25 December 2022

1. Introduction

Wastewater treatment is always associated with the formation of sludge. Sediments are mainly formed in the process of mechanical and biological treatment. Due to their properties, they may pose a potential threat to the environment in the event of improper management as they may contain, among others, heavy metals and pathogenic organisms. Therefore, sludge treatment and disposal should always be considered as an integral part of treatment of wastewater [1].

The methods of sludge treatment aim to change its properties so that it can be safely disposed of or returned to the environment as fertiliser. It is dictated not only by formal, legal, and economic but also by ecological considerations. Sewage sludge is a rich source of organic matter and biogenic elements, and therefore it can be used for agricultural purposes, fertilisation of soils and plants as a valuable source of nitrogen and phosphorus, compost production, and the reclamation of degraded land. Proper sewage sludge management is essential from the point of view of the circular economy and the depletion of non-renewable resources of minerals from which phosphorus is obtained. Sewage sludge from domestic sewage treatment plants is a rich source of this element. In 2020, a total of 989.5 thousand tonnes of sludge from industrial and municipal wastewater treatment plants were generated in Poland of which 160.4 thousand tonnes were used in agriculture, 26.5 thousand tonnes for land reclamation, including land for agricultural purposes, and 30.5 thousand tonnes for the cultivation of crops intended for the production of compost [2].

One of the methods of changing sewage sludge into biomass that can be used naturally is Autothermal Thermophilic Aerobic Digestion (ATAD). The process ensures complete stabilisation and sanitisation of the sludge and reduces its solid content [3,4]. It is an effective method that runs in a fully automated hermetic installation, does not require the use of enrichment materials (unlike, for example, composting), and requires a small area for the construction of the installation [5].

It may be applied in the wastewater treatment plants with population equivalent (p.e.) up to 30,000 or flow capacity up to 20,000 m^3 d^{-1} [5]. In general, stabilisation is understood here as reduction of the organic matter or volatile solids (VS) concentration, and pasteurization is understood as pathogen elimination via heat treatment [6]. The process is carried out in well insulated reactors into which sludge containing high levels of organic matter and pathogens is loaded. The sludge is mixed and aerated. The composition of the process microflora differs from conventional activated sludge. It consists of 95% Bacillus, Thermus or Acitiomycetes bacteria. Most strains belong to the Bacillus stearothermophilus species, which are active at 40–80 °C [7]. The aerobic digestion process consists of two steps: the direct oxidation of biodegradable matter and endogenous respiration, where cellular material is oxidized [8]. During digestion, thermophiles release metabolic energy, spontaneously raising the temperature of the reactor to the thermophilic range (45–65 °C). These temperatures are lethal to pathogens, resulting in pasteurisation of the sludge [9]. In technological facilities, the process is usually carried out in two reactors working in sequence, with a total sludge retention time of 5–9 days. The end-product of ATAD is considered to be Class A Biosolids that can be applied on agricultural land without restrictions [10]. The process can be carried out on a variety of waste sludge from human, animal, food, or pharmaceutical waste [11] and is also suitable for treating organic liquid waste (1–6% total solid content) [12].

Stabilized sludge, before further development, is subject to a dewatering process. Dewatering facilitates the storage, transport and release of sludge into the environment. Mechanical sludge dewatering produces effluent. This liquid may contain a high concentration of organic substances [13], nitrogen and phosphorus [14]. In most wastewater treatment plants, effluent from sludge dewatering, stabilised with both aerobic and anaerobic methods, is directed to treatment together with raw sewage [15,16]. The increased concentration of nutrients and organic substances in the effluent impacts wastewater treatment technology [17]. While this problem can be solved in new treatment plants at the design stage, modernising the sludge node by introducing the ATAD technology in the existing treatment plant may cause operational problems due to increased nitrogen and phosphorus loads reaching the biological node. Attempts to solve the problem of effluent from sludge stabilisation (both aerobic and anaerobic) by using separate treatment systems are the subject of many studies [17–20]. The main directions of management of effluent from aerobically and anaerobically stabilised sludge include controlled discharge in the mainstream of wastewater, chemical [21] and biological [22] treatment, and bioaugmentation [23]. Due to the high concentration of ammoniacal nitrogen in the effluent, it may be reasonable to use, among others, methods: SHARON (Single Reactor System for High Ammonia Removal Over Nitrite process) [24] and ANAMMOX (Anaerobic Ammonium Oxidation) [25–27]. However, using these methods may be economically unjustified in small-scale treatment plants [13]. As an alternative to biological methods, Huang et al. [28] suggest that chemical precipitation may prove economically viable to remove ammoniacal nitrogen and phosphorus from supernatants after anaerobic digestion.

As described above, water rejected from the dewatered sludge after the ATAD process can cause problems with efficiency wastewater treatment, particularly in facilities where ATAD digestion has been introduced in place of simple sludge hygienisation methods. While the characteristics of rejected water from the anaerobic sludge digestion are widely discussed in the literature, there are fewer studies on rejected waters in the ATAD process.

In this study, the biodegradability (biodegradability index BI—BOD_5/COD ratio) and the kinetics of the biochemical decomposition of the organic matter contained in the effluent from dewatering sludge stabilised in the ATAD process was determined. The research was carried out in three wastewater treatment plants, during operation under typical conditions. Samples were taken from the locations numbered 2-rejected water from dewatering on the process flow diagrams. In addition, the article presents the basic characteristics of these wastewaters, i.e., the concentration of nitrogen and phosphorus and their loads entering the biological node with the raw wastewater.

2. Materials and Methods

2.1. Characteristics of Analysed Wastewater Treatment Plants (WWTPs)

The sewage treatment plants discussed in the study are located in north-eastern Poland, in Pisz, Dąbrowa Białostocka and Wysokie Mazowieckie. These are municipal wastewater treatment plants and are typical facilities in Polish conditions, except for the use of Autothermal Thermophilic Aerobic Digestion (ATAD). Figures 1–3 show simplified process flow diagrams of the considered treatment plants.

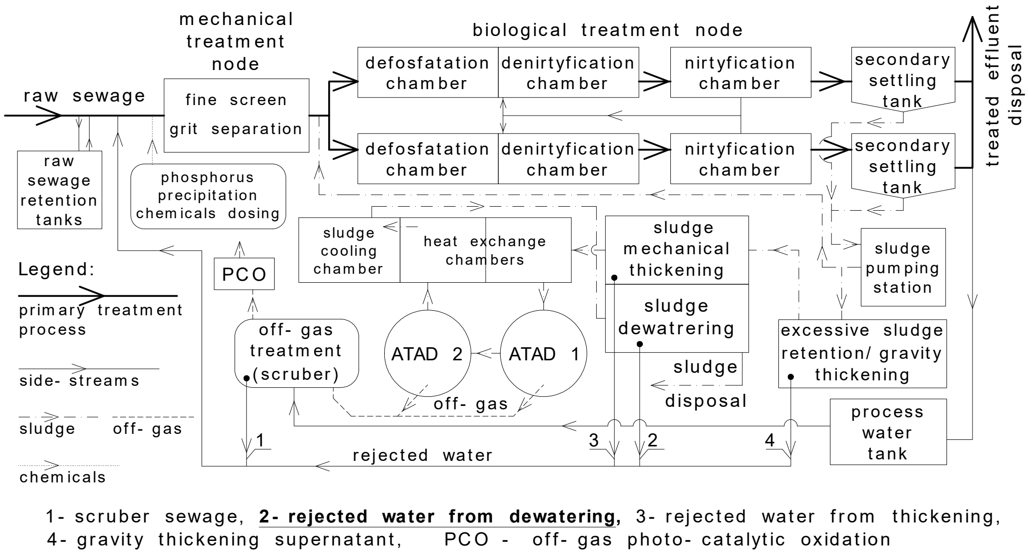

Figure 1. Simplified process flow diagram of WWTP1.

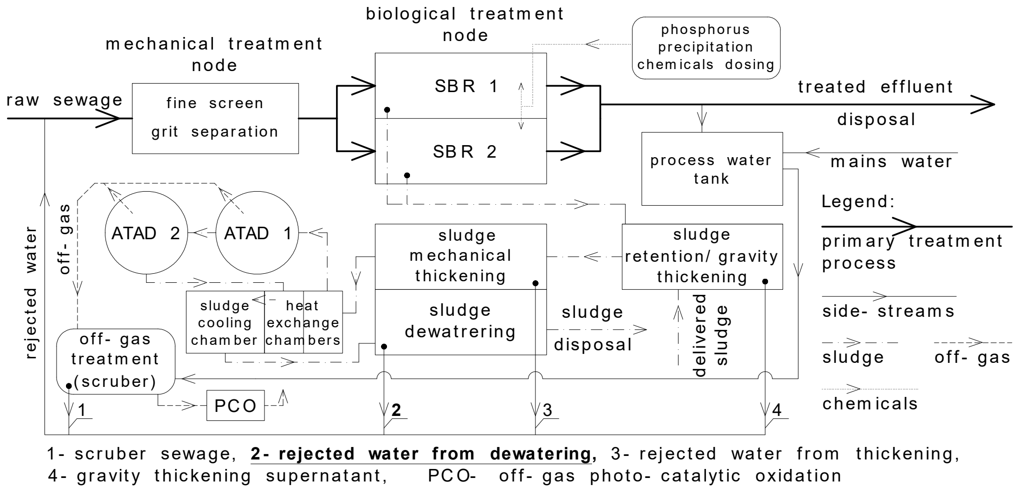

Figure 2. Simplified process flow diagram of WWTP2.

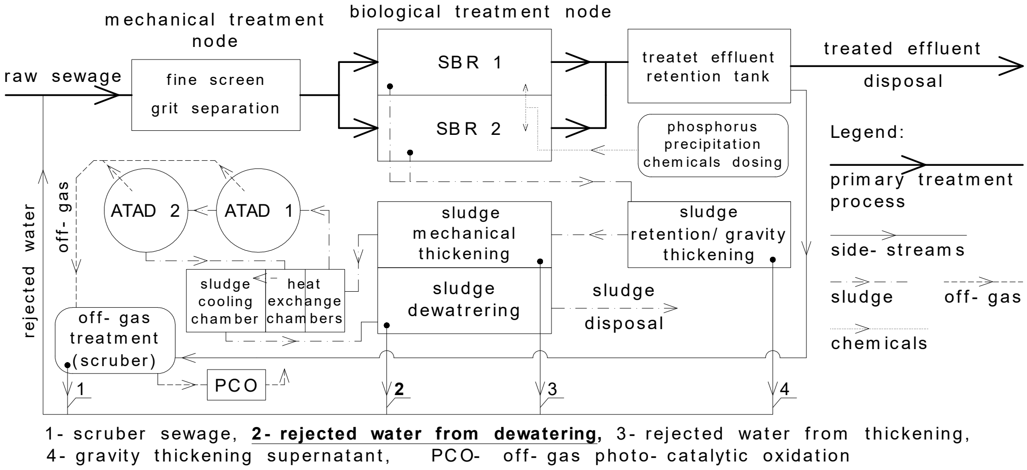

Figure 3. Simplified process flow diagram of WWTP3.

2.1.1. Pisz (WWTP1)

The sewage treatment plant receives municipal wastewater from the city of Pisz (approximately 19,400 inhabitants) and the surrounding sewerage of villages. The designed capacity of the sewage treatment plant is 3500 m^3 d^{-1}, and the capacity of the sludge node—2000 kg DM d^{-1}. A compact device is used in the mechanical wastewater treatment unit: fine screen a grit trap. Biological treatment is carried out using low-loaded activated sludge in A2/O technology (anaerobic chamber—dephosphatation; anoxic chamber—denitrification; aeration chamber—nitrification) with chemical phosphorus precipitation. Excess sludge separated in secondary settling tanks is subjected to gravity and mechanical thickening (up to 5% DM) and then stabilised in the ATAD technology. Stabilisation takes place in two reactors operating in series. The total time to retain the sludge is 6 days. In the second chamber, the minimum temperature is 60 °C. The waste gas generated in the process is cleaned in a water scrubber and then deodorised in an installation that uses photo-catalytic oxidation. The stabilised sludge is dewatered in a belt press. The dry matter content in the dewatered sludge is 14–18% on average. The effluents from the sludge processing are directed together with the raw sewage to the beginning of the technological system.

2.1.2. Dąbrowa Białostocka (WWTP2)

The municipal treatment plant has been operating since 2006, and the designed capacity is 1500 m^3 d^{-1}. The main technological facilities of the wastewater treatment plant include the mechanical and biological wastewater treatment node and the sludge node. In the mechanical treatment node, a compact unit consisting of a fine screen and a grit trap is used. Mechanically treated wastewater is directed to two SBR reactors operating in parallel, with a phase shift. Biological wastewater treatment takes place in reactors using the low-loaded activated sludge method. In 2006, the technology of processing aerobically stabilised activated sludge in SBR reactors was based on mechanical dewatering and lime sanitisation. In 2013, the facility was modernised, and an ATAD installation was built. The first stage of sludge treatment is the mechanical thickening to 4% of dry matter with a centrifuge. Stabilisation is carried out in two chambers ATAD working in serial arrangement. The total time of keeping the sludge in the chambers is 6 days, and the temperature in the second chamber is 55–60 °C. After stabilisation, the sludge is dewatered in a decanter centrifuge to approximately 18–20% dry matter. The rejected water from sludge dewatering is treated together with the raw sewage. In the ATAD process, in addition to the excess sludge, delivered sludge from pre-treatment dairy sewage is also stabilised. The total capacity of the sludge node is 1200 kg DM d^{-1}.

2.1.3. Wysokie Mazowieckie (WWTP3)

The sewage treatment plant in Wysokie Mazowieckie has been in use since 2016. The designed capacity is 1344 m^3 d^{-1}. The technological process includes mechanical wastewater treatment, biological treatment in the SBR technology with chemical phosphorus precipitation and sludge stabilisation in the ATAD technology. For mechanical treatment, a compact device is used—a fine screen and a grit trap. The biological stage of wastewater treatment is carried out in two parallel SBR reactors. The technology of low-loaded activated sludge is used. The excess sludge formed in biological treatment is collected in a gravity thickener, where the hydration is reduced to approximately 98.5–98%. Then, the sludge is mechanically thickened in a decanter centrifuge to a dry matter content of approximately 5%. Due to the low temperatures of non-stabilised sludge occurring during winter, the technological system enables heat recovery from the sludge after stabilising to obtain the proper sanitisation temperature and stable operation of ATAD reactors. Stabilisation takes place in a system of two reactors operating in serial arrangement. The sludge from the first chamber is batch sent to the second chamber after an aliquot of the stabilised sludge has been removed from the second reactor. The process is carried out so that the temperature in the second reactor exceeds 56 °C, and the holding time is 6 days. After cooling, the stabilised sludge is mechanically dewatered on a screw press to a dry matter content of 18–27% (average 22%). The effluents from the stabilised sludge dewatering are directed to the biological treatment system with a raw sewage. The capacity of the sludge node is 835 kg DM d^{-1}.

2.2. Analytical Methods

The study of effluent from mechanical sludge dewatering after the ATAD process was carried out in 2016–2021. In each treatment plant, 8 to 10 samples were collected in that period. The pH and EC were determined in situ using Hach's HQ40D digital multi-parameter meter. The remaining determinations were made in laboratory conditions. Until delivery to the laboratory, the samples were stored at approximately 4 °C. Analytik Jena TOC multi NC2100 analyser was used to measure TOC. Colourimetric methods determined the following: COD—after digestion of 2 h (a $K_2Cr_2O_7$ method, 620 nm); NH_4-N: salicylate method (655 nm); TP: molybdovanadate with acid persulphate digestion method (420 nm). Total phosphorus was tested by mineralising the sample in persulphuric acid, then determined by the molybdovanadate method. Hach DR3900 and DR1900 spectrophotometers were used for colourimetric determinations. TKN determinations were made after the sample mineralisation with sulphuric acid in the presence of a catalyst (K_2SO_4, TiO_2, $CuSO_4$). The mineralised sample was steam distilled to a boric acid solution and then titrated with sulphuric acid, using the potentiometric method, to pH 4.65. The determinations were made of the Buchi KjelDigester K-446 mineraliser, the KjelMaster K-355 distiller and the SI ANALYTICS GmbH TitroLine EASY titrator. Ions were determined using a Thermo Scientific ICS 5000+ ion chromatograph. BOD determinations were carried out using the manometric method. The OxiTop system, WTW, was used with an OC110 controller for custom BOD measurements. Oxygen consumption was measured at 20 °C. To eliminate oxygen consumption in the nitrification process, a nitrification inhibitor was added to the samples: TCMP—2-chloro-6-(trichloromethyl)-pyridine (N-Serve) solution in an amount appropriate to obtain a concentration of 1.5 mg L^{-1} in the sample. Measurements of oxygen consumption were recorded at 100-min intervals for 5 days.

2.3. Calculations

The rate of biochemical decomposition of organic matter was estimated assuming that the mineralisation reaction follows the differential equation of first-order physicochemical reactions:

$$-\frac{d[BOD]}{dt} = k[BOD] \tag{1}$$

where

$[BOD]$—content of an organic matter undergoing biochemical degradation, expressed as biochemical oxygen demand, mg O_2 L^{-1}
t—reaction time, d
k—reaction rate constant, d^{-1}

After integrating Equation (1), we get:

$$\frac{[BOD]_t}{[BOD]_0} = e^{-kt} \tag{2}$$

where

$[BOD]_0$—content of the organic substance undergoing biochemical decomposition in the initial incubation time, total *BOD*,
$[BOD]_t$—content of remaining organic matter undergoing biochemical decomposition over time t.

Under this assumption, the process of oxygen consumption for the biochemical decomposition of organic matter (X) can be written by the equation:

$$X_t = [BOD]_0\left(1 - e^{-kt}\right) \tag{3}$$

where

X_t—oxygen consumption in the process of biochemical decomposition of organic matter over time t, mg O_2 L^{-1}.

The empirical values of oxygen consumption, measured every 100 min for 5 days (x_i), are:

$$x_i = [BOD]_0 - [BOD]_i \tag{4}$$

The reaction rate constants k were determined using a linear estimation, the method of least squares and successive approximations.

The equation of a straight line has the form:

$$y = bx + a \tag{5}$$

where according to the regression line equation:

$$b = \frac{\sum X \cdot Y - N \cdot \overline{X} \cdot \overline{Y}}{\sum X^2 - N \cdot \overline{X}^{\,2}} \tag{6}$$

$$\text{that is : } b = \frac{\overline{X \cdot Y} - \overline{X} \cdot \overline{Y}}{\overline{X^2} - \overline{X}^{\,2}} \tag{7}$$

$$a = \overline{Y} - b \cdot \overline{X} \tag{8}$$

where

Y—dependent variable,
X—predictor,
N—number of observations,
$\overline{value}$—arithmetical mean of the value.

After logarithmisation of the Equation (2) we get:

$$\ln[BOD]_t = \ln\,[BOD]_0 - k\,t \tag{9}$$

Referring to Equations (5) and (9): dependent variable $y = \ln[BOD]_t$, predictor $x = t$ (time), $b = -k$, a = $\ln[BOD]_0$.

From Formula (7) we get:

$$-k = \frac{\overline{t \cdot ln[BOD]_t} - \bar{t} \cdot \overline{ln[BOD]_t}}{\overline{t^2} - (\bar{t})^2} \tag{10}$$

and from Formula (8):

$$\ln[BOD]_0 = \overline{\ln[BOD]_t} + k\,\bar{t} \tag{11}$$

Due to the unknown value of BOD_0, the calculations were made using the method of successive approximations, assuming successive BOD_A values (as expected values of BOD_0) higher than the BOD_5 reading, until the ratio of assumed BOD_A to BOD_0 calculated from Equation (2) equal to 1.00 was obtained. The calculations were made using the Solver tool in the spreadsheet.

3. Results and Discussion

The basic parameters of effluent from the dewatering of sludge stabilised in the ATAD technology obtained as a result of the conducted research are presented in Tables 1–3. In WWTP1 and WWTP2, the sludge is dewatered using decanter centrifuges. WWTP3 uses a screw press. Clearly, higher indicators related to the organic substance, N-NH_4^+ and TKN, were observed in WWTP3, while the lowest concentration of COD, BOD, TP and PO_4^{3-} in WWTP2. The differences may result from discrepancies in the composition of the stabilised sludge (WWTP2 is sludge from dairy wastewater treatment) and sludge dewatering technology (in WWTP3, sludge is dewatered on a screw press to an average dry matter content of 22%). According to the research carried out by Borowski [29], the concentration of ammoniacal nitrogen in the supernatant after the ATAD process ranges from 290 to 715 mg N L^{-1} and total nitrogen from 975 to 1568 mg N L^{-1}. The total phosphorus content ranged from 217 to 327 mg P L^{-1}. A similar range of ammoniacal nitrogen concentration was obtained in the analysed treatment plants. Total phosphorus concentrations were lower, as were total nitrogen concentrations in WWTP1 and WWTP2.

In dairy sewage treatment plants, where the sludge is aerobically stabilised, the rejected water from the sludge dewatering is characterised by a lower content of nitrogen, phosphorus and organic matter and is on average N-NH_4^+ 26.0 mg N L^{-1}, TKN 39.2 mg N L^{-1}, TP 8.6 mg P L^{-1}, COD 235 mg O_2 L^{-1}, BOD_5 132 mg O_2 L^{-1} [10]. The effluents from sludge dewatering after anaerobic digestion (AD) are characterised by the concentration of ammoniacal nitrogen from 400 to 1500 mg N L^{-1} and a low COD value [30].

Table 1. The characteristics of rejected water from ATAD process (n = 8), WWTP 1.

	COD mg O_2 L^{-1}	BOD_5 mg O_2 L^{-1}	TKN mg N L^{-1}	N-NH_4^+ mg N L^{-1}	N-NO_3^- mg N L^{-1}	TP mg P L^{-1}	P-PO_4^{3-} mg P L^{-1}
Mean	1424	251	485	318	1.20	202	168
Minimum	473	135	290	144	0.18	130	74
Maximum	2868	467	1033	811	2.52	420	390
Std. dev.	688	105	242	214	0.79	90	91

Table 2. The characteristics of rejected water from ATAD process (n = 8), WWTP 2.

	COD mg O_2 L^{-1}	BOD_5 mg O_2 L^{-1}	TKN mg N L^{-1}	N-NH_4^+ mg N L^{-1}	N-NO_3^- mg N L^{-1}	TP mg P L^{-1}	P-PO_4^{3-} mg P L^{-1}
Mean	767	205	581	321	1.85	96	72
Minimum	219	35	183	114	0.18	50	19
Maximum	1840	449	1157	648	4.86	146	126
Std. dev.	530	154	324	184	1.83	39	39

Table 3. The characteristics of rejected water from ATAD process (n = 10), WWTP 3.

	COD mg O_2 L^{-1}	BOD_5 mg O_2 L^{-1}	TKN mg N L^{-1}	N-NH_4^+ mg N L^{-1}	N-NO_3^- mg N L^{-1}	TP mg P L^{-1}	P-PO_4^{3-} mg P L^{-1}
Mean	4884	730	1573	736	1.72	281	172
Minimum	2064	288	887	351	0.14	142	49
Maximum	8040	2061	3140	1541	8.65	484	254
Std. dev.	2419	603	667	359	2.65	105	54

When considering nitrogen, phosphorus and organic matter loads returning to the main wastewater stream, the volume of rejected water generated should be determined. In all the described sewage treatment plants, due to the processing of excess sludge, supernatant from gravity thickening, effluent from mechanical thickening, dewatering of stabilised sludge and sewage from the off-gas treatment unit (scrubber) are introduced into the main sewage stream. In the 40 studied sewage treatment plants in Poland, where the anaerobic stabilisation technology is used, the rejected water amount may reach 2.7–7% [31]. In the treatment plants analysed in this study, the share of rejected water volume in the volume of raw sewage was higher. The discussed values are summarised in Table 4.

Table 4. Quantity of the rejected water in main stream of sewage (design values).

	WWTP1	WWTP2	WWTP3
sewage flow, m^3 d^{-1}	3260	1500	1573
sewage sludge mass, kg d.m. d^{-1}	2000	1200 *	835
rejected water, %			
gravity thickening	3.1	4.0	3.1
mechanical thickening	1.8	2.4	1.9
mechanical dewatering	1.0	1.4	1.1
off-gas treatment	2.9	4.8	2.9
TOTAL, %	8.8	12.6	9.0

* biological sludge from pretreatment of wastewater from the dairy industry is delivered to the treatment plant.

Effluents from the dewatering of sludge stabilised in the ATAD technology in WWTP1—WWTP3 treatment plants contribute significant nitrogen and phosphorus loads to the main wastewater stream. The TKN load ranged from approximately 6 to 17%, and TP from approx. 8 to 14%. Higher values were observed in the treatment plant, where a screw press (WWTP3) was used. The source of nutrients in the supernatant liquid is the mineralisation process of the organic substance in the stabilised sludge. Bartkowska [32] states that the loss of dry matter in two-stage ATAD systems due to mineralisation exceeds 38%. The effluents' organic matter content depends on the dewatering unit's efficiency. The values of the BOD_5 load recycled to the main sewage stream accounted for from 0.5 to 1.6% of the load in raw sewage. In the case of a sludge dewatering in decanter centrifuges, COD in the effluent accounted for 1.2 to 1.5% of the raw sewage load, while the effluent from the screw press (WWTP3) contained more, and the effluent load accounted for 7.1% of the COD load flowing into the raw sewage treatment plant. Janus and van der Roest [17] suggest that the nitrogen load from the sludge digestion side stream contains up to 25% of the total nitrogen load while contributing only 2% of the total flow to a treatment plant. This additional stream increases loading to the mainstream process, resulting in larger bioreactors, increased energy expenditure and a potentially decreased effluent quality.

Based on the BOD_5 and COD tests, the biodegradability index (BI) values were determined. The basic statistics of BI are presented in Table 5. The obtained values are lower than the typical values in raw sewage, which averaged 0.47 [33]. BI in raw sewage flowing to the studied sewage treatment plants ranged from 0.47 to 0.75. A more comprehensive range of the index values (0.3–0.96) was obtained by Abdallaa and Hammam when investigating wastewater treatment plants in Egypt [34]. Low index values in effluent from sludge dewatering indicate their low susceptibility to biological treatment, which is justified for BI > 0.5 [35,36]

Table 5. Basic statistic of Biodegradability Index.

	WWTP 1	WWTP 2	WWTP 3
Mean	0.26	0.20	0.17
Minimum	0.08	0.09	0.14
Maximum	0.63	0.36	0.27
Std. dev.	0.21	0.09	0.06

The kinetics of the biochemical decomposition of organic matter was determined, assuming that it follows the first-order equation. A similar assumption was made in early works on the dynamic modelling of wastewater treatment processes [37]. Later models developed included increasing state variables and process descriptions based on widely accepted Monod-type kinetics [38,39]. The reaction rate constants were determined for 8 to 10 measurements of oxygen consumption in samples incubated under laboratory conditions taken from each treatment plant. According to Pahlavanzadeh et al. [40], higher constant values indicate a higher rate of biochemical decomposition of organic matter. Based on the adopted assumptions, it was possible to obtain a high value of R^2, a measure of the model fit to the empirical values obtained as a result of the research. The value of R^2 ranged from 0.952 to 0.995 in WWTP1, 0.953 to 0.981 in WWTP2 and 0.952 to 0.990 in WWTP3. Figures 4–9 show the empirical and calculated oxygen consumption values for the biochemical mineralisation of organic matter. Selected effluent samples, i.e., with the highest and the lowest R^2 values, in each treatment plant were presented.

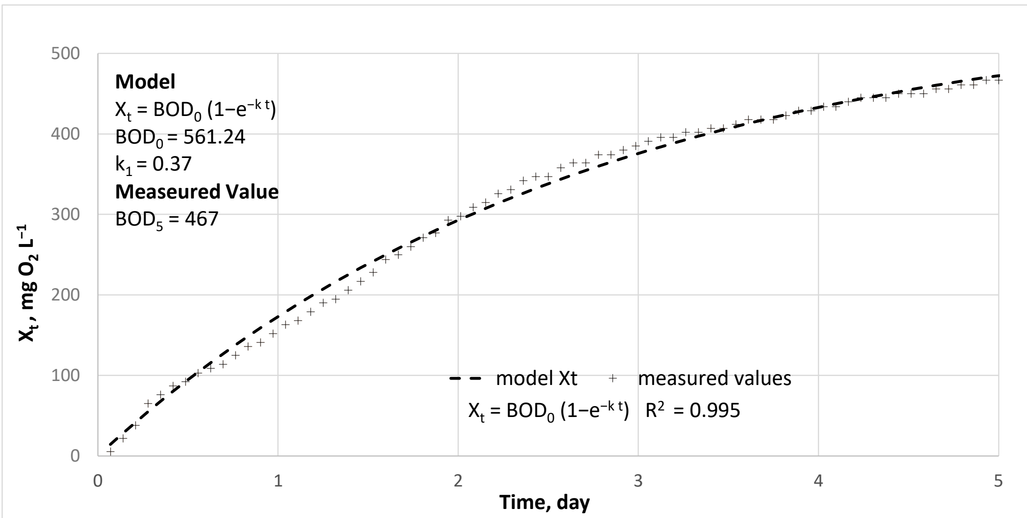

Figure 4. Value of biochemical oxygen consumption in rejected water, WWTP 1, sample 1.

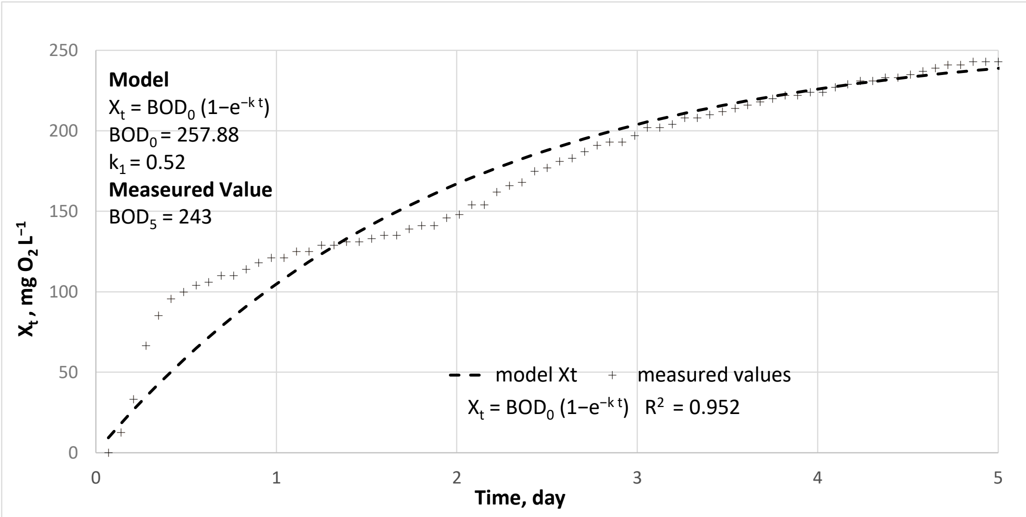

Figure 5. Value of biochemical oxygen consumption in rejected water, WWTP 1, sample 2.

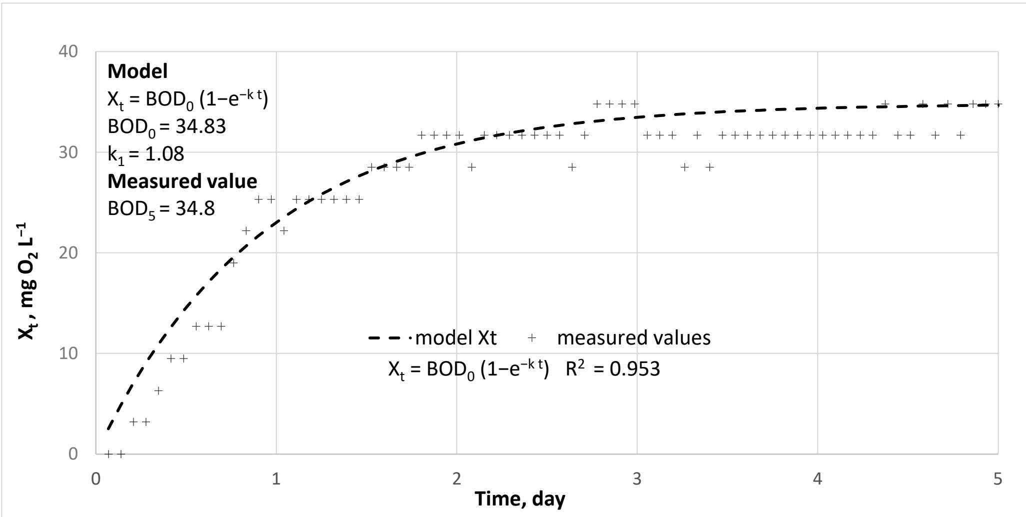

Figure 6. Value of biochemical oxygen consumption in rejected water, WWTP 2, sample 1.

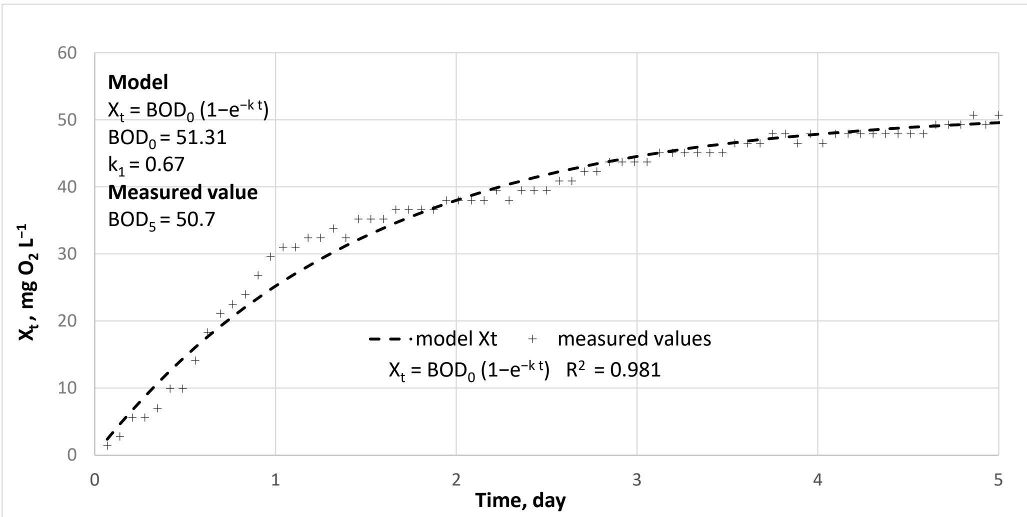

Figure 7. Value of biochemical oxygen consumption in rejected water, WWTP 2, sample 2.

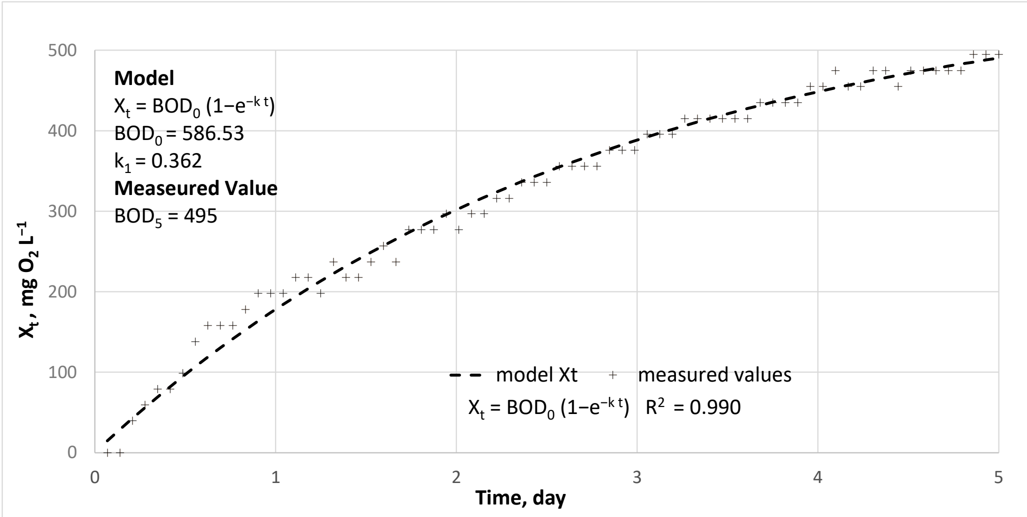

Figure 8. Value of biochemical oxygen consumption in rejected water, WWTP 3, sample 1.

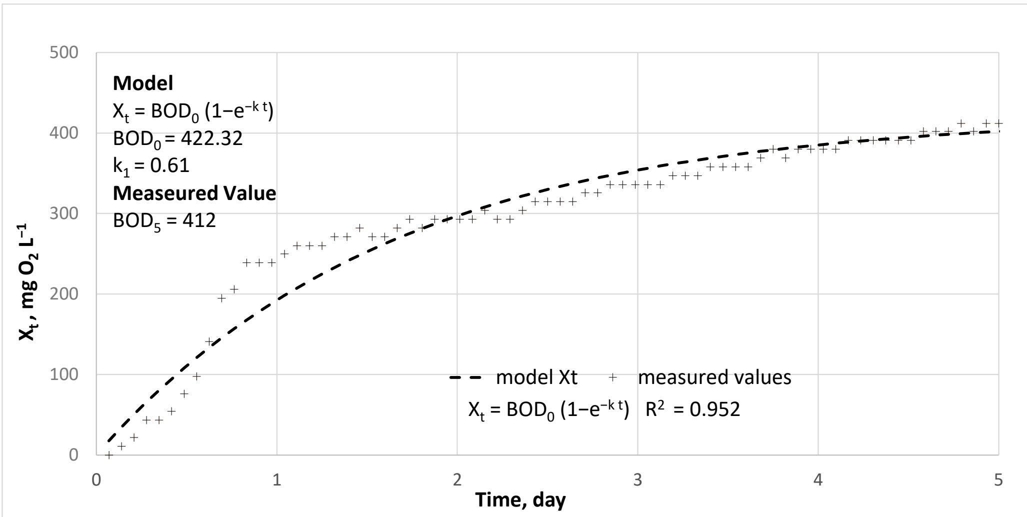

Figure 9. Value of biochemical oxygen consumption in rejected water, WWTP 3, sample 2.

The mean values and the error and standard deviation of the reaction rate constants for all tested samples are presented in Figure 10.

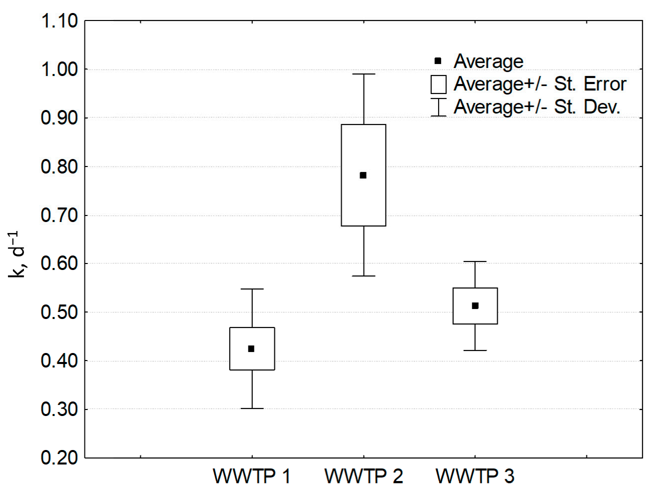

Figure 10. Differentiation of biodegradation rates *k* in analysed samples.

The mean values of the reaction rate constant in WWTP1 and WWTP3 were similar and amounted to 0.424 and 0.513 d^{-1}, respectively. Higher values were observed in WWTP2, and the mean was 0.782 d^{-1}. The biologically degradable substance in WWTP2 effluent was mineralised faster than WWTP1 and WWTP3. It is probably due to different properties of the stabilised sludge—in WWTP2, apart from excess sludge, sludge brought from dairy sewage pre-treatment is stabilised. The values of the constant k of the first-order reaction in domestic wastewater range from 0.1 to 0.6 d^{-1} [41]. According to Sun and Saeed [42], the k of the kinetics of the reaction of reducing the BOD_5 value in the wastewater treated on vertical reed beds was 0.0964 d^{-1}.

Knowledge of the characteristics of rejected water should form the basis for assessing whether the introduction of ATAD technology into existing treatment plants will result in the malfunction of the biological reactors, as well as the possible design of economically viable separate treatment methods. This is particularly important for facilities below 30,000 p.e. using SBR technology.

4. Conclusions

The volume of the liquid generated in the dehydration process did not exceed 1.4% of the average daily sewage inflow. The TKN load from the leachate returned to the biological process node concerning the load in raw sewage in the studied objects ranged from approximately 6 to 17%, and in the case of total phosphorus: 8% to 14%. The organic matter in the sludge-dehydrating effluent was not susceptible to further biodegradation. The BI factor (BOD_5/COD) ranged from 0.17 to 0.26 on average. The rate of biochemical decomposition of organic matter (expressed as BOD_5) contained in the leachate can be described by the first-order kinetic equation. The oxygen consumption rate for the mineralisation of biodegradable organic matter does not differ from the values observed in typical domestic wastewater. The reaction rate constant k ranged from 0.214 to 1.081, and the mean values for the tested objects were estimated at 0.424 (WWTP1), 0.782 (WWTP2) and 0.513 (WWTP3).

Funding: The research was carried out under project number WZ/WB-IIŚ/03/2022 at the Bialystok University of Technology. It was funded by a research subvention provided by the Ministry of Education and Science.

Institutional Review Board Statement: Not applicable.

Informed Consent Statement: Not applicable.

Data Availability Statement: Not applicable.

Conflicts of Interest: The author declares no conflict of interest. The funders had no role in the design of the study; in the collection, analyses, or interpretation of data; in the writing of the manuscript; or in the decision to publish the results.

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