POLITECHNIKA LUBELSKA

Water Supply and Wastewater Disposal

edited by
Henryk Sobczuk
Beata Kowalska

Lublin 2018

MONOGRAFIE

Water Supply and Wastewater Disposal

Monografie – Politechnika Lubelska

Politechnika Lubelska
Wydział Inżynierii Środowiska
ul. Nadbystrzycka 40B
20-618 Lublin

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Publication approved by the Rector of Lublin University of Technology

ISBN: 978-83-7947-314-4

Publisher: Lublin University of Technology
ul. Nadbystrzycka 38D, 20-618 Lublin, Poland
Realization: Lublin University of Technology Library
ul. Nadbystrzycka 36A, 20-618 Lublin, Poland
tel. (81) 538-46-59, email: wydawca@pollub.pl
www.biblioteka.pollub.pl
Printed by : TOP Agencja Reklamowa Agnieszka Łuczak
www.agencjatop.pl

The digital version is available at the Digital Library of Lublin University of Technology: www.bc.pollub.pl
Circulation: 100 copies

Contents

Phosphogypsum utilization in the dephosphotation treatment of wastewater and sewage sludge

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Abstract

The department of applied ecology at the Sumy State University is developing phosphorus recovery processes from organic and mineral secondary sources like wastewater, sewage sludge and phosphogypsum under sulphatereduction condition to produce environment safe products. The latest results of the development are shown and discussed in this paper. The possibility of the phosphogyFpsum utilization as a mineral suppler for bacteria growth in the process of anaerobic stabilization of sewage sludge for the extraction of nutrients (phosphates) was studied. Sewage sludge were placed in bioreactors with addition of phosphogypsum ($CaSO_4 \cdot 2H_2O$). Phosphate concentration in anaerobic bioreactors varied in a relatively wide range from 32 mg PO_4^{3-}/dm^3 to 67 mg PO_4^{3-}/dm^3. The concentration of phosphate ions in the outflow was observed in the range from 112 mg PO_4^{3-}/dm^3 to 179 mg PO_4^{3-}/dm^3. Accordingly, the degree of phosphate ions increase occurred from 62% to 80% with the stimulating action of the phosphogypsum addition. Together with the release of phosphates, the removal of the organic substrate was measured. In this case, the COD outflow of the system was in the range of values 110-181 mg O_2/dm^3. The rational dose of phosphogypsum was determined. The anaerobic process of phosphate release was enhanced by phosphogypsum introducing into the system and, accordingly, stimulation of the sulfate-reduction bacteria growth with metabolites generation and destruction of metal-phosphate complexes and indirectly stimulation of phosphate mobilizing bacteria activity. Thus, the release of phosphate into solution was intensified.

***Keywords**:* phosphogypsum utilization, phosphorus recovery, sewage sludge, wastewater

1. *Introduction*

Wastewater is a major P sink and, during wastewater treatment, P is removed with the sewage sludge. The direct recycling of P through agricultural application of sewage sludge is becoming increasingly difficult and a growing amount of sewage sludge is incinerated and the contained P is lost as the ashes are disposed. Consequently, the recycling of P from wastewater and sewage sludge can contribute to conserve global P reserves (Cornel et al., 2013).

The phosphorus cycle is crucial because phosphorus is usually the limiting nutrient in ecosystem. Fig. 1 illustrates schematic representation of the phosphorus cycle presents sources, consumption, collection, treatment and end of life with natural process of time consuming recycling for phosphorus.

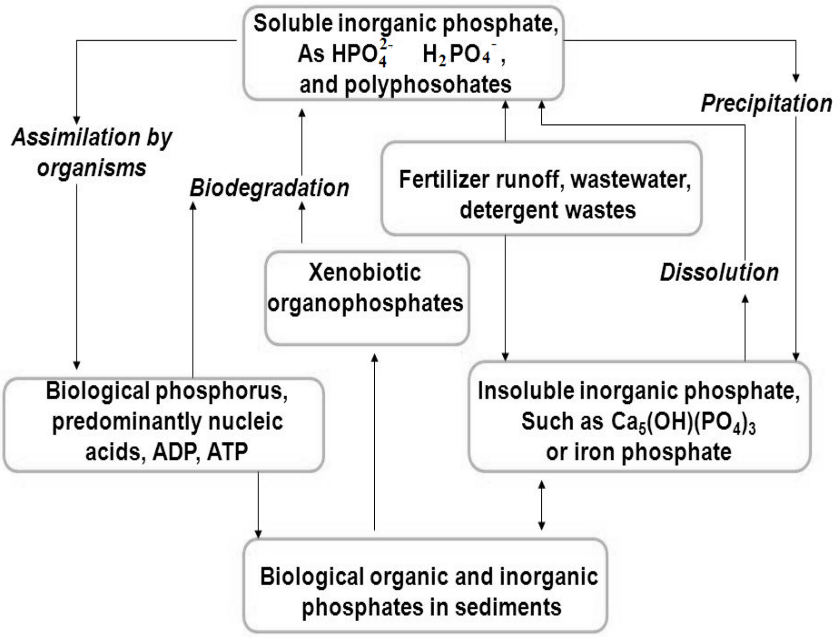

Fig. 1. The phosphorus cycle (http://slideplayer.com/slide/3514207/)

The municipal wastewater characteristics are very site-specific, and depends on various factors such as, sewer infiltration, discharges of industrial wastewater, use of phosphate detergents, type of sewer system (combined or separated) (Petzet et al., 2013). It has been found that municipal wastewaters typically have a variation between 4 to 11 mg/dm^3 of phosphorus as P, where various places have an average between 6 and 8 mg/dm^3 (Cornel et al., 2009, Tchobanoglous et al., 2014).

In Ukraine, it was found that approx. 8-10% of the phosphorus could be removed using primary settling with mechanical removal and at biological treatment by

35-50%. (Dolina, 2011). When using an activated sludge system about 15% to 25% of phosphorus can be removed as the bacteria use phosphorus for growth. The phosphorus is incorporated in the sludge mass and removed through the removal of the surplus sludge (Henze et al., 2008). However, effluent requirements are often below 1.0 mg/dm^3 or more than 90% removal, so the rest of the phosphorus has to be removed with an addition of other processes. As the second step in phosphorus recovery is to release phosphorus from the sludge bound compounds that may be done by different methods for example through the use of physical, mechanical, biological or chemical means. During this step phosphorus is transferred into a relatively small stream compared with the influent flow. The phosphate concentration in this stream may be 10~50 times higher than the influent phosphate concentration (Mrowiec B., 2003, Stark, 2006).

The results obtained by Ivar Urdalen (2013) estimated that enhanced biological phosphorus removal (EBPR) is a much more favorable technology than chemical phosphorus removal. The phosphorus is taken up in the activated sludge and it is relatively easy to recover the phosphorus compared to chemical sludge. Due to the lack of chemicals used in this process, EBPR systems have lower sludge production, lower chemical costs and the sludge contains less metals. It has a higher value for agriculture as the phosphorus is not bound to metals, and is therefore more available to the plants. However, the EBPR process is very dependent on the wastewater characteristics and is less stable and flexible compared to chemical precipitation (Janssen et al., 2002, Tchobanoglous et al., 2014). It normally involves more complex control and larger reactor volumes as well. However, the results obtained by Nan Shen et al. (2016) suggest the performance of EBPR can be affected by available carbon sources types in the wastewater that may induce different functional microbial communities in the process that was.

Phosphorus can be recovered as different products from the wastewater treatment plants. The desorption solution can be enriched with phosphate ions through repetitive application of the same solution and used as a potential P-rich resource or alternatively serve as a source for further precipitation of a solid P-product.

Use of recycled phosphorus reduces the dependency on raw phosphate rocks, ensures food security, decreasing the adverse effect on health and environment because of eutrophication (Chapagain, 2016).

The possibility of co-processing of wastewater, sewage sludge and phosphogypsum in the anaerobic condition with the heavy metals precipitation with biogenic hydrogen sulfide was substantiated in the department of applied ecology, Sumy State University (Plyatsuk and Chernysh, 2014–2016). And current investigation is conducted with process of phosphorus removal from sewage sludge and phosphogypsum.

Thus this paper focuses on the study the dephosphotation treatment of sewage sludge together with phosphogypsum under biosulfidogenic conditions. Thus, objectives of this work next:

- determine the relationship between phosphates release and the concentration of phosphogypsum decrease under sulfate reduction conditions;
- study the synergistic effect of the combined the COD loading and the phosphogypsum dosage on the metabolic activity of anaerobic microorganism groups in the process of phosphate release.

2. *Materials and Methods*

Raw sewage sludge used in this work was collected from WWTP corresponding to a population of approximately 290.000 inhabitants located in city Sumy (Ukraine). A mixture (50:50, v/v) of sewage sludge from sludge beds and excess activated sludge collected from the thickener and flotator, respectively, was used as feeding of the anaerobic pilot plant.

Phosphogypsum is formed in an amount of about 100 tons annually in Sumy region (Ukraine). Currently, over 14 million tons of phosphogypsum are accumulated. Phosphogypsum is a multi tonnage waste of extraction phosphoric acid production. Solid waste generated in the process of sulfuric acid decomposition of natural phosphate raw material and the solid phase (calcium sulfate) separation from phosphoric acid solutions. The reaction proceeds as follows:

$$Ca_5(PO_4)_3F + 5H_2SO_4 \rightarrow 5CaSO_4 + 3H_3PO_4 + HF \quad (1)$$

The precipitate consists mainly of calcium sulfate dehydrate ($CaSO_4 \times 2H_2O$) and contains impurities of phosphate, which is not decomposed, phosphates and silicates. The quantitative content of impurities depends on the mineral composition of the feedstock, smooth flow of production, serviceability of equipment and process discipline, etc. (Tab. 1). The solubility of phosphogypsum is 2100 mg in 1 liter water (35 ^{o}C).

Table 1. Composition of phosphogypsum from dump of PJSC "Sumykhimprom" (Sumy region, Ukraine) in terms of oxides, %

CaO	**SO_3**	**P_2O_5**	**SiO_2**	**Fe_2O_3**
36.0–34.0	39.0–38.0	3.0–2.7	1.2–2.3	2.2–3.4
MgO	**ZnO**	**CuO**	**F**	**H_2O crystl.**
0.002–0.004	0.03–0.06	0.002–0.0035	0.016–0.002	19.64–20.74

Using phosphogypsum as a mineral substrate for sulfatereduction bacteria growth has the following advantages: low-cost raw material base; enrichment biogenic elements (calcium, phosphorus, etc.) of sewage sludge; sulfur compounds contained in the phosphogypsum can be freely used by bacteria as a mineral substrate for their growth, which is due to the high sulfate/sulfite ions affinity of microbial cells; reducing of chemical waste dump on the environment.

2.1. Anaerobic microbiological degradation pilot plant

Experimental setup consists of anaerobic bioreactor, where the process proceeds directly to sulfate reduction. The covered cylindrically shaped anaerobic bioreactor with a zinc acetate trap for sulfide was used. The digested sludge outlet on the bottom of the reactor and it was used for sampling purposes. The bioreactor had a total volume of 6 dm^3. Initial pH-level of system was from 6.5 to 7.5. The anaerobic bioreactor was equipped with a temperature regulation device and was kept in a constant temperature at 35^oC in the incubator. The incubator was made own special for experiment.

In order to maintain anaerobic conditions in the flask, aluminum foil was used to exclude light and to prevent the growth of photosynthetic bacteria. Syringes were used for sampling purposes and a sealed nitrogen injection port was created. The feeding for reactor is stored at 4 ^oC, from where it is pumped to reactor. The stable mixture is put into bioreactor.

Monitor parameters: moisture content of the substrate; COD; pH level; concentration of phosphogypsum in the anaerobic system; concentrations of sulfur and phosphorus compounds in the liquid and solid phases.

2.2. Methods of investigation

Elemental analysis of the samples (liquid and solid phases) was carried out on the X-ray fluorescence analyzer Elvax. Limits of detection of impurities are not less than 10 ppm.

X-ray diffraction studies of the material structure were made on automated DRON-4-0. Automation System DRON-4-07 is based on microprocessor controller, which provides control of the goniometer GUR-9 and data transmission in digital form on a PC. The experimental results are transmitted directly to the support package experiment DifWin-1 for pre-processing. Identification of the crystalline phases was carried out by filing JCPDS (Joint Committee on Powder Diffraction Standards).

pH was analyzed by pX-meter pX-150 (ionometer).

A Merck® spectroquant test kit was used to determine the total and soluble COD of the reactor. Prior to COD determination, all samples were acidified with concentrated HCl to pH = 2 in order to remove any dissolved sulphide.

The anaerobic bacteria' colonies and extracellular structures were electron-microscopic studied. Micrographs of microbial preparations were prepared and processed using digital image output system «SEO Scan ICX 285 AK-F IEE-1394» and morphometrical program «SEO Image Lab 2.0». Additional research produced by scanning electron microscope and X-ray REMMA102 DRON-4-07. Electron microscopic investigation was performed after freezing samples of culture media in liquid nitrogen and freeze-dried in a vacuum.

3. *Results and Discussion*

The dose of phosphogypsum introduced into an anaerobic bioreactor, as a source of sulfate and other macro- and microelements (Ca, P, K etc.) for the stimulation of sulfatereduction bacteria growth, and affected the amount of phosphate removal. The decrease of phosphogypsum concentration had a close correlation with the release of phosphate ions (Fig. 2).

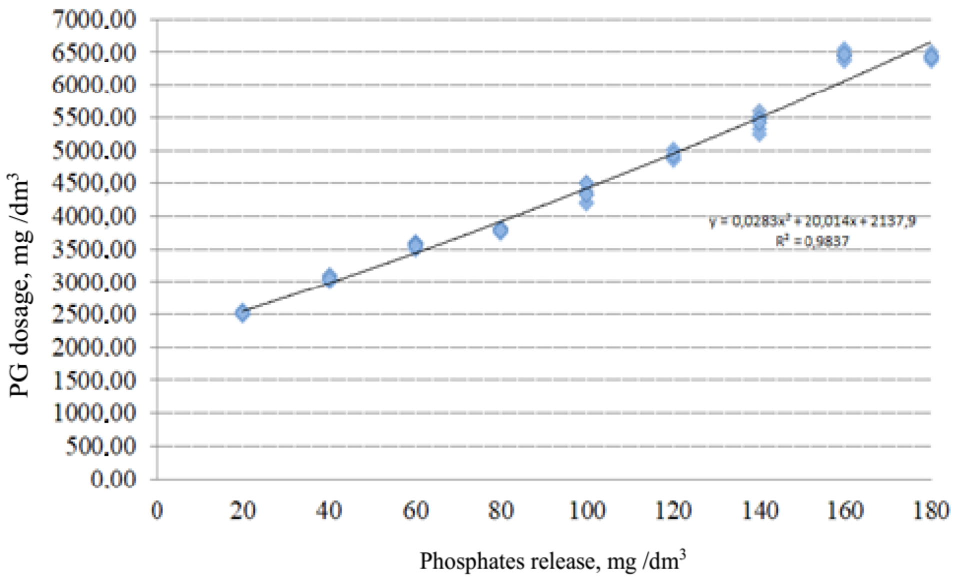

Fig. 2. The relationship between phosphates release and the concentration of phosphogypsum decrease under sulfate reduction conditions

It should be noted that this process conducted with biochemical conversion of phosphogypsum under anaerobic conditions. The obtained dependence of the process of phosphates release (M (Y)) on the concentration of phosphogypsum (X_1) was approximated by the regression equation:

$$M(Y) = 0.0283X_1^2 + 20.014X_1 + 2137.9 \; (R^2 = 0.9837) \qquad (2)$$

According to our research (Fig. 2) increasing phosphogypsum concentration up to 7000 mg/dm^3 wasn't impact on phosphate increase in solution was observed. Thus, a rational dose of phosphogypsum was 6500 mg/dm^3.

Phosphate concentration in anaerobic bioreactors varied in a relatively wide range from 32 mg PO_4^{3-}/dm^3 to 67 mg PO_4^{3-}/dm^3. The concentration of phosphate ions in the outflow was observed in the range from 112 mg PO_4^{3-}/dm^3 to 179 mg PO_4^{3-}/dm^3 (Fig. 3). This means that as a result of the microorganisms activity, an increase in the phosphates concentration in the soluble 2.5–5.0 times was observed at the stimulating action of the phosphogypsum additive.

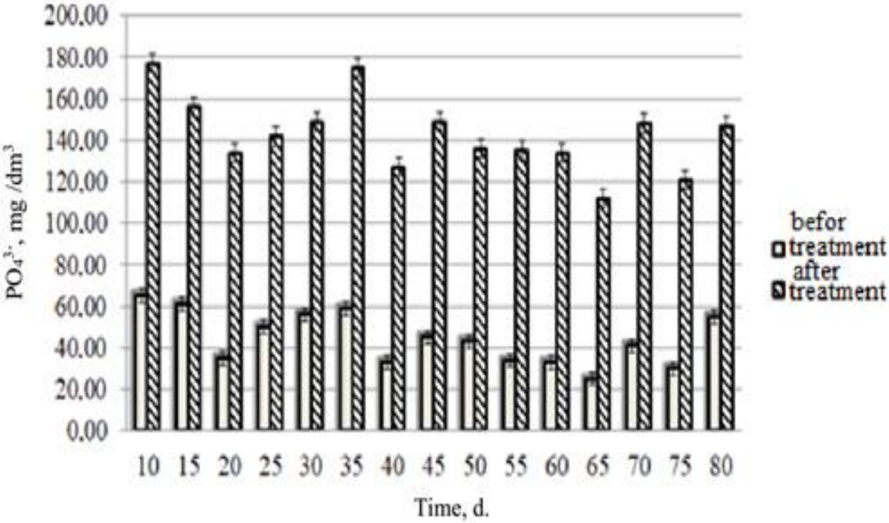

Fig. 3. Concentration of phosphates before and after treatment that released into the solution under sulfate reduction conditions with phosphogypsum processed

Accordingly, the degree of phosphate ions increase (DI_P) in solution was calculated by the formula:

$$DI_P = \frac{(C_1 - C_0)}{C_1} \cdot 100\%, \quad (3)$$

where: the concentration of phosphates after (C_1) and before treatment (C_0) in the liquid phase.

In Fig. 4 was shown that the degree of phosphate ions increase changed from 62% to 80%.

Together with the release of phosphates, the removal of the organic substrate was measured, which was expressed in changes of the COD. The initial COD of water extraction (1: 1) of sewage sludge treated in an anaerobic reactor was changed from 300 mg O_2/dm^3 to 396 mg O_2/dm^3. The final COD of a similar flow filtrate was much lower and was changed of values 110–181 mg O_2/dm^3.

The maximum COD (181 mg O_2/dm^3) was measured in the sludge at the initial stage of the dephosphotation. When the induction stage was passing, after 5 days in the stationary mode of operation (10–15 days) all measurements showed a value of COD below 145 mg O_2/dm^3.

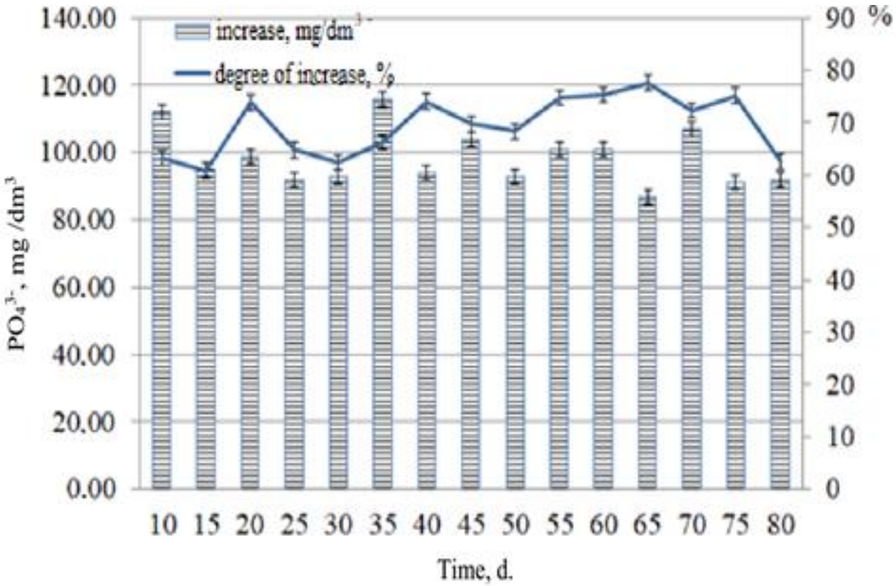

Fig. 4. Increase in the concentration of phosphate in the liquid phase after the treatment under sulfate reduction conditions with phosphogypsum processed

The presence of a source of carbon and energy affects the biochemical process of separating phosphates from wastewater and sewage sludge. Such as in anaerobic conditions, phosphate-membrane microorganisms convert organic compounds and convert slow-acting phosphorus compounds into a soluble form (*Bacillus* (*B. megaterium*), *Penicillium*, *Aspergillus*, etc.).

The process of anaerobic phosphate release was enhanced by introducing phosphogypsum into the system and, accordingly, by stimulating the development of sulfatereduction bacteria that secrete products of their own metabolism in the medium and indirectly stimulate the activity of phosphate-mobilizing microorganisms and intensify the allocation of phosphate ions in solution.

In order to understand the synergistic effect of the combined the COD loading and the phosphogypsum dosage on the metabolic activity of anaerobic microorganism groups in the process of phosphate release, the effect of the ratio of COD and phosphogypsum dosage (COD/PG) was investigated (Fig. 5).

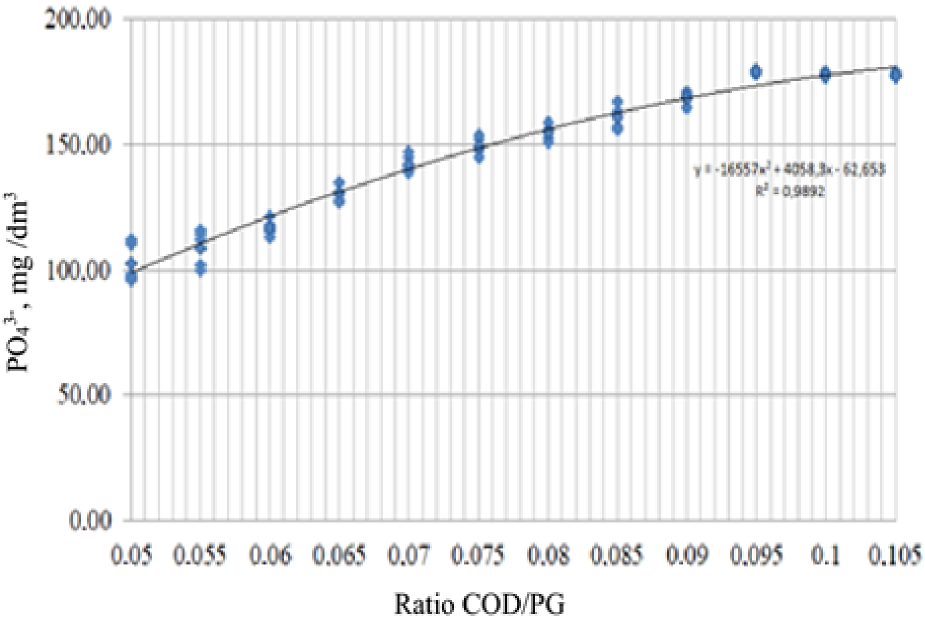

Fig. 5. Influence of the ratio of COD/PG on the process of phosphates release

Thus, according to the obtained results (Fig. 4), the effect on phosphates release (M (Y)) from sewage sludge was correlated with the ratio of COD/PG (X_2) was determined and regression equation was formed:

$$M(Y) = -16557X_2^2 + 4058.3X_2 - 62.653 \; (R^2 = 0.9892) \quad (4)$$

Along with an increase COD/PG to 0.1, an increase in the amount of phosphate in solution was observed. With a further increase in the ratio of COD/PG, increase of phosphate ions wasn't observed.

In general, the regularities of phosphorus biochemical transformations in the process of anaerobic conversion of sewage sludge and phosphogypsum were reduced to the following provisions:

1 iron (III), which binds to P, appears in the lack of oxygen access (in an anaerobic bioreactor), and Fe (III) is reduced by the organic matter that presented in sewage sludge during the microbial reduction. Since the Fe (II) salts, including $Fe_3(PO_4)_2$, are soluble, Fe^{2+} and P ions begin to diffuse;

2 the process of fermentation of organic substances, decrease COD and the separation of polyphosphates $(HPO_3)_n$ from the silt cells into the liquid phase occurs is observed under sulfate reduction conditions;

3 metal ions, in particular Fe^{2+}, are reacted with dissolved hydrogen sulfide to form persulfides and complex sulfides. In the process of isomorphism, a complex sulfide fraction is formed that binds heavy metals;

4 phosphate ions, in turn, react with calcium ions to form low soluble compounds (not more than 10% of the mass concentration of phosphate ions). At the same time, much of the phosphate ions pass into the liquid phase.

4. Summary and Conclusions

The effect of phosphogypsum dosage on the process of dephosphotation under sulfate reduction conditions was carried out. Thus, the final concentration of phosphates in the liquid phase of sewage sludge was observed in the 2.5–5.0 times higher compared with the incoming flow. Together with the release of phosphates, the removal of the organic substrate was measured. In this case, the COD outflow of the system was in the range of values 110–181 mg O_2/dm^3. The rational dose of phosphogypsum was determined.

The regression equations of the dependences of the phosphate release on the concentration of phosphogypsum (X_1) and the ratio of COD/PG (X_2) were obtained. It has been determined that the process of anaerobic phosphate release is enhanced by phosphogypsum introduce for stimulation of the bacteria metabolism. In this case, phosphogypsum disposal is carried out, which contributes to the reduction of the load on the environment from the storage sites of this multi-tonnage chemical industry waste.

The implementation of further research will focus on the identification of environmentally safe areas of product use from anaerobic conversion of sludge and phosphogypsum.

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Issues related to the realization of water supply system investments in rural areas in Poland

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Abstract

The development of municipal infrastructure responsible for water supply and sewage disposal is significant for the rational management of natural resources (water), as well as for improving the standard of living of residents, and it also contributes to the improvement of the natural environment.

Designing the water supply system requires providing designers with a lot of information, most of which can be found in local spatial development plans approved for particular municipalities or their parts. In Poland, especially in rural areas, these plans are often incomplete or they do not exist at all. This often results in the wrong selection of the designed water pipes diameter and the need to rebuild relatively recently built sections. Another negative factor is also functioning of the criterion of the lowest price when selecting service providers, imposed by the Public Procurement Act. Following this criterion results, in the majority of cases, in obtaining a poor quality project as well as performing an investment project at a low technical level.

The article presents the problems of water supply and sewage management in settlement units in rural areas in Poland, the characteristics of these areas have been described, and the possible solutions for the water supply system in rural areas have been discussed.

The article shows the most common exploitative problems resulting from wrong assumptions and an incorrect design of the water supply system, its components, as well as problems connected with an improper selection of equipment.

The article discusses the aspects related to the implementation of water supply investments, the examples of realization of such investments, as well as the possible directions of development of water supply systems in settlement units in rural areas.

Keywords

water supply system, water and wastewater management in rural areas, water and sewage infrastructure, subsidies from the UE

1. Introduction

The dynamic development of water supply and sewage infrastructure in rural areas in Poland is largely due to the requirements set by directives and subsidies from the European Union.

According to the legal regulations in Poland, supplying a settlement unit in rural areas with a water supply system and collective sewage disposal is the duty of municipal authorities (Dz.U. 2001, 2005, 2009).

The largest investment outlays (about PLN 500 million) intended for equipping the water supply infrastructure of rural areas in Poland, in 2003–2011 were incurred by Mazowieckie, Kujawsko-Pomorskie, Wielkopolskie, Zachodniopomorskie and Dolnośląskie Voivodeships. These investments could, to a large extent, be implemented thanks to the financial support of the European Union (Krawczyk, 2014).

However, there are still difficulties both in terms of obtaining EU subsidies for realising the investments, and in terms of solutions for water supply systems of rural settlement units, resulting from the characteristics of these areas. Their vastness and scattered housing, as well as low population density, make it difficult to plan investments and equip the areas with communal infrastructure. The question of the quality of the water supplied is not without significance, because the areas in question are closely connected with the production of food - the cultivation of plants and breeding animals.

The article presents possible tendencies of equipping rural areas with water supply infrastructure and identifies the problems encountered by municipalities at every stage of investment implementation.

2. Characteristics of rural areas in terms of water supply infrastructure

In recent years, the water supply network has been the fastest growing element of economic infrastructure in rural areas. In Poland, in the last decade, the length of the water supply network in rural areas has increased from 190.7 thousand. km in 2005 to 231.0 thousand km of network in 2015, i.e. by 21.1%. The number of connections in rural areas increased by almost 633 thousand. i.e. by 23% (Figure 1).

The most significant increase in the length of the water supply network was observed in rural areas of the following voivodeships: Zachodniopomorskie – by over 39%, Warmńsko-Mazurskie - by almost 34% and Mazowieckie - by almost 33%.

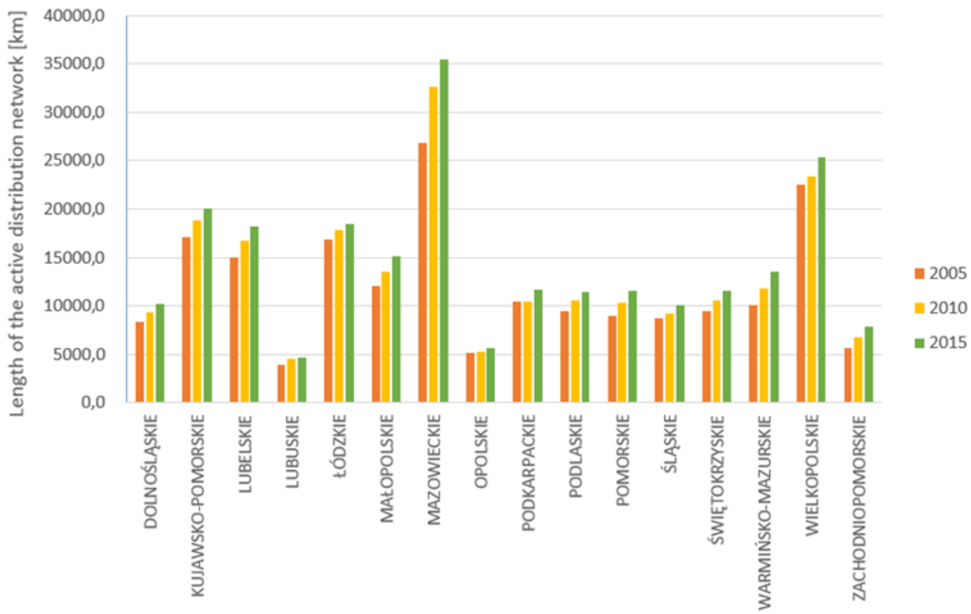

Fig. 1. Condition of water supply infrastructure in 2005–2015 (source: own study based on GUS data)

In 2015, the length of the water supply network in Poland reached almost 298 thousand. km, and the number of connections reached almost 5.5 million. Over 77.5% of the length of the water supply network and approximately 61.8% of the connections to the buildings were located in

rural areas. Comparing to the previous year, the length of the water supply network increased in rural areas by more than 4.1 thousand. km, and the number of connections increased by 69.2 thousand. (Fig. 2) (GUS, 2005–2015).

On the basis of the mentioned indicators in the analyzed time, it can be concluded that a sustained increase in the development of water supply systems in the entire country was documented. As for the analysis of the increase in the length of the water supply networks under construction in particular voivodeships, it turns out that there is a large diversity of the analyzed indicators (Figure 1). There are regions of the country where the development of the water supply is very dynamic (for example, Mazowieckie, Wielkopolskie, Kujawsko-Pomorskie Voivodeships) and there are such regions where much less dynamic growth has been recorded.

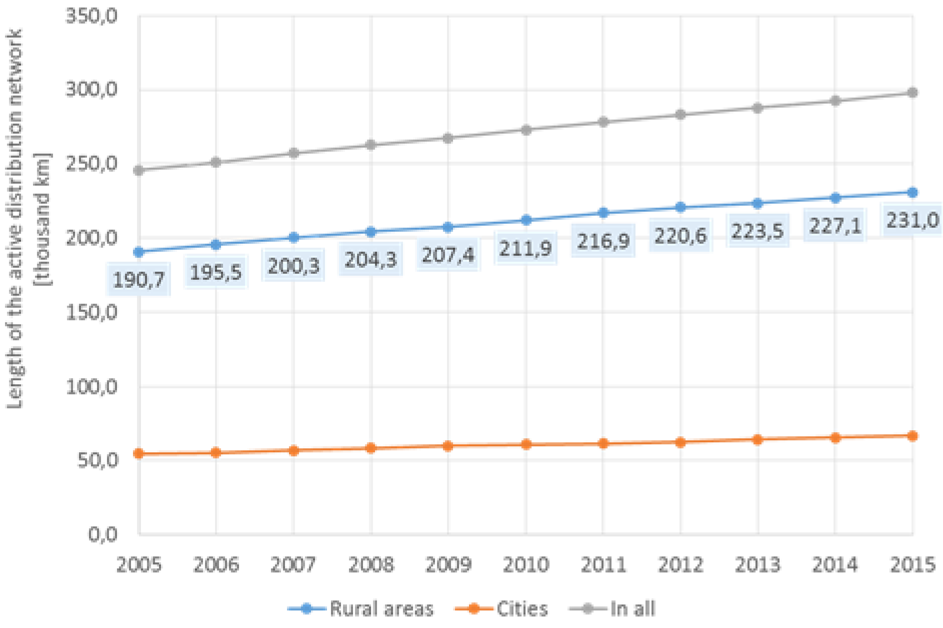

Fig. 2. Length of the active water distribution network in rural areas in Poland (source: own study based on GUS data)

It is difficult to indicate one reason for this differentiation. The picture presented in the graphs consists of many factors, the most important of which are characterized in the next part of the article.

3. *Problems related to the investment of water supply infrastructure*

3.1. Problems at the stage of applying for investment financing

A crucial problem at the initial stage of investment implementation is a total lack of preparation of the municipality authorities for submitting the application for subsidy.

In order to obtain funds as part of European Union support, the municipal authorities should, above all, be up-to-date with the current programs enabling investment in water and sewage infrastructure. Due to a lot of competition among those applying for EU subsidies in this field, the submission of an application itself does not guarantee receiving a grant. For this reason, even before submitting the application, the municipality authorities should clearly identify the problems which they want to solve with the help of funds, as well as the financial possibilities of their own budget for this purpose. They should also be prepared when it comes to their knowledge of possible solutions appropriate for the municipality, they should also be aware of the number and type of documentation that will be necessary when submitting the application. The investments implemented by the municipality may cause an increase in new investors' interest, which in the future will result in an increase in the income of municipality budget.

Another issue is the appearance of new trends in technical solutions and modern design methods that draw the attention of both local authorities and residents. It is very important for the municipal authorities to get support from the local community in terms of implemented investments, which is why they often undergo the pressure and choose the solutions that are not always optimal and economically justified in a given area; very often they are expensive and generate high operating costs which the municipality cannot afford.

An additional problem related to obtaining EU funds is a long period of time between the moment of submitting the application and signing the contract for a subsidy. Implementation and its effects are visible sometimes only after several years, therefore applications for EU support are submitted hastily, without predicting future costs and burdens for the municipality.

3.2. Problems at the stage of the implementation of investment

In many municipalities in Poland there is no water supply concept. The basic problem associated with planning and developing rational water and sewage management is lack of data on water demand and sewage management, lack of strategic compilation on the well-balanced development of neighboring areas, insufficient information on the use of water and the expected structure and the amount of sewage.

Failure to complete the initial tasks results in dimensioning network fragments without any connection with the overall concept of water supply of the municipality as well as with the need to reconstruct relatively newly built sections of the network, and also with burdening designers with the necessity to select of wire diameters in the absence of data related to the water supply balance and planned development of the municipality, etc. Operational problems include water thefts connected with the availability of overground hydrants away from building sites, as well as thefts of water supply utilities, such as latches, cast iron boxes, etc.

Difficulties faced by the municipality prior to the investment implementation are the necessity to carry out compilations on the location of water resources and its acquisition as well as treatment methods. The problem is also to choose a design and an executive unit to implement the investment project. Most often, the cheapest contractor wins, which frequently results in using the cheapest elements of fixture, connective pieces, fittings, which after a short time of work malfunction and create operational problems. Some municipal utilities secure themselves against such problems by introducing into the contracts their requirements regarding water supply equipment elements, eg requirements for fittings (class, producer) – technical requirements are included in the specification of essential terms of the contract (SIWZ).

To sum up, the factors that can have a significant impact on the success of applying for co-financing the implementation of investments are primarily: identification of the investment demand through a wide analysis of the area, prior preparation of the project from the technical and formal-legal side, reliable, efficient and qualified team preparing and implementing the project, an attempt to predict possible operating costs generated by chosen solutions, as well as the right choice of the contractor of the investment, proper cooperation at every stage of the project (Krawczyk, 2014).

4. Possible solutions for water supply systems in rural areas

In the rural areas there will be two models of the water supply system - individual and collective one. Each of these systems should meet, according to the definition, the quantitative criterion, that is, provide water to the recipients in the appropriate quantity, as well as the quality criterion, i.e. the water supplied should have a certain quality (the same as treated water).

In the collective system, water is transported to individual recipients from a remote treatment station by means of water mains. The use of such a solution in case of single, remote settlement units in rural areas requires the construction of long sections, which entails high costs. If the farms do not perform any additional activities (cultivation, breeding), the water demand is small, which makes it difficult to dimension the pipes properly. At the same time, the collective system should anticipate its further development (Fitbór, 2014).

Another aspect which should be taken into account when planning the investment of a collective water demand system is to meet the fire protection requirements. The lowest requirements that must be met by the water supply network (regarding settlement units not exceeding 2000 inhabitants) is to ensure a capability of $5dm^3/s$, with a nominal diameter of the supply line equalling DN80. What frequently happens is that the fulfillment of this condition really means over-dimensioning water pipes and results in too low efficiency (due to low water consumption), which might have a negative impact on the quality parameters (due to too low water flow velocities). The collective system can be used in settlement units with high-density housing, where water is supplied to households located not far from each other, additionally conducting intensive production activities requiring large amounts of water.

Individual water supply systems are an alternative to the extensive magistral waterworks. These include local water treatment stations and individual water supply devices.

In order to meet the quantitative and qualitative criteria of such systems, analyzes of water coming from the source chosen for this particular purpose should be carried out already at the stage of creating the technological concept of the local water treatment station. On such a basis, it is possible to properly select individual processes, which will ensure the achievement of the required water quality parameters. The individual system of supplying settlement units with water allows to shorten the way of its transmission to individual households, as well as to reduce the length of water distribution pipes. However, it should be remembered that the costs of such a solution are also high - what should be taken into account are the costs of drilling wells, as well as the fairly high costs of individual technological line devices.

In case of planning the investment of water supply for settlement units in rural areas, a detailed analysis of the area should be carried out each time. The analysis should be based on the correct determination of water demand and it should take into account the possible development of the area. Workig out development plans plays an important role here. The analysis should also indicate at what maximum distances between settlement units and a water treatment station the purposefulness of water distribution, using magistral water pipes, is still profitable. Sometimes individual water systems and minimization of the amount of water taken in due to the use of alternative sources, eg for production purposes (rainwater tanks), questions the legitimacy of building an expensive treated water distribution system. At high costs of building the water mains and their length, supplying a small amount of water is no more profitable. Another thing is an attempt to predict the tendencies of development, as well as the possible activities of the areas.

5. *Further development plans of water supply systems in rural areas*

The water supply system that has been properly designed can be streamlined and modernized so that it runs efficiently and for a long time and minimizes unforeseen operating costs.

An important aspect is the elimination or limitation of water losses by monitoring important water supply networks and their operating parameters. Thanks to the metering of the network, it is possible to detect places of theft of water, as well as to control leaks on the network.

Collected documentation of the operated system in the form of GIS databases related to hydraulic modeling of network operation can also be helpful.

Introducing to the operational practice the GIS databases and monitoring the operation of water supply networks is a difficult enterprise in terms of logistics, content and investment. In particular, these problems occur in case of small water supply companies, mostly rural and municipal ones (Eksploatacja, 2013).

On the other hand, these water supply systems are not very complicated in terms of hydraulics, they consist of relatively few sections, nodes and rings (counted in tens, not in thousands, as in case of water systems serving municipal settlement units). The implementation of the GIS database or selecting posts for monitoring flows or pressure is quite simple at this stage of water supply system development (Kwietniewski, 2008).

An additional problem is the lack of staff of specialists in municipal enterprises operating local water supply systems, who could implement and use the aforementioned systems.

6. Summary and conclusions

In recent years, an increase in municipal spending on water supply and sewage infrastructure has been observed. It was influenced by the possibility of obtaining subsidies from various European funding programs. Despite the fact that water supply ceases to be a problem from the point of view of investment expenses, there is still a problem with finding an optimal solution in terms of economic viability and its quality.

The nature of rural housing imposes the need to choose between a collective water supply and sewage system, and individual, local solutions. This choice should be made taking into account technological, technical and economic aspects including both investment expenditure and operating costs. Municipalities should try to obtain EU subsidies for the development of rural areas, which will allow them to limit the implementation of investments selected on the basis of the lowest price criterion, the consequence of which are improperly operating systems.

Each decision of the planned investment should be preceded by an economic and technological analysis that takes into account the costs of the solution applied, as well as the issues regarding water quality and its stability, which are particularly significant in case of its distribution over great distances.

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Economic aspects of decentralized stormwater management – case study

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Abstract

Centralized stormwater management causing several important environmental and hydrologic impacts may be replaced by the decentralized stormwater management allowing to restore the natural water balance of urbanized catchment and to limit the pollution of surface water. However, assessment of stormwater system should be performed on all circles of sustainability, i.e. environmental, social and economic. The environmentally and technologically suitable on-site systems may present significant capital and operation costs limiting their financial profitability and public acceptance.

This paper present assessment of economic aspects of three proposed concepts of on-site, decentralized, stormwater management systems allowing partial treatment, retention, reuse and infiltration. The studied devices were designed for a group of buildings, parking lots and football pitch of school, located in a eastern part of Poland. The developed variants of on-site stormwater management covered application of drainage cells and drainage chambers in various combinations, additionally supplied by sediments and oils separators. The performed economic analysis was based on popular indicator of investment cost-efficiency: Net Present Value (*NPV*), Benefit-Cost Rate (*BCR*) and Dynamic Generation Cost (*DGC*). Our analysis was performed for 30 years time duration and with assumption of partial reuse of retained water. Additionally, the profitability of the developed concepts was tested in case of the possible obligatory stormwater fee. The performed calculations showed relatively low profitability and quite similar cost of the environmental effect for all tested systems. Thus, the social acceptance of rather costly non-affordable on-site systems may be limited, so their sustainability may be questioned.

Keywords

sustainable storm water management, economics, cost-efficiency

1. Introduction

The Water Frame Directive (2000/60/EC) sets biding ecological and chemical goals of surface and ground water bodies protection, related to introduced ecological standards, river basing management, sustainable water use and increased public involvement (van Rijswick, 2003; Termeer et al., 2011). Thus, to limit the emissions, waste water should be collected and treated by the appropriate and cost-effective methods as close to the source of pollution as possible. The above may be realized by sustainable stormwater management systems allowing on-site treatment, storage, infiltration and reuse, able to reduce environmental pressure caused by rainwater management (Niemcynowicz, 1993; Butler and Parkinson, 1997; Kärrman, 2001).

However, sustainable stormwater management systems should be assessed by the multicriteria analyses covering all circles of sustainability (Ellis et al., 2004; Palme et al., 2005; Benzerra, 2012) including environmental, social and economic (Harris et al., 2001, Harding, 2006), supported also by moral, technical, legal and political aspects (Pawłowski, 2009). So, assessment of stormwater management should be related to human health, environmental impacts and pressure, technology and operation, affordability, cost-efficiency and institutional appropriateness (Peter and Nkambule, 2012; Mara, 2007; Seleman and Bhat, 2016). In our opinion the nowadays available technology and know-how allow to achieve the comparable environmental, technical, social and legal indicators of sustainability. However, taking into account the fact that, the financial resources and proper management are required to sustain the long-term durability of decentralized stormwater on-site systems, these systems should be affordable in construction, operation and maintenance even for the local communities using their own resources or outside, e.g. governmental, founding. So, sustainability of on-site rainwater management may be negatively affected by maintenance and operation costs leading to possible low profitability and reduced acceptance by local communities or even single stake holders, combined with short-term thinking and actions related to all aspects of sustainability, from economics and investments to institutional and technical aspects (Harding, 2006; Widomski et al., 2011; Kwangware, 2014; Frone and Frone, 2015; Lewicka et al., 2016). So, in our opinion, analysis of economic sustainability seems to be very important in case of decision making and conceptual designing of sustainable on-site, decentralized stormwater systems. It should be performed not only basing on the direct design, capital, operation and maintenance costs but on the long term cost-effective analysis, based on several indicators of profitability allowing to assess its cost efficiency (Ellis et al., 2004; Becla et al., 2012).

This paper presents the attempt of practical assessment of multivariate cost efficiently analysis in assessment of several proposed decentralized systems of sustainable stormwater management for the selected group of public buildings located in SE part of Poland. The cost-efficiency of proposed on-site management were compared to financial profitability of the traditional centralized manner of stormwater management allowing the same ecological effect. There was also tested the influence of possible introduction of mandatory stormwater charge on long-term economic sustainability of proposed designs.

2. Materials and Methods

The presented studies were performed for three variants of decentralized sustainable stormwater management designs for a group of buildings, parking lots and football pitch of school, located in Chelm, Lublin Voivodeship, Poland. The total drained area was equal to 10947 m^2. The assumed areas of different subcatchments, together with the assumed values of runoff coefficients for specific type of surface sealing are presented in Tab. 1.

Table 1. Area of studied catchments and applied run-off coefficients

Subcatchment	**Area [m^2]**	**Runoff coeff. [-]**
Roof	2450	0.95
Concrete playground	1626	0.9
Parking lot	770	0.8
Grass playground	6101	0.2

The studied school is located in the region of mean yearly precipitation equal to 600 mm, with depth of soil freezing 1.2 m. The time of representative rainfall event was assumed as 15 minutes while its frequency c=5 years. The saturated hydraulic conductivity of the local sandy loam was assessed as equal to $Ks=7.5\cdot10^{-5}$ $m\cdot s^{-1}$.

The area is supplied with all the typical municipal networks: water supply, sanitary sewerage, stormwater system, gas, electric power supply as well as telecommunication.

Generally, the tested variants of sustainable stormwater management were design to collect, retain and infiltrate runoff water, only some excessive amount of water was assumed to be discharged to municipal stormwater system.

There were three variants of sustainable on-site, decentralized stormwater management developed:

- Variant I assumed management of runoff from grass playground by 10 S.C.-740 drainage chambers, by Ekobudex while stormwater from roofs, concrete playground and parking lot was designed as managed by set of 536 STORMBOX drainage boxes by Pipelife. The sediments separator CS 2000 was assumed to treat stormwater before entering infiltration devices. The runoff water from parking lot will be additionally treated by ACO Coalisator CCB Bypass 10/80/2500 oil derivatives separator.

- Variant II assumed application of 638 STORMBOX drainage boxes by Pipelife to store and infiltrate runoff water to soil. Again, the ACO CS 2000 was design to remove sediments from runoff water before entering drainage boxes. Stormwater from parking lot will be additionally treated by ACO Coalisator CCB Bypass 10/80/2500 oil derivatives separator.
- Variant III assumed application 34 S.C.-740 drainage chambers, by Ekobudex for sustainable management of storm water, allowing storage and infiltration. Sediments removal was designed as performed by ACO CS 2000 while ACO Coalisator CCB Bypass 10/80/2500 was selected to removal of oil derivatives.

In all the developed variants rain water on roofs will be collected by PVC drainpipes and transported by 200 mm PVC pipes to management devices. The linear drainage ACO Multiline V 100, Drainlock, class B 125 was design to collect and transport surface runoff water from parking lot and sport playgrounds. There were also designed catch basins with sediments collection for areas covered with grass. All the designed devices and pipelines were supported by the required inspection manholes, PRO 315 and PRO 630 by Pipelife.

Additionally, to allow comparison and further assessment, the traditional centralized system of runoff collection and discharge to stormwater network was designed. It was consisting of 160 mm PVD drain pipes, linear drainage and 200 mm PVC pipelines supported by PRO 315 and PRO 630 by Pipelife manholes as well as ACO CS 2000 and ACO Coalisator CCB Bypass separators, in order to obtain the same ecological effect.

For all the proposed variants of sustainable stormwater management system for the studied school the preliminary cost estimation and exploitation costs assumption, required for cost efficiency assessment, were performed. The preliminary cost assessment was based on the unit capital investment costs for each element of the designed system, i.e. realization of pipelines, manholes, linear drainage and storage/infiltration devices, including workload prices. The assumption of future exploitation costs was based on several sources, including publically available financial reports for the similar objects manuals, exploitation guidelines, services prices etc and covered all required services, including mainly control and flushing of pipelines and infiltration devices as well as operation costs of oil derivatives separator. The presentation of assumed investment and exploitation costs (recalculated for actual market prices from PLN to Euro) for each applied variant of the study was included in Tab. 2.

Table 2. Assumed investment and operation costs of tested stormwater management variants

Variant	**Investment costs [Euro]**	**Annual operation and maintenance costs [Euro]**
Decentralized I	86035	2228
Decentralized II	85565	2002
Decentralized III	73453	1691
Centralized	53297	1412

The assessment of economical aspects of sustainable stormwater management designed for selected school was based on three popular dynamic indicators of investments profitability, allowing to express the discounted value of money in time, usually used in multivariate analysis during conceptual stage of design. The following indicators were applied: Net Present Value (*NPV*), Benefit-Cost Rate (*BCR*) and Dynamic Generation Cost (*DGC*).

Net Present Value indicator presents collected sum of discounted cash flows, i.e. inflows and outflows (benefits and costs) reduced by the investment capital costs (Berry, 2007). The NPV for a positively assessed investment should have value even to, or greater than, zero ($NPV \geq 0$). *NPV*, including variable value of money and presented in monetary units for time duration of investment (n) may be calculated as follows (Miłaszewski, 2003):

$$NPV = \sum_{t=0}^{n} \frac{R_t}{(1+i)^t} \tag{1}$$

where: R_t – net cash flow (sum of all financial effects) for a i year of investment operation (Euro), i – discount rate (%), t – year.

Benefit-Cost Rate presents dimensionless ratio of benefits to costs of investment in studied year. The value of *BCR* indicator for profitable investment should be $BCR \geq 1$. *BCR* may be calculated using the below formula (Miłaszewski, 2003):

$$BCR = \frac{PV_b}{PV_c} \tag{2}$$

where: PV_b – present value of benefits (Euro), PV_c – present value of costs (Euro).

The Dynamic Generation Cost (*DGC*) expresses the discounted revenues equal to discounted costs of investment. So, *DGC* presents the price of ecological effect of the investment, i.e. discounted value, in Euro, of 1 m^3 of collected, preliminary treated and managed stormwater. This method presents the costs of investment in the popular, easily understandable values so is rather easily intelligible for designers, decision makers and authorities or representatives of local societies/governments. The principle of *DGC* application is very simple: the lower value of *DGC*, the more acceptable economically investment is. The *DGC* may be calculated using the formula (Rączka, 2002):

$$DGC = p_{EE} = \frac{\sum_{0}^{t=n} \frac{IC_t - EC_t}{(1+i)^t}}{\sum_{0}^{t=n} \frac{EE_t}{(1+i)^t}} \tag{3}$$

where: IC_t – investment costs in given year (Euro), EC_t – exploitation costs in given year (Euro), t – year of investment time duration, from 0 to n, where n is the last assessed year of investment activity (year), I – discount rate (%), p_{EE} – price of the ecological unit effect of the investment (Euro·m^{-3}), EE_t – ecological unit in given year (m^3).

As it was already presented, assessment of benefits is required for the proper calculations of *NPV* and *BCR*. Usually, on-site, local stormwater management systems do not generate incomes. But, on the other hand, several significant financial savings are possible. In this study, the *NPV* and *BCR* indicators were calculated for two cases: i) possible savings available due to usage of collected, treated and stored rain wastewater for grass playground/pitch watering and flushing sealed areas (parking lot and concrete playground); ii) as above but including also possible savings related to avoiding stormwater fee, in many regions of Poland obligatory for discharging surface runoff to stormwater network. Possible savings related to usage of stored water for flushing and grass watering were calculated basing on actual mean price of water in the region, flushed area and required dose, according to actual standards (Dz.U. 2002 nr 8 poz. 70). The assumed additional possible savings related to avoided surface runoff charge were based on calculated mean available price of 1 m^3 of discharged wastewater equal to 1.27 Euro (5.47 PLN), including VAT (www.retencja.pl). The described runoff water charge is based on the actual biding Polish law (Dz.U. 2006 nr 127 poz. 886; M.P. 2016 poz. 718) and is mandatory in numerous local water supply and sewage disposal companies.

All the selected economic indicators were calculated for the assumed period of operation equal to 30 years and discount rate 6%.

3. Results and Discussion

Table 3 presents values of all calculated indicators of economic efficiency for tested stormwater management systems, both, decentralized and centralized. Values of indicators presented in Tab. 3 suggest that in both cases, without introduced stormwater charge and with mandatory payment of the charge, which may be avoided by on-site runoff management, the profitability of investment is very low. In case of all variants, with or without mandatory stormwater charge payment, the investment and exploitation costs are significantly greater than the possible savings, due to avoiding payment and reuse of retained water. It is also visible that financial effectively of centralized system is directly related to local regulations concerning mandatory payment of rainwater charge.

Table 3. Calculated indicators of cost-efficiency for all developed variants of on-site stormwater management

Variant	*NPV* [Euro]		*BCR* [-]		*DGC* [Euro m^{-3}]	
	No charge	Charge introduced	No charge	Charge introduced	No charge	Charge introduced
Decentralized I	-102257	-75239	0.140	0.367	2.40	2.40
Decentralized II	-98451	-71433	0.145	0.380	2.33	2.33
Decentralized III	-81739	-54721	0.169	0.444	1.99	1.99
Centralized	-74138	-101156	0	0	1.50	2.04

The lowest cost value of ecological effect represented by *DGC* for cases without mandatory payment of stormwater fee was noted for centralized stormwater management, while the lowest among decentralized systems was observed for Variant III. There was also observed characteristic null value of *BCR* for centralized stormwater management, typical for investments generating no incomes or allowing no savings which could be treated as incomes.

Figures 1, 2 and 3 present direct comparison of *NPV*, *BCR* and *DGC* for all tested cases of decentralized management, also including mandatory payment of stormwater charge.

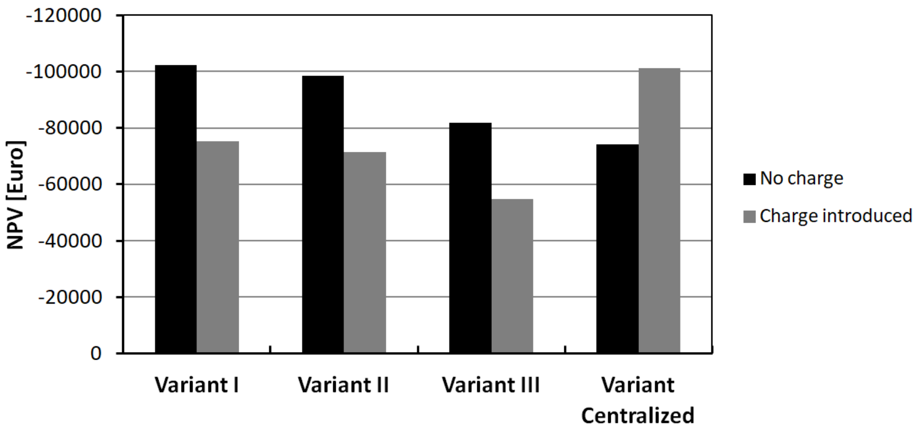

Fig. 1. Comparison of calculated NPV indicator for all tested cases

As it is visible in Fig. 1 in all compared cases *NPV* has negative values, both for benefits covering only possible savings due to water reuse as well as savings due to reuse and avoiding mandatory payment.

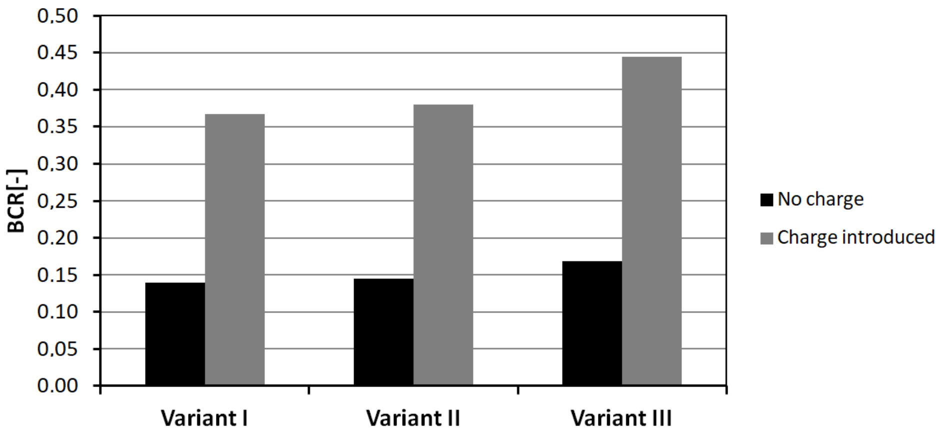

Fig. 2. Comparison of calculated BCR profitability indicator for all tested cases

The similar situation may be observed in Fig. 2 in case of *BCR* indicators calculated for all tested variants, also with taking into account stormwater charge. In all cases *BCR* < 1, thus all the variants of on-site decentralized storm water management presented negative profitability. Even in cases, for which avoiding payment of mandatory runoff water charge increased the possible savings, the observed increase in *BCR* value did not allow reaching the threshold *BCR* value of 1.0.

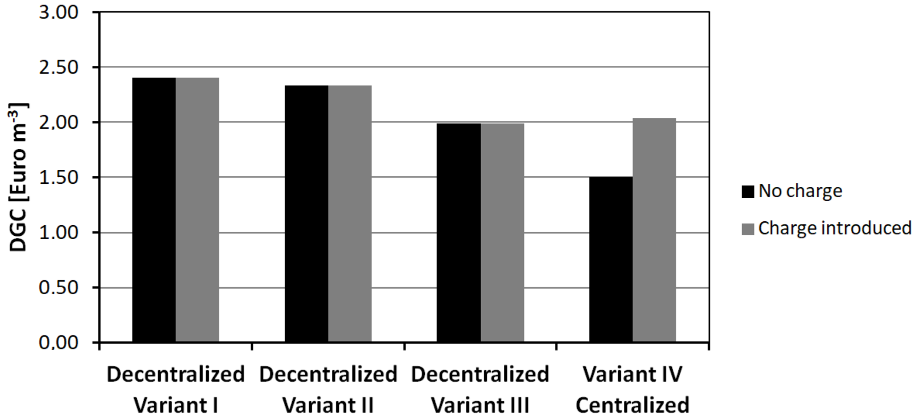

Fig. 3. Comparison of calculated DGC (price of ecological effect) indicator for all tested cases

In cases without mandatory payment of stormwater charge the *DGC* indicator, presenting the price of ecological effect was the lowest for the centralized variant of rainwater management (see Fig. 3). However, a very interesting observation was made, after introduction of mandatory stormwater charge, the *DGC* for centralized variant increased (because the payments elevate the operation and maintenance costs) to the level slightly higher with decentralized Variant III. Nevertheless, especially in case when payment of stormawter fee is not obligatory, the centralized variant presents significant economic advantage, combined with simple operation and limited services required, possibly reducing financial attractiveness, and influencing public acceptance of decentralized, on-site systems.

4. Summary and Conclusions

The performed sustainable economic analyses of three proposed variants of decentralized, on-site systems of storm sewage collection, management and treatment allowed the following conclusions:

- The applied simple and understandable indicators of cost efficiency assessment, usually used in the decision making process of the technical infrastructural investment allowed also assessment of economical sustainability of the on-site stormwater management proposed concepts.
- Our studies showed that despite meeting all environmental requirements the proposed on-site decentralized ways of stormwater management presented low profitability related to relatively high investment and maintenance costs as well as low possible savings.
- The tested variant of centralized rainwater management showed some economic advantages in case without mandatory payment of stormwater charge, presenting the lowest cost of ecological effect due to lower investment and operation costs.
- Thus, taking into account low profitability, even with included possible avoiding of stormwater charge, public acceptance of presented systems of on-site stormwater management may be questioned.
- Assuring sustainability of decentralized stormwater management, in conditions of Poland, especially for local, individual stakeholders may be a hard task, without any outside, governmental of EU community, financial support.

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Analysis of Changes in Waste Accumulation in Small Communities

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Abstract

The quantity and quality of municipal waste collected is basic information on the design and construction of waste management systems and installations, which are the basic (source) elements, and at the same time, they allow to anticipate effects of recovery or recycling of waste. The research on the quantity and composition of waste can allow to create a huge database, and supplemented with years of time can provide the material for analysis and to predict changes as well as to adjust the leachate treatment processes from landfills.

The analysis of such databases may be hampered by the fact that the data may cover types of buildings, different areas of urban development, different seasons, and equipment in technical and sanitary installations.

The aim of the article is to present the results of the research on the accumulation of waste from the residents of a small settlement unit and their analysis using the so-called pivot table. This analysis will allow the presentation of selected results and will facilitate their interpretation so that they will become more readable to the user and decision maker in the field of waste management.

Keywords:

Waste management system, households waste, waste removal, waste accumulation index, Pivot Tables

1. Introduction

Waste management systems are very complex which, along with the development of the territory concerned, require of a development concepts and continuous analysis. Municipal waste is characterized by variability in quantity and composition, and additionally, their parameters change over time.

Knowing the quantitative parameters of waste that is changing as a result of various conditions is important for the planning of the waste management system, including logistics and the design of specific solutions for systems or technologies that collect, recover, recycle and ultimately dispose of waste (2008/98/EC).

Factors that may affect such variation include the season, the degree of development and size of the surveyed region, the standard of living of the inhabitants, the type of dwellings and the number of public utilities. Therefore, for example, in urban areas where there is high saturation in shopping and service facilities, the amount of waste may vary considerably from those in residential areas (Barlaz at al., 2003).

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Despite numerous researches conducted so far in Poland and abroad, there are no sufficient theoretical bases for estimating the variations in the number of waste characteristics in a particular region. In each case, a series of studies must be carried out that provide the information needed to produce the optimum disposal method and the necessary basic engineering calculations. In addition, it is important to note that even full-scale waste surveys provide only information on current waste properties while the plant is designed to work over many years and must be adapted to the variable characteristics of the waste delivered. Therefore, for project purposes, information will be needed to predict the properties of the waste in the future. However, forecasting of all phenomena, including the properties of waste, is complex and as all forecasts have the property that in many cases it does not work, at least the general direction of change is necessary (2008/98/EC; Generowicz at al., 2014).

The aim of this article is to attempt to develop a method for forecasting and analyzing data on waste accumulation using Pivot Tables and data on waste collected on the urban transport routes of the waste collection company in a small locality near Cracow.

2. Materials and Methods

2.1. Waste characterization

For modeling purposes and easier balancing of the waste stream from the entire region, it is most often broken down into a number of streams characterized by: origins (households, businesses, shops, etc.), areas, differentiation resulting from different heating methods, habits of residents, etc. (Caruso at al.,1993, Kolekar at al., 2016)

Due to the fact that the characteristics of the waste can be variable both on individual days of the week, months, and at different times of the year, the factorial values are only likely to be obtained as a measure of specific cycle time period. Qualitative research results are obtained from measurements taken during the full week of each month of the year, in each study subject or environment. A systematic review of quantitative and qualitative changes is an important element of environmental monitoring. This provides opportunities to track trends in property changes and waste volumes, which allow us to develop ways and opportunities for recovery, recycling and disposal of waste. Systematic and properly monitored quantity and quality of municipal waste allows to determine the quantity, composition and the scientific possibilities of waste or to determine the tendency of changes over time (den Boer at al., 2010; Generowicz at al., 2011, Morrissey at al., 2004).

The scope of essential full waste testing should include four basic groups of indicators, ie accumulation index, physical properties, fertilization properties and fuel properties. Comprehensive testing of all indicators generally consists of two parts: field measurements and laboratory tests. Organization of research includes: selecting the required number of test environments, transit routes and objects in selected environments, as well as gathering information and characteristics of the examined objects and routes (Generowicz at al., 2011; Hajduga and Generowicz, 2017).

The choice of environments and objects for testing is not normalized, but results from the knowledge and needs of a particular region. They must be selected to provide the basis for the most reliable test results for the entire unit under study. Within each environment, a transit route or object is characteristic of the environment. The details of the address and function of the buildings or their activity, the number of inhabitants or employees, the living area (usable area), the type of facility, the type and quantity of waste containers, the types and systems of waste segregation should be collected from the survey area (Georgopoulou at al., 2008; Jamróz at al., 2012).

In order to perform full measurements and obtain reliable results, it should be considered: environment of diverse and characteristic technological properties of waste, ie multi-family housing development, modern; Old, compact housing in the city center, with significant saturation of service and administrative facilities; Single family housing built or distributed; Rural, farm development (Georgopoulou at al., 2008; Jamróz at al., 2012).

Each of these types of buildings will have a different accumulation and other technological characteristics of the waste. On the basis of measurements in individual environments it will be possible to determine average values of waste characteristics for a given settlement unit. Generally waste tests should be based on industry or state standards (Georgopoulou at al., 2008; Generowicz at al., 2012; Jamróz at al., 2012).

2.2. Accumulation of waste - research methodology

The basic quantitative characteristics of waste is their accumulation. Methods of determination of accumulation indices by mass and volume measurement are described in Polish industry standard BN - 87/9103 - 04. Quantitative tests allow to calculate the number of necessary means of transport, set the frequency of exports, necessary cubature and landfill area, also size and capacity of other disposal objects.

As a result, research allow to determine both the mass and volume indicator of received municipal waste. Mass index is the amount of solid waste collected in a unit of time, expressed in units of mass, referenced to one resident per square meter area or other unit of measurement. Accumulation ratios may be expressed in kilograms per capita per day [kg per capita/day]; In kilograms, per capita per year [kg per capita/year]; In kilograms per square meter of area per year [kg/m^3/year]. Volume indicator of waste accumulation is the volume of waste collected in a specified unit of time, loosely packed (no artificial compaction) in containers, expressed in units of volume, referenced to one resident, unit area or other unit of conversion (Sieja, 2006).

The reliable mean values of the waste accumulation index, can only be obtained, as the mean of the annual test cycle. Measurements are made according to a predetermined schedule, at specified intervals, which is then taken into account in the calculations (Sieja, 2006).

As it was already mentioned, permanent transit trails are selected for conducting environmental studies which they should be inventoried. Stocktaking should include basic information about the number of inhabitants in each property, type of public facilities, type and number of containers. Field tests are conducted according to the form (Fig. 1) (Generowicz at al., 2011; Jamróz at al., 2012).

Date:			**City:**				
Transit route number:							
Car brand and number:							
Weight of the empty car:							
Weight of the loaded car							
No.	**Street name and house No.**	**Number of container**	**Level of filling the containers [%]/Containers capacity [m³]**				
1	Wiejska 9	3	90 /1,1	75 /0,11	80 /1,1	-	-
2	Wiejska 11	2					

Fig. 1. Exemplary form for conducting field measurements

Prior to entering the route, the car is weighed. The volume filling of the containers is determined by the visual assessment of the filling of the container, specified in [%] of the filling. After the measurements are completed, the car is re-weighed to determine the weight of the waste taken from the route being tested (Beigl at al. 2004; Skalmowski at al., 2004; Generowicz at al., 2011). Volume accumulation index is calculated in accordance with Eq. (1):

$$b_{obj} = \frac{\sum_{i=1}^{n} v_i}{M \cdot d} \left[\frac{dm^3}{capita \cdot day}\right] \left[\frac{m^3}{capita \cdot year}\right] \tag{1}$$

where: v_i – volume of waste in individual containers [dm^3], *capita* – number of inhabitants on the route under study and *day* – number of days of waste collection.

The mass accumulation index is calculated in accordance with Eq. (2):

$$b_m = \frac{m}{M \cdot d} \left[\frac{kg}{capita \cdot day}\right] \left[\frac{kg}{capita \cdot year}\right] \tag{2}$$

where: *m* – mass of waste collected from the object or measuring route, calculated as the mass difference of the car to carry waste: full and empty [kg], *capita* – number of inhabitants on the route under study and *day* – number of days of waste collection.

In addition, by knowing the weight and volume accumulation indices, the bulk density (weight by volume) of waste according to Eq. (3) can be calculated:

$$G = \frac{b_m}{b_{vol}} \left[\frac{kg}{m^3}\right] \tag{3}$$

where: b_m – mass accumulation index $\left[\frac{\text{kg}}{\text{capita} \cdot \text{year}}\right]$ and b_{vol} – volume accumulation index $\left[\frac{\text{kg}}{\text{capita} \cdot \text{year}}\right]$.

2.3. Pivot Tables

Being able to quickly analyze data can help make better decisions. But sometimes it’s hard to know where to start, especially when you have a lot of data. PivotTables are a great way to summarize, analyze, explore, and present data. PivotTables are highly flexible and can be quickly adjusted depending on how you need to display your results. You can also create Pivot Charts based on PivotTables that will automatically update when Pivot Tables do.

3. Description of the area and the results of the analysis

The research of the settlement unit was carried out within one calendar year on routes serviced within three days of the week ie. Wednesdays, Thursdays and Fridays. Considered site is located about 50 km from Krakow and has about 40 thousand inhabitants. The studied routes included 24 streets both single and multi-family (Tab. 1), which were divided into 12 routes (marked alphabetically from A to L).

Table 1. The names of the streets surveyed with the division into single and multi-family housing (elaboration of authors)

No.	Street	Single family housing	Multi-family housing
1	29 Listopada		X
2	Borek Szlachecki	X	
3	Daszyńskiego		X
4	Dąbrowskiego		X
5	Głowackiego		X
6	Jaśminowa	X	
7	Jodłowa	X	
8	Kalinowa	X	
9	Kilińskiego		X
10	Kopernika	X	
11	Kościuszki		X
12	Krakowska		X
13	Kraszewskiego		X
14	Niepodległości		X
15	Piłsudskiego		X
16	Poniatowskiego		X
17	Sadowa	X	
18	Słoneczna		X
19	Spokojna		X
20	Spółdzielcza		X
21	Wesoła		X
22	Wspólna		X
23	Wyspiańskiego	X	
24	Żwirki i Wigury		X

Field studies allowed the predetermined indexes to be extracted, and the use of PivotTable has significantly facilitated their analysis and interpretation. Below will find the results of the research both in tabular form (Tab. from 2 to 11) and graphs (Fig. from 2 to 12).

Table 2. Chart of sum of volume of collected waste (elaboration of authors)

Street	Sum of collected waste during time of observation [dm^3]
29 Listopada	38559.64
Borek Szlachecki	184422.50
Daszyńskiego	11057.20
Dąbrowskiego	55668.80
Głowackiego	65273.18
Jaśminowa	2880.00
Jodłowa	7836.40
Kalinowa	1381.20
Kilińskiego	32564.40
Kopernika	100873.80
Kościuszki	34225.40
Krakowska	16222.20
Kraszewskiego	149099.80
Niepodległości	29099.95
Piłsudskiego	1100.00
Poniatowskiego	28164.40
Sadowa	107387.50
Słoneczna	5505.50
Spokojna	56045.00
Spółdzielcza	25523.30
Wesoła	134600.40
Wspólna	32564.40
Wyspiańskiego	20620.00
Żwirki i Wigury	13092.20
TOTAL	**1153767.17**

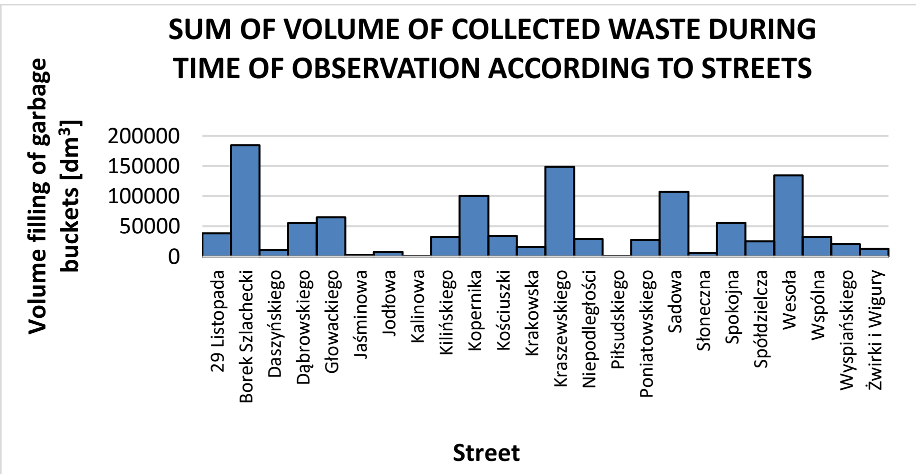

Fig. 2. Chart of sum of volume of collected waste (elaboration of authors)

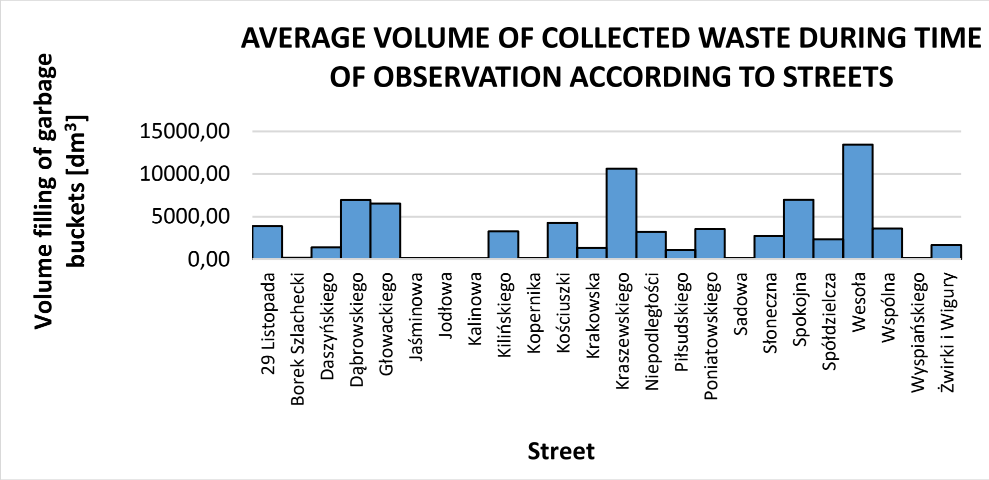

Fig. 3. Chart of average volume of collected waste (elaboration of authors)

Table 3. Sum of volume of collected waste according to the season of the year (elaboration of authors)

Season of the year	**Sum of volume of collected waste during time of observation [dm^3]**
Summer	478365.28
Winter	675401.89

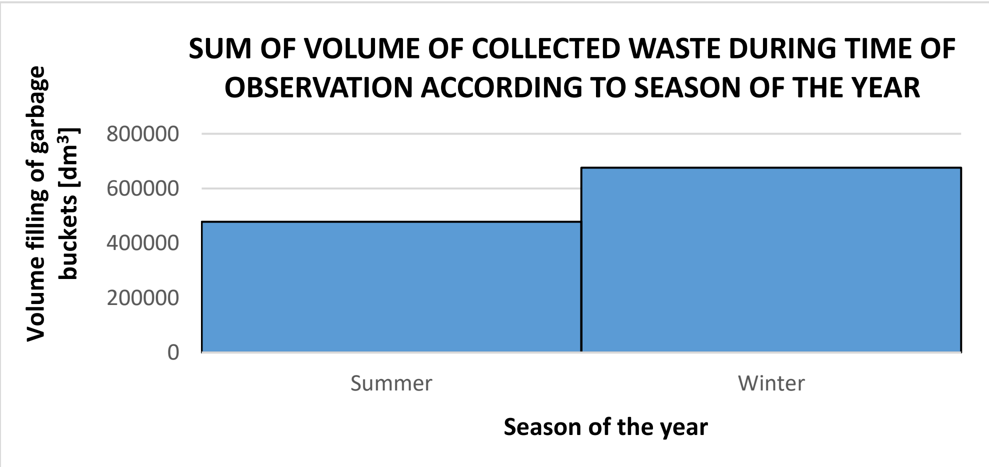

Fig. 4. Sum of volume of collected waste during time of observation according to season of the year (elaboration of authors)

Tab. 4. Volume of collected waste according to the season of the year – summer (elaboration of authors)

Street	**Sum of volume of collected waste during time of observation [dm^3]**
Summer	**478365.28**
29 Listopada	9955.00
Borek Szlachecki	39522.50
Daszyńskiego	4730.00
Dąbrowskiego	20735.00
Głowackiego	30064.37
Jodłowa	3343.90
Kalinowa	481.20
Kilińskiego	14960.00
Kopernika	56227.30
Kościuszki	12221.00
Krakowska	7920.00
Kraszewskiego	72630.00
Niepodległości	5995.00
Piłsudskiego	1100.00
Poniatowskiego	11110.00
Sadowa	50975.00
Spokojna	25795.00
Spółdzielcza	12320.00
Wesoła	56760.00
Wspólna	15510.00
Wyspiańskiego	20620.00
Żwirki i Wigury	5390.00

Table 5. Volume of collected waste according to the season of the year – winter (elaboration of authors)

Street	Sum of volume of collected waste during time of observation [dm^3]
Winter	**675401.89**
29 Listopada	28604.64
Borek Szlachecki	144900.00
Daszyńskiego	6327.20
Dąbrowskiego	34933.80
Głowackiego	35208.80
Jaśminowa	2880.00
Jodłowa	4492.50
Kalinowa	900.00
Kilińskiego	17604.40
Kopernika	44646.50
Kościuszki	22004.40
Krakowska	8302.20
Kraszewskiego	76469.80
Niepodległości	23104.95
Poniatowskiego	17054.40
Sadowa	56412.50
Słoneczna	5505.50
Spokojna	30250.00
Spółdzielcza	13203.30
Wesoła	77840.40
Wspólna	17054.40
Żwirki i Wigury	7702.20

Based on the results obtained, it can be seen that more waste is collected in multi-family housing as in the winter season. This is due to the way people live, holidays, etc. In addition, it is often the case that outsiders are using garbage dumps at blockades to avoid additional waste for over-debris.

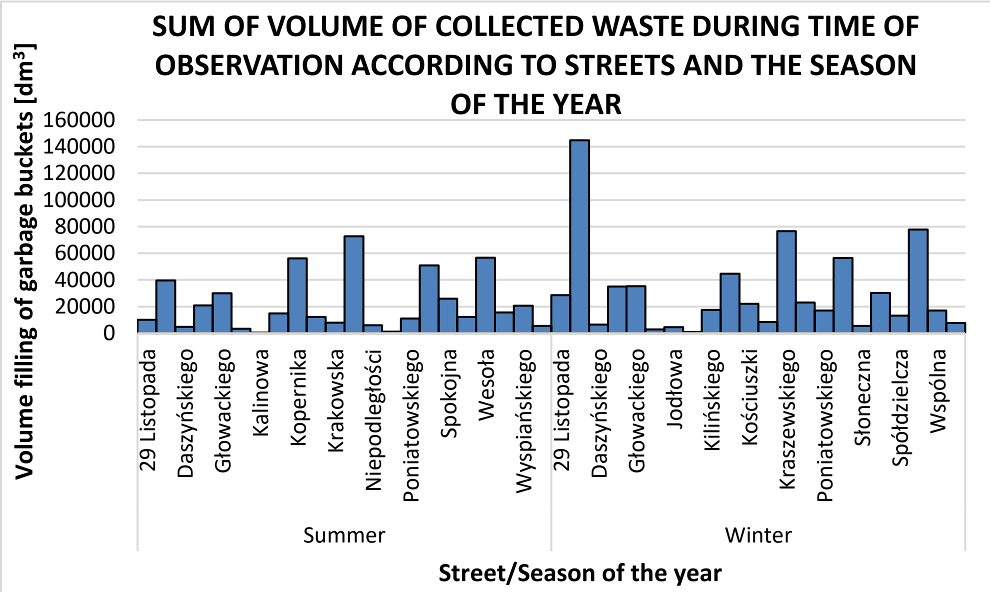

Fig. 5. Chart of volume of collected waste according to the season of the year (elaboration of authors)

Table 6. Sum of volume of collected waste according to routes (elaboration of authors)

Route	Sum of volume of collected waste during time of observation [dm^3]
A	122173.90
B	29390.00
C	49385.00
D	9900.00
E	105075.54
F	39050.00
G	99605.00
H	135885.23
I	184422.50
J	150630.00
K	39600.00
L	188650.00
TOTAL	**1153767.17**

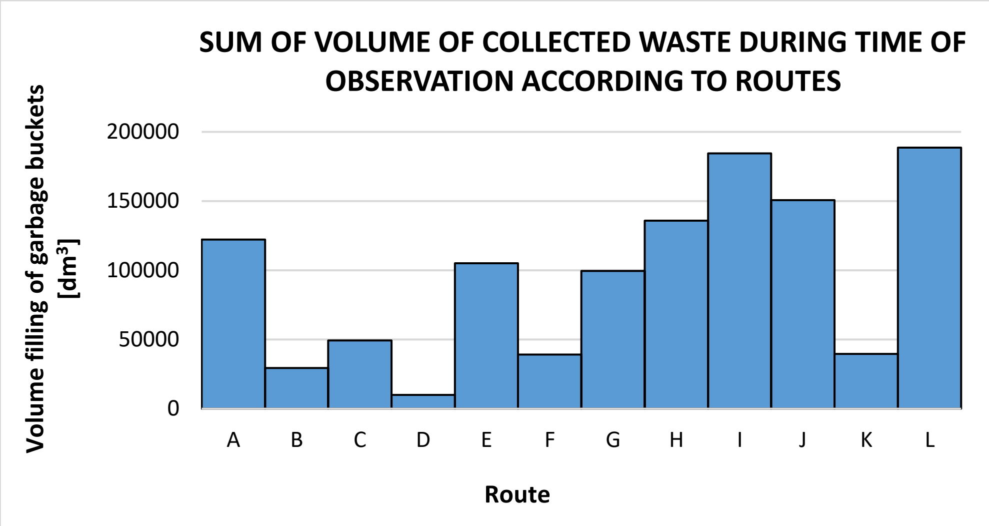

Fig. 6. Sum of volume of collected waste during time of observation according to routes (elaboration of authors)

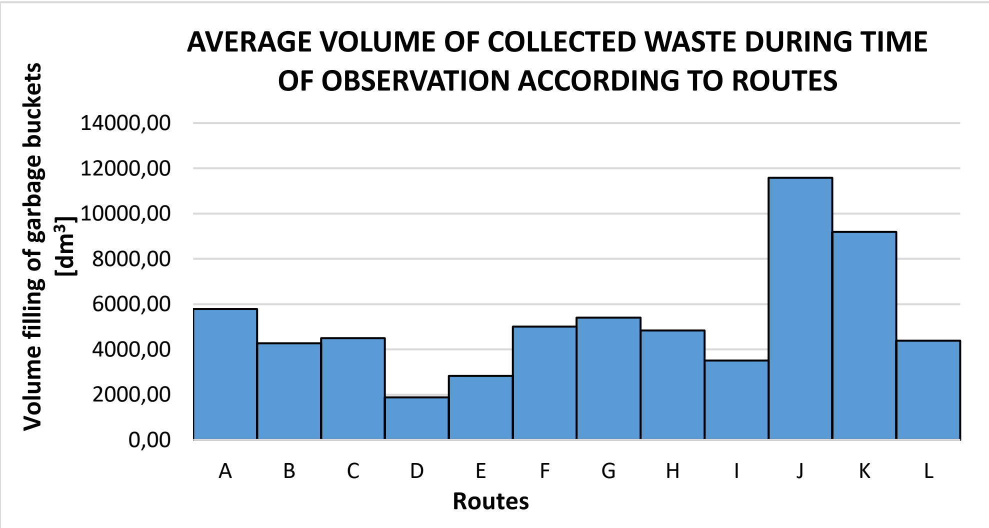

Fig. 7. Average sum of volume of collected waste during time of observation according to routes (elaboration of authors)

Table 7. Sum of volume of collected waste according to day of the week (elaboration of authors)

Day of the week	Sum of volume of collected waste during time of observation [dm^3]
Friday	128286.50
Thursday	740605.77
Wednesday	284874.90

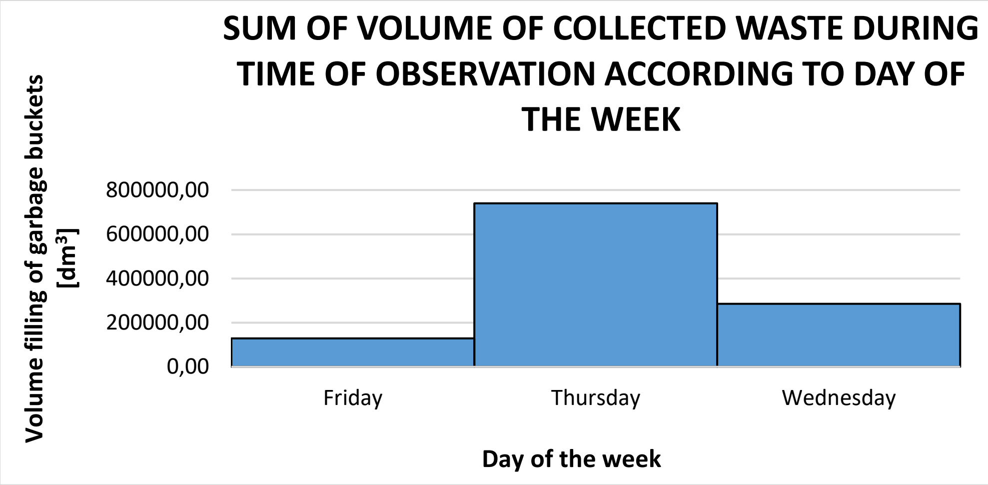

Fig. 8. Sum of volume of collected waste during time of observation according to day of the week (elaboration of authors)

Table 8. Average value of the Accumulation Index according to routes (elaboration of authors)

Route	Average value of the Accumulation Index
A	0.84
B	0.56
C	0.55
D	0.37
E	0.53
F	0.35
G	0.58
H	0.49
I	0.60
J	0.81
K	1.52
L	0.47

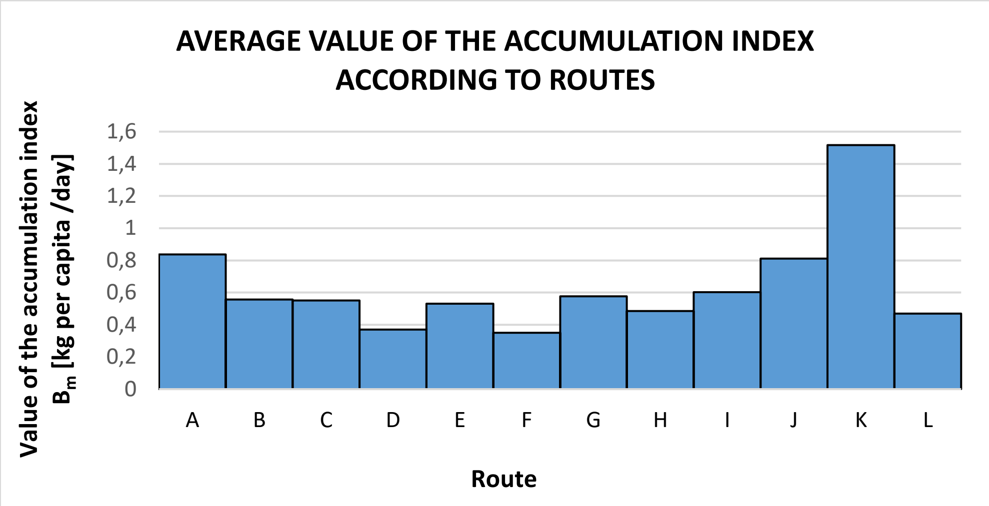

Fig. 9. Average value of the Accumulation Index according to routes (elaboration of authors)

Table 9. Average value of the Accumulation Index according to the season of the year (elaboration of authors)

Season of the year	**Average value of the Accumulation Index**
Summer	0.54
Winter	0.68
Annual average	**0.61**

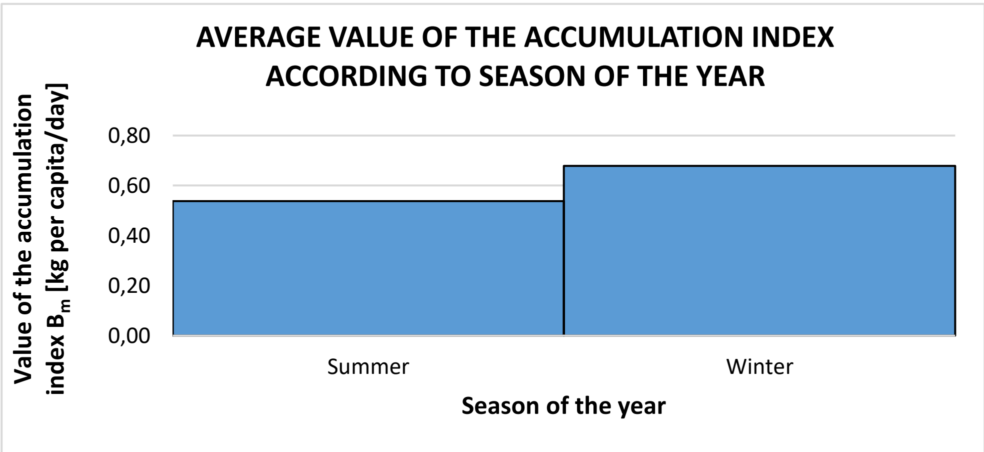

Fig. 10. Average value of the Accumulation Index according to season of the year (elaboration of authors)

Table 10. Average value of Volume Indicator of Waste Accumulation B_{vol} (elaboration of authors)

Row labels	**Average value of Volume Indicator of Waste Accumulation B_{vol} [m^3 per capita/year]**
A	1.10
B	1.10
C	1.14
D	1.50
E	2.49
F	1.01
G	1.58
H	1.99
I	1.01
J	1.67
K	2.39
L	2.51
Total average	**1.62**

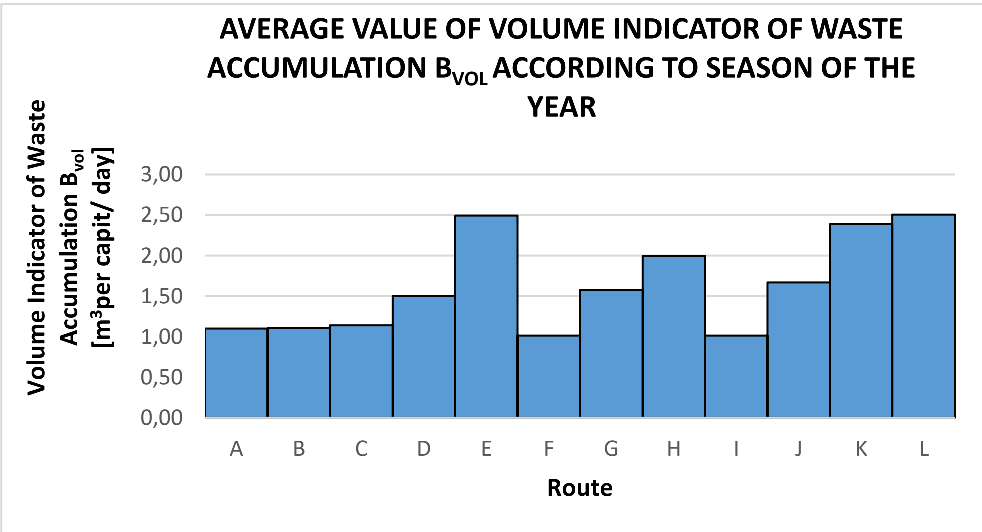

Fig. 11. Average value of Volume Indicator of Waste Accumulation B_{vol} (elaboration of authors)

Table 11. Average value of Volume Indicator of Waste Accumulation B_{vol} according to season of the year (elaboration of authors)

Season of the year	**Average value of Volume Indicator of Waste Accumulation B_{vol} [m^3 per capita/year]**
Summer	1.35
Winter	1.86
Annual average	**1.62**

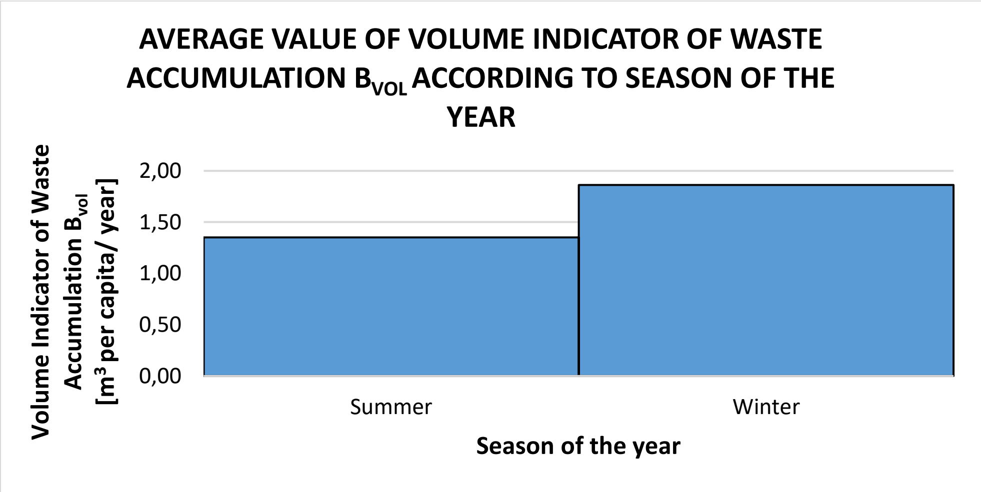

Fig. 12. Average value of Volume Indicator of Waste Accumulation B_{vol} according to season of the year (elaboration of authors)

The results clearly show that some waste collection routes are much more burdensome than the rest, as are the days of the week. For this reason, you can try to either translate individual streets into other routes or weekdays, because of the lack of data from Monday and Tuesday, you can not recommend it. It is important to note that observations from one year may not be sufficient. They should last a few years so that it is possible to take advantage of the city's underdevelopment.

4. Summary and Conclusions

- Waste research is a very important part of waste management planning. Their successive monitoring and monitoring allow you to plan the functioning of the entire system as well as operational activities during its operation.
- The research has allowed us to determine the accumulation of weight and volume waste, which lies within the limits of the literature and can be used as data for similar settlement units.

- Such research is a measurable effect of the implementation of waste management plans for regional systems, and their results may be the basis for calculating real waste handling charges and the use of waste management systems.
- The research was carried out for single-family and multi-family buildings with a population of around 6,000, this value was adopted on the basis of the average number of dwellings in the building with the assumption of three residents per dwelling. The average value of mass waste accumulation for designated routes is: 222.65 [kg per capita and per year], which is in the range of 180–330 [kg per capita and per year] (according to the Institute of Urban Development, 2004).
- The average volume of waste accumulation for routes of designated routes is: 1.62 [m^3 per capita per year], which is in the range of 1.5–1.9 [m^3 per capita and per year] (according to the Institute of Urban Development, 2004). As the research included the removal of urban waste, the calculated mass storage index did not reach the upper limit of the range.
- The increased rates of indicators in the winter season are affected by the fact that the calendar year was divided into only two periods: winter and summer. Both March and October were included in the winter season, when residents, especially in small towns, carry out cleaning works around their property. In addition, it is the holiday season, which also affects the increased amount of waste.

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Dairy processing wastewater treatment

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Abstract

The dairy industry is one of the leading in the food industry of Ukraine. However, in the production of dairy products, a large amount of highly concentrated wastewater is formed. Wastewater from the dairy industry is characterized by high concentrations of organic contaminants, elevated concentrations of suspended solids, fats and phosphates. Concentrations of contaminants and pH values vary significantly during the day. Due to the reduced use of water during dairy processing, the concentration of wastewater contaminants has recently increased significantly. All these circumstances require the use of new efficient technologies for dairy enterprises wastewater treatment. In the NUWEE a new effective technology for dairy industry wastewater treatment was developed and tested under production conditions. The technology provides removal of large solid particles from dairy processing wastewater on screens, sand and other large mineral particles - on sand traps. For wastewater equalization and leveling the pH value, preventing the wastewater souring, they are fed directly into the aeration tank-mixer with a jet aeration system. Separation of the sludge mixture is carried out in a secondary settling tank, built directly into the aeration tank. In Shostka city dairy plant the aeration tank provided a reduction in suspended solids average concentrations by 52.5%, COD by 94.4% BOD_{20} by 98.4%, BOD_5 by 98.6%, ammonia nitrogen by 93.3%, phosphates by 89.6% and fats by 80.3% before discharging treating wastewater to the city sewer. On the F/M ratio from 80 to 1450 mg COD/d per g MLVSS the efficiency of biological treatment by COD is in the range of 90–98%. The oxidizing power of aeration tanks for BOD_{20} was in the range of 407–2276 mg/(m^3 day). At elevated concentrations of suspended solids in raw wastewater, pressure flotation should be used before biological treatment. The efficiency of Zolotonosha butter factory wastewater reagentless flotation treatment by COD was 10.9–67.8% (average 30.8%), and the residual concentration of suspended solids did not exceed 210 mg/dm^3. If it is necessary to discharge treated wastewater into surface water, two-step biological treatment and post-treatment of wastewater on filters with floating polystyrene loading should be used.

Keywords

dairy processing wastewater, wastewater contaminants concentration, aeration tanks-settlers of large hydraulic height, settlers-flotators

1. Introduction

According to the State Statistics Service of Ukraine in 2016 it was produced (ths. tons): liquid milk processed 926, cream butter 101; fresh not fermented cheese 69,6; fat cheese 113; yoghurts and other fermented or sour milk and cream 420 (Ukrstat, 2017). The dairy industry is one of the leaders in the food industry of Ukraine. However, the production of dairy products is accompanied by the formation of a large quantity of highly concentrated wastewater, which has a negative effect on the environment.

Facilities calculation for the dairy wastewater treatment are usually provided on the basis of ‘Designer's guide” (Designer's, 1981). However, for more than thirty-five years since its publication, significant technological changes have taken place in the dairy industry. For modern enterprises with new types of dairy products it is sometimes practically impossible to attribute it to one of the types indicated in the “Designer's Guide” (Tab. 1). Reduction water use in production led to a change in the composition and properties of the dairy industry wastewater, which should be taken into account in calculations of treatment facilities.

Table 1. Characteristics of wastewater dairy processors (Designer's, 1981)

Parameter	**The values of wastewater pollutants concentrations for enterprises**		
	City dairy plant	**Dry and condensed milk plant**	**Cheese factories**
TSS [mg/dm^3]	350	350	500
COD [mg/dm^3]	1400	1200	3000
BOD_{20} [mg/dm^3]	1200	1000	2400
Total nitrogen [mg/dm^3]	60	50	90
Phosphorus [mg/dm^3]	8	7	16
Fats [mg/dm^3]	100	100	100

Notes: 1. The characteristics of butter plants and the production of whole milk substitutes wastewater are roughly accepted for data on dry and condensed milk production, while production of casein and sour cream-cheese products is based on data for cheese factories.
2. The concentration of fats in waste water from the shops producing high-fat products (butter, cream, sour cream) is 200–400 mg/dm^3.

Since 1981 in the food wastewater treatment branch research laboratory at NUWEE more than 100 dairy enterprises in Ukraine were surveyed. This allowed the accumulation and generalization of a significant factual material concerning the composition and properties of wastewater from various dairy enterprises. For example, parameters of wastewater pollution of several dairy enterprises producing hard and processed cheeses, are shown in Table 2.

Table 2. Wastewater characterization of different groups of dairy-processing plants (*Kovalchuk, 2012*)

Parameter	**Wastewater pollution for some enterprises producing**			
	hard cheeses			**processed cheeses**
pH	4.18–6.37 5.56	4.0–6.0 5.4	5.5–9.08 6.98	6.8–7.2 7.0
TSS [mg/dm^3]	248–867 493	273–634 396	214–613 385	255–1884 634
COD [mg/dm^3]	910–6664 4116	2222–6370 3721	1133–4860 2242	720–3480 1528
BOD_{20} [mg/dm^3]	760–4508 3547	-	-	418–1960 982
BOD_5 [mg/dm^3]	590–3925 3335	1333–3680 2141	506–2840 1428	390–1330 699
Fats [mg/dm^3]	32–146 66	-	180–375 276	-
Ammonia nitrogen [mg/dm^3]	5.8–8.8 7.2	14.0–30.2 22.6	0–18 4.3	9.2–15.7 12.4
Nitrate (N) [mg/dm^3]	29.4–37 33.2	-	0–0.9 0.1	-
Phosphate [mg/dm^3]	49–295 210	-	26–45 37	-

Minimum and maximum value in the ranges and mean value below the ranges.

Wastewater from hard cheese production enterprises is poorly acidic or is close to neutral pH reaction. The meaning of their COD, BOD_{20} and BOD_5 slightly higher than indicated in Table 1. They are characterized by low ammonium nitrogen content. However, the phosphate content is quite significant, which is due to the use of detergents. The established analytical dependence between the values of COD, BOD and suspended solids is shown in Figure 1 and in Table 3.

Investigation of wastewater sedimentation from cheeses processing showed that simultaneously with the decrease of suspended solids, there was a decrease in COD waste water (average values) by 56.6%, BOD_{20} by 34.2%, BOD_5 by 33.4%.

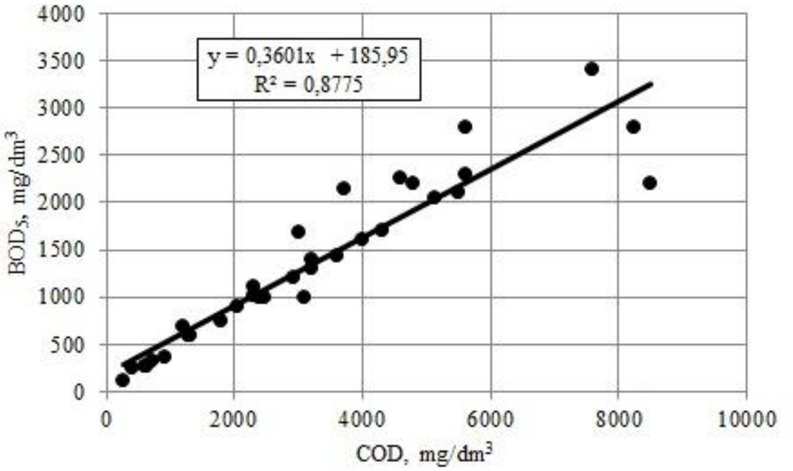

Fig. 1. The relationship between BOD_5 and COD cheese factories wastewater (Kovalchuk, 2012)

Table 3. Analytical relationship between concentrations of wastewater contaminants of some dairy processors (Kovalchuk, 2012)

Name of enterprises	**Analytical expressions**	***R*²**
Cheese factories	$BOD_5 = 0.36COD + 186, mg/dm^3$	0.88
Cheese factories	$BOD_{20} = 0.8COD, mg/dm^3$	Shifrin and Mishukov, 1968
Cheese factories	$BOD_5 = 0.7BOD_{20} + 40, mg/dm^3$	Shifrin and Mishukov, 1968
Butter factory	$BOD_{20} = 0.83COD + 30, mg/dm^3$	0.94
Butter factory	$TSS = 0{,}169COD + 150, мg/dm^3$	0.28
City dairy plant	$BOD_{20} = 0.867COD - 33, mg/dm^3$	Shifrin and Mishukov, 1968
City dairy plant	$BOD_5 = 0.42COD + 910, mg/dm^3$	0.71
City dairy plant	$TSS = 0.255COD + 2660, mg/dm^3$	0.19

Figure 2 shows the relationship between the concentration of suspended solids and COD in the butter factory wastewater. However, the value of coefficient determination R^2 for this dependence is 0.28, and the value of R^2 for the similar dependence obtained for the city dairy plant wastewater is only 0.19 (see Table 3). Therefore, it can only be argued that the concentration of suspended solids for most dairy industry enterprises is approximately 15–25% of the value of the COD, which is fully adequate with the conclusions made in (Shifrin and Mishukov, 1968; Shifrin, 1981).

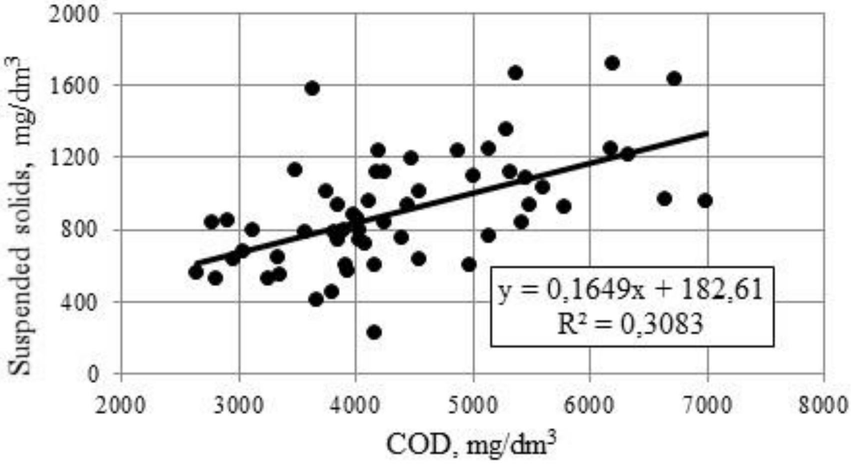

Fig.2. The relationship between concentrations of suspended solids and COD butter factories wastewater (Shifrin and Mishukov, 1968)

As follows from Table 4, the concentration of organic wastewater contaminants for butter factories, city dairy plant and milk condensed plant in terms of COD, BOD_5 and BOD_{20} generally also exceed the values shown in Table 1. It is obvious that the application of new technologies reduces milk losses in production. However, at the same time there is a significant reduction in the amount of water used: - when washing tanks and flasks for the transportation of milk; - when cleaning and washing process equipment and pipelines; - at elimination of spills and leaks; - when cleaning workshops, etc. This leads to increasing of wastewater pollution concentrations. In 1981 the specific amount of wastewater generated at the dairy industry enterprises was 1.6–5.5 m^3 per ton of processed milk (Designer's, 1981), while according to the World Bank data it is now 1–2 m^3/t (World, 1998), and the goal is to reduce the cost of generated wastewater to 1 m^3 per ton of processed milk. For dairy enterprises in the United Kingdom, a reduction in the specific consumption of water from 5.8 to 0.6 m^3/t of processed milk with an increase in annual productivity of enterprises from 80 to 520 thousand tons of milk per year is established (Komisja, 1996; Carawan at al.). In one of the milk processing enterprises in Poland, growth of wastewater BOD_5 from 1703 mg/dm^3 in 2000 to 2582 mg/dm^3 in 2010, or 1.5 times (Dąbrowski, 2011), was marked.

Table. 4. Wastewater characteristics of butter factories, city milk plants and milk condensed plants (Kovalchuk, 2012)

Parameter	**The values of wastewater pollutants concentrations for enterprises**			
	Butter factories	**City milk plants**		**Milk condensed plants**
pH	5.9	6.13–11.9 9.0	4.4–8.6 6.4	-
TSS [mg/dm^3]	2860	30–1204 366	494–3069 902	448–602
COD [mgO_2/dm^3]	5304	164–10547 2569	687–10270 5650	1100–2210
BOD_{20} [mgO_2/dm^3]	-	-	-	920–1870
BOD_5 [mgO_2/dm^3]	3126	115–6245 1723	410–5893 3559	890–1440
Fats [mg/dm^3]	-	18–131 82	-	-
Ammonia nitrogen [mg/dm^3]	9.11	0.8–50 12.5	95–262 113	25–39
Nitrite (N) [mg/dm^3]	2.23	-	-	-
Nitrate (N) [mg/dm^3]	56.3	-	-	-
Phosphate [mg/dm^3]	-	2.2–39 13	-	-

Increased concentrations of wastewater pollutants are also characteristic for the new profile dairy industry enterprises. In particular, the wastewater of the enterprise of condensed milk whey production is characterized by the following indicators: pH 6–9; suspended matter 2264 mg/dm^3; COD 1736 mg/dm^3; BOD_5 1447 mg/dm^3; total nitrogen 48.3 mg/dm^3; phosphates 6.8 mg/dm^3; fats 62.4 mg/dm^3. The concentrations of suspended matter in ice cream production wastewater can reach 8000 mg/dm^3; COD 6000 mg/dm^3; BOD_{20} 4400 mg/dm^3; fats 2500 mg/dm^3; ammonia nitrogen 12 mg/dm^3; total phosphorus 30 mg/dm^3; detergents 2 mg/dm^3. The pH value of this category of wastewater is 6.0–11.0 (World, 1998).

2. *Materials and Methods*

Depending on the purpose and production capacity dairy industry enterprises working full time or on a shift-basis. The generated wastewater is highly concentrated on the contents of suspended solids, fats and contaminants, oxidized by biochemical and chemical methods, have low ammonia nitrogen and high phosphate concentration. The pH of wastewater can fluctuate in considerable limits - from 4 to

11, which is determined by the applied technologies of milk processing and washing of equipment.

Biological treatment is the main method of milk processing enterprises wastewater treatment. For this purposes are used aeration tanks or biofilters (Designer's, 1981; Shifrin at al., 1981; Shifrin and Mishukov, 1968). Preliminary wastewater treatment is carried out on grates, sand traps, fat traps, clarifier-digesters. Abroad (Baisali, 2006; Bharati and Shinkar, 2013a; Bharati and Shinkar, 2013b; Kushwaha, 2011; Use, 2008) preliminary wastewater treatment is carried out on grates, sand traps, in equalizers and in flotators operating with the use of reagents-neutralizers, coagulants and flocculants. When discharged treated wastewater into the urban sewage systems is limited to anaerobic treatment, and when discharging into open water used anaerobic-aerobic treatment.

The choice of concrete dairy enterprise wastewater treatment technological scheme is a complex technological, technical and economic task. First of all, the wastewater treatment technological scheme depends on the quality parameters of treated wastewater, the required degree of wastewater treatment and their cost. Extremely important is the impact on the technological scheme of local conditions – the location of treatment facilities in or outside the enterprise, the size and topography of the area for treatment facilities, geology, groundwater level, sanitary protection zone sizes, availability of sources of power supply, communications, etc.

Today in many dairy enterprises realize a reconstruction or building of new wastewater treatment plants, usually using foreign equipment or technology. Their adaptation to local conditions require consideration of many features relating to the composition and properties of treated wastewater, the required degree of wastewater treatment, technical level of treatment facilities, etc.

In particular, it should be taken into account that the dairy processing enterprises wastewater has high bacterial contamination. The major part of bacteria is the lactic bacteria, represented by homo- and hetero-fermentative bacteria. Along with the lactic bacteria, the pathogens of alcohol, vinegar and butyric acid fermentation, as well as mushrooms and yeast, enter and develop into wastewater. The presence in wastewater homo-fermentative lactic bacteria is causing the rapid wastewater souring under anaerobic conditions, because the end product of their life is lactic acid. Therefore, the duration of wastewater in anaerobic conditions should be minimal. Creation of early aerobic conditions by aeration of wastewater: inhibits the development of lactic anaerobic microorganisms; create favorable conditions for the development of aerobic and facultative aerobic microorganisms that form the activated sludge biocenosis for degradation of organic matter in the biological wastewater treatment (Guseva, 1988).

Thus, in order to prevent the development of lactic fermentation and reducing pH in dairy processing wastewater, it is necessary to minimize the duration of wastewater entering to aerobic biological treatment facilities. It should also abandon the use of separately positioned equalizators, and at low concentrations of suspended solids - from the primary wastewater settling. However, it must be considered that in the absence of primary settling and increasing concentrations of suspended solids in the treated wastewater may deteriorate own their biological treatment. Therefore, pre-treatment of dairy processing wastewater may be appropriate application of flotation, which is not cause souring of wastewater due to their saturation by air oxygen.

For biological treatment of dairy processing wastewater appropriate to apply complete mix aeration tanks which for a long time aeration will also serve as equalizators.

For removal of large solid particles from dairy processing wastewater, it was used screens, for sand and other large mineral impurities removal – sand traps, for colloidal and dissolved organic impurities – aeration tanks with jet aeration (Kovalchuk, 2016). In the case of discharge into open waters as a basis can be taken technological schema designed to discharge wastewater into urban sewerage. However, it should be applied two-stage complete biological treatment and deep post-treatment wastewater on filters with floating polystyrene loading. With increased content of suspended solids in wastewater, pressure flotation can be included in the process flow diagram.

The purpose of these studies was an experimental verification of the operation of the proposed facilities and technological schemes for dairy industry wastewater treatment under current production conditions.

The research of work efficiency of the proposed facilities and technological schemes was carried out on wastewater treatment plant in the Shostka city dairy plant and Zolotonosha butter factory, that were built on assumption of developed technology presented in this paper.

The wastewater treatment facilities of the Shostka city dairy plant (Fig. 3) include a raw wastewater pump station, a grate and a horizontal sand trap (Fig. 4) located in the pumping station, and two aeration tanks-settlers with jet aeration and regeneration of activated sludge (Kovalchuk, 2016). The treated wastewater is discharged into the city sewer system.

Fig. 3. Wastewater treatment plant of Shostka city dairy plant

Fig. 4. A grate and a horizontal sand trap in Shostka city dairy plant

3. Results and Discussion

The wastewater treatment facilities of the Shostka city dairy plant, which have been in operation since 2003, are an example of the successful application of the technological method, combining the processes of equalization and biological treatment of wastewater. A two-year observation of the aeration tank-settlers operation showed that during the first year the pH of the treated wastewater, measured at the pumping station, fluctuated within fairly wide limits, from 2.24 to 12.46. At the outlet from the first aeration tank-settler (fig. 5) it was already in the range 6.8–8.06 (average 7.47), and at the outlet from the second aeration tank-settler – 6.94–7.94 (average 7.50). The next year, the pH of the treated wastewater changed in a slightly narrower range, from 4.80 to 11.55. At the outlet from the first aeration tank, the pH was 7.30–8.06 (average 7.77), and at the outlet from the second aeration tank-settler 7.35–7.61 (average 7.47).

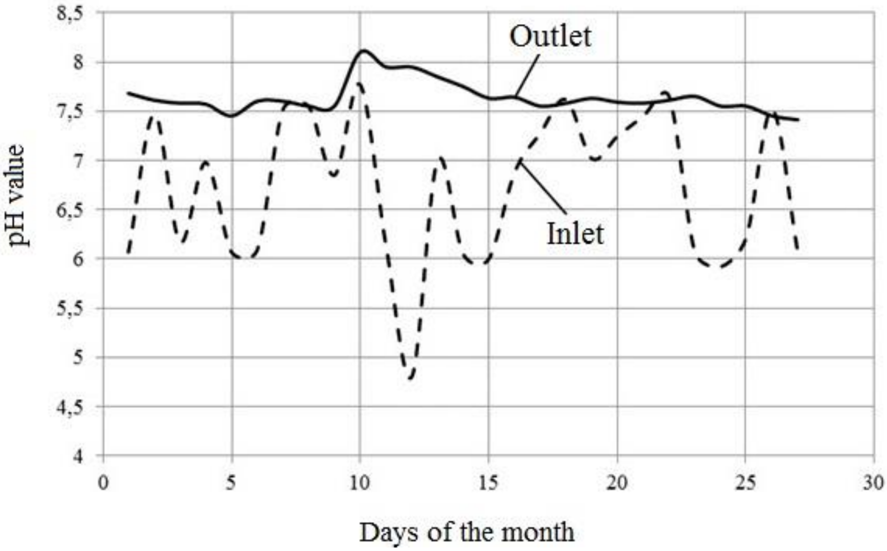

Fig.5. The pH values at the inlet and outlet of the first aeration tank

As can be seen from Figure 3, the Shostka city dairy plant aeration tank-settlers are made of metal and are therefore subject to the influence of ambient air temperature, especially in winter. Capture and dispersion of cold air in the sludge mixture in winter undoubtedly leads to a decrease in its temperature, which leads to a decrease in the rate of oxidation of organic contaminants. On the other hand, aerobic processes occurring in the aeration zone during the biological treatment of highly concentrated dairy processing wastewater lead to the release of heat and, as a consequence, to an increase in the temperature of the sludge mixture.

The determination of the temperature of wastewater during the year of operation of treatment plants showed the following. The average temperature of the raw wastewater entering the pumping station of the Shostka city dairy plant was 25.4°C (boundaries of changing 20.5–37 °C). The average temperature of the sludge mixture in one of the aeration tank-settler was 26.8 °C (20–32 °C), and in the regenerator of the same aeration tank - 26.6 °C (20–32.5 °C). The average temperature of the sludge mixture in the second aeration tank-settler was 25.2 °C (18–32.5 °C), and in the regenerator of the same aeration tank - 25.1 °C (18–32.5 °C). In this connection, it can be concluded that when using metallic aeration tanks with surface aeration does not significantly reduce the temperature of the sludge mixture and they can be used without hindrance in the winter.

Despite fluctuations in the concentrations of pollutants, temperature and pH of the treated wastewater, the operation of the Shostka city dairy plant aeration tanks were stable and ensured the achievement of the necessary contaminant concentrations of in the treated wastewater (Tab. 5).

Table 5. Results of Shostka city dairy plant wastewater treatment

Parameter	**The values of wastewater pollutants concentrations**		**Purification efficiency [%]**
	before biological treatment	**after biological treatment**	
TSS [mg/dm^3]	248–867 493	8.6–367 234	52.5
COD [mgO_2/dm^3]	910–6664 4116	45–739 231	94.4
BOD_{20} [mgO_2/dm^3]	760–4508 3547	12.5–613 58	98.4
BOD_5 [mgO_2/dm^3]	590–3925 3335	5.8–581 44	98.6
Ammonia nitrogen [mg/dm^3]	5.8–8.8 7.2	0–0.87 0.48	93.3
Nitrite (N) [mg/dm^3]	-	-	-
Nitrate (N) [mg/dm^3]	0–37	-	-
Phosphate [mg/dm^3]	49–295 210	0–61.3 21.9	89.6
Fats [mg/dm^3]	66	13	80.3

Dependence of work efficiency on the F/M ratio (Fig. 6) shows that in the range from 80 to 1450 mg COD/d per g MLVSS the efficiency of biological treatment by COD is in the range of 90–98%.

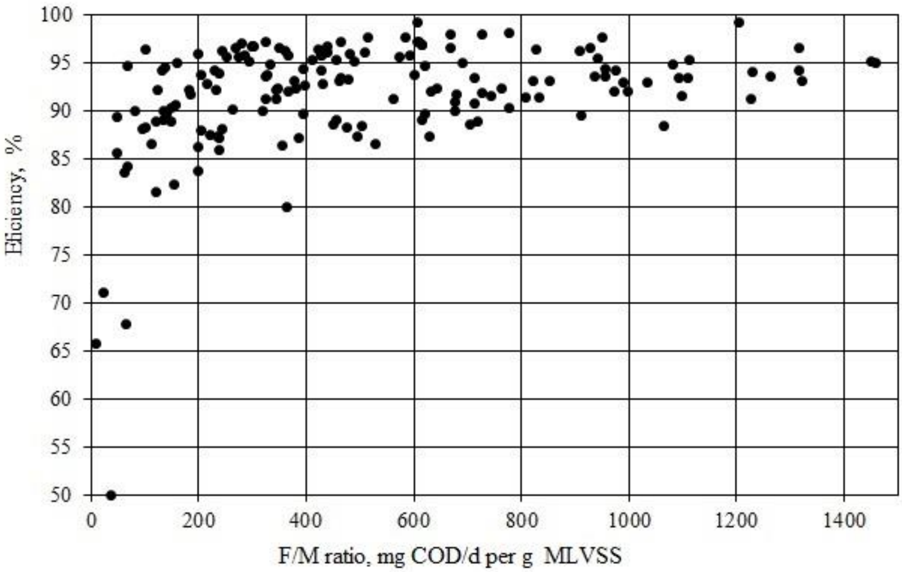

Fig. 6. Effect of F/M ratio on COD removal efficiency

During the observations, the active sludge concentration in the aeration zone was within the range of 2.85–5.7 g/dm^3, and in the regenerator it was 2.6–7.4 g/dm^3. The average value of the sludge index was somewhat elevated and amounted to 150–243 cm^3/g, which, however, with elevated height of the sludge zone, did not lead to an excess of permissible concentrations of suspended solids when discharging treated sewage into the city sewage system. The oxidizing power of aeration tanks for BOD_{20} was in the range of 407–2276 g/(m^3 d). The activated sludge yield in the process of biological wastewater treatment is 0.88–1.63 g per 1 g of the BOD removed.

In processing the results of biological treatment of the Shostka city dairy plant wastewaters was established that they are accurately described by the Eckenfelder equation (Buraczewski, 1981). It was found that the value of the constant *K* depends on the efficiency of the biological treatment *E*, and at values greater than 50–60% can be determined from the empirical equation (Fig. 7)

$$K = 0.126(1 - E)^{-0.89}, dm^3/(g.hr). \qquad (1)$$

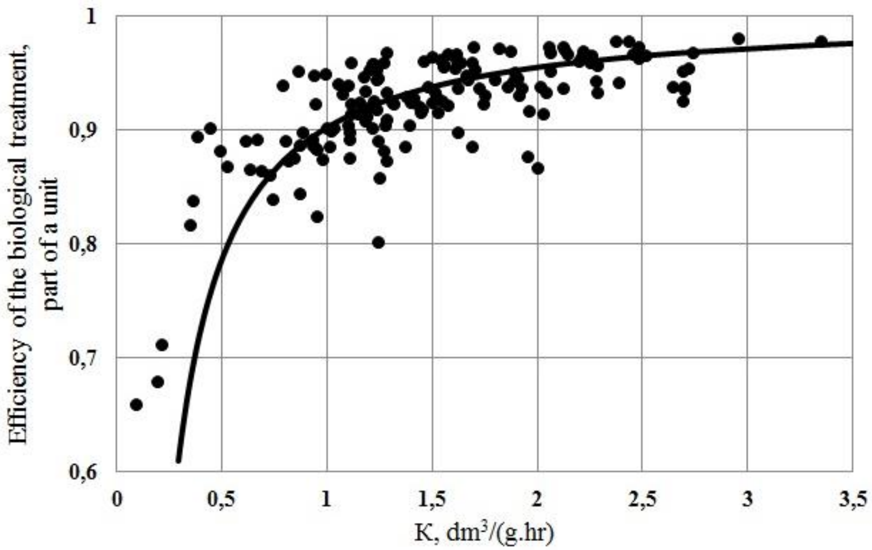

Fig.7. Effect of COD removal efficiency on constant *K*

When using aeration tanks with large hydraulic height and surface jet aeration, should be ensured effective mixing of the sludge mixture to prevent the settling of activated sludge to the bottom. As a result of theoretical investigations on the basis of energy dissipation it was found that when the specific jet aeration capacity greater than 8 W/m^3 sedimentation of active sludge is impossible (Kovalchuk, 2009). Experimental verification of this conclusion was performed in Shostka city dairy plant by measuring the flow rate in aeration tanks. Measured averaged longitudinal component of the flow velocity using non-contact current meters connected with milliamperemeter – at a distance of 0.5 m from the outer constructions wall and at the middle radius aeration zone. As a result of the measurement, a decrease in the flow velocity over the height of the aeration tank from 20 to 120 sm/s (Fig. 8) was established, which is sufficient to maintain the active sludge in the suspended state.

Due to water consumption decrease in dairies industry, the concentration of pollutants, including suspended solids, has increased significantly. Figure 9 shows the change in suspended solids and COD concentrations in the butter factory wastewaters, which was analyzed every hour starting at 8.00. As can be seen from the figure, the suspended solids and COD concentration change very significantly by the hours of the day, with their average daily values being, respectively, 2786 and 1522 mg/dm^3. It is obvious that the wastewater inflowing to the aeration tank with such concentrations of contaminants will negatively affect the biological treatment process, and will cause an increased in activated sludge yields.

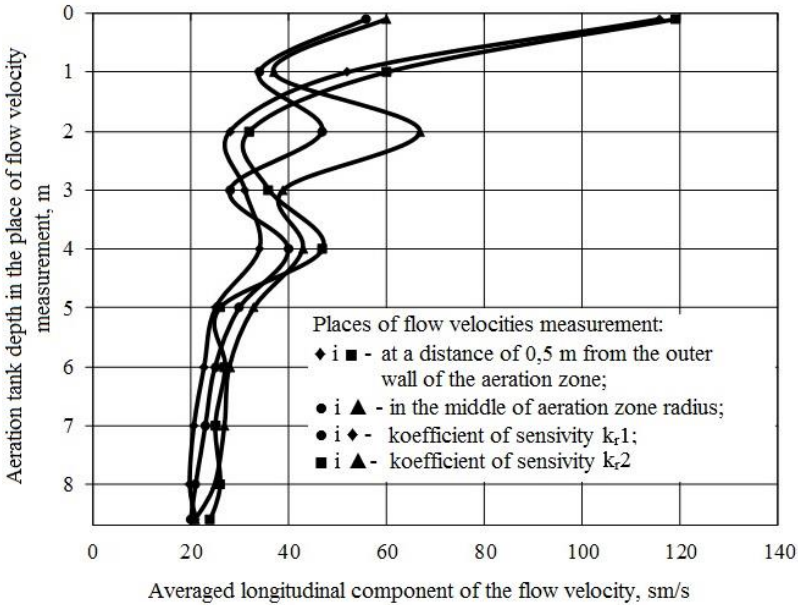

Fig.8. Dependence of the horizontal component of the average flow velocity on the depth of the aeration tank

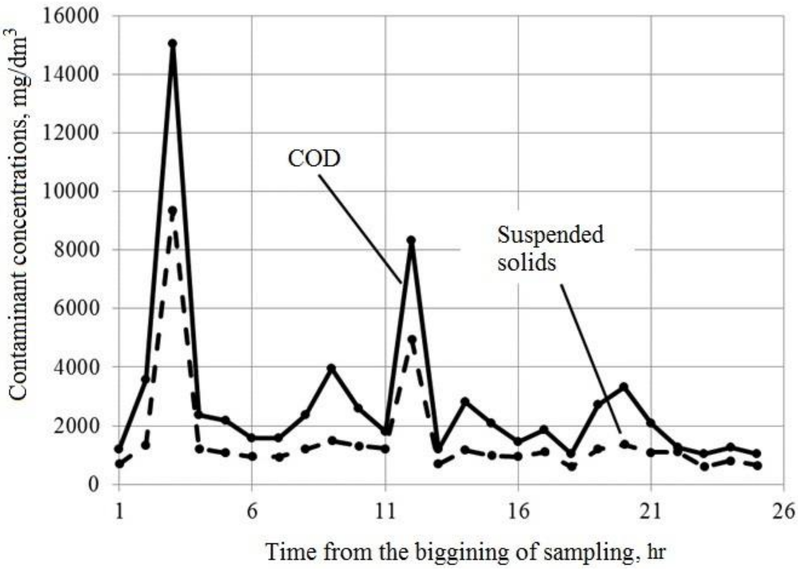

Fig. 9. Change in the wastewater pollution concentration by the hour of the day

For the reducing of suspended solids concentrations in wastewater prior to their supply to the aeration tanks, it is advisable to use flotation. The using of reagent flotation with aluminum sulphate are show that it reducing the suspended solids concentrations by 94.6–96.3%, fats by 91.5–94.5%, phosphates by 63.1–64. 5%. At the same time, the COD of wastewater is reduced by 36.1–40.5%, and BOD_5 - by 34.2–39.2% (Shevchenko, 2015).

We carried out studies of the reagentless wastewater flotation treatment of the Zolotonosha butter factory in a settler-flotator with a diameter of 7.2 m (Fig. 10). As shown by the obtained results, the efficiency of wastewater flotation treatment was 10.9–67.8% for COD (an average of 30.8%). The average concentrations of suspended solids in treated wastewater were 213 mg/dm^3. The use of reagent in flotation significantly complicates the technological scheme of wastewater treatment, requires a reagent preparing device and deteriorate the quality of the primary and flotation sludge. It can be concluded that it is advisable to use reagentless wastewater flotation treatment of dairy processing enterprises.

Fig.10. Settler-flotator diametr 7,2 m

At present, the construction of aeration tank-settler with a surface jet aeration (Fig. 11) has been completed on Zolotonosha butter factory for the biological wastewater treatment.

Fig. 11. Aerotank-settler with surface jet aeration

The commissioning of an aerotank-settler will allow to discharging the excess active sludge before the settler-flotator. This will increase the overall treatment efficiency due to the flotation biocoagulation process in the settler-flotator – the biosorption of contaminants by excessive activated sludge, followed by its flotation separation from wastewater. The use of this technological method will also allow the flotation sludge thickening, which will significantly reduce the total amount of sludge.

4. Summary and Conclusions

- Wastewater from the dairy industry is characterized by a high content of organic contaminants and phosphates, in most cases has a weakly acid reaction. Due to the reduction in the use of water in the dairy processing wastewater concentration, including suspended solids, tend to increase.
- Concentrations of dairy wastewater contamination significantly fluctuate during the day, and wastewater quickly turns sour. To avoid this, wastewater is recommended to be sent directly to the aeration tank-mixers, which ensures their simultaneous biological oxidation, equalization and neutralization. At high

concentrations of suspended solids, wastewater must be pre-treatment by the pressure flotation.

- As it was investigated that under production conditions, the efficiency of butter factory wastewater reagentless flotation treatment by COD was 10.9–67.8% (in the middle – 30.8%), and the residual concentration of suspended solids did not exceed 210 mg/dm^3.
- When the aeration tank-settlers with high hydraulic height and surface jet aeration is using, the sludge mixer moves at a speed of 1.2 to 0.2 m/s and maintains the activated sludge in a suspended state. On the F/M ratio from 80 to 1450 mg COD/d per g MLVSS the efficiency of the city dairy plant biological treatment by COD is in the range of 90–98%.

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The assessment of the degree of water sources diversification as a part of the water system safety audit

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Abstract

In this paper, the exemplary case study of the assessment of the degree of water sources diversification is presented. The analyzed water supply system is composed of 7 pump stations, with individual water intakes, treatment and pumping subsystems. The diversification evaluation was pursued by different coefficients: number of water sources N, water demand reserve Q_{res}, Pielou indicator P, modified (three-parametric) Pielou indicator P_{mod} and reliability index value K of a water intake system.

The obtained results suggest relatively high degree of diversification and therefore high safety of water delivery to water distribution systems. However, the pursued analysis revealed several doubts in application of a Pielou indicator – further research on this coefficient is highly recommended.

Keywords

water supply system, water sources, diversification

1. Introduction

Water supply systems are important elements of the critical infrastructure, which proper operation determines both comfort and safety of water consumers. Therefore, the World Health Organization WHO (Guidelines…, 2011) order is to evaluate security plans for water supply systems. However, the definition of these plans firstly requires implementation of a safety audit of the water supply system (Szpak and Tchórzewska-Cieślak, 2015). The key element of the audit is the analysis of a subsystems security – including water intake, treatment and pumping (Brown and Ward, 2013). One of the basic methods of increasing the safety of water supply delivery is diversification of water sources (Seghezzo et al., 2013). Many different law regulations are implemented in order to ease the diversification process (Drusiani et al., 2013). The analysis of a subsystem's security is usually divided into two steps: 1) the graphical analysis of a technological scheme of a sub-system, 2) the analysis of a functioning reliability of several technological devices (Völker et al., 2012). Despite the fact, that the evaluation of a security sub-system's security is relatively easy for a single pumps station, the complexity of an analysis is significantly greater with multi pump stations in the analyzed system. That is why, the safety analysis are often supported by numerical indicators, very useful in interpretation of analyzed data. In reference to the critical infrastructure elements, such indicators were used for the first time in energetics (Muller et al., 2008). Later, such indicators were also applied in water supply sector (Ermini et al., 2015). However, it is worth to notice, that the majority of these indicators were connected with energy efficiency, costs-absorption or water age determination. In literature, there are no indicators of diversification of sources of water pumped into water supply system. The aim of this paper is to present indicators, which were used do assess the degree of water sources diversification of water sources in an exemplary city.

2. Materials and Methods

2.1. Materials

The assessment of the degree of water sources diversification was conducted for a real water supply system delivering water to approx. 360 000 inhabitants. Daily water demand: average 46.5 thous. m^3/d, maximum 60.0 thous m^3/d. Distribution network contain approx. 800 km of water supply pipes. Water system contains 7 zones with individual water supply stations. Each station consists of an individual groundwater intake, treatment and pumping station. The significant differentiation in stations efficiency can be observed (Tab. 1).

Table 1. Maximum and average daily efficiency of particular water stations

Station	Daily efficiency			
	Average		**Maximum**	
	m^3/d	%	m^3/d	%
1	8400	10.45	10800	10.40
2	39600	49.25	51000	49.13
3	9600	11.94	12000	11.56
4	12000	14.92	15600	15.03
5	7800	9.70	10200	9.83
6	2400	2.99	3000	2.89
7	600	0.75	1200	1.16
Total	**80400**	**100.00**	**103800**	**100.00**

The analyzed water supply system consist of N=7 water supply stations. In all water supply stations there are 15 different storage tanks of total water storage volume equal to 35 010 m^3 (Tab. 2). Water is pumped into distribution network through 11 pressured pipes, with diameters: 150÷600 mm. The flowability of these pipes was calculated in reference to Darcy-Weisbach formula:

- Pipe hydraulic resistance A [s^2/m^6] (Equation (1)):

$$A = \alpha \cdot \frac{8}{\pi^2 \cdot d^5 \cdot g} \tag{1}$$

where: λ – friction losses coefficient (0,030 [-]), d – pipe diameter [m], g – standard gravity [m/s^2];

- Total pipe flowability M [m^6/s^2] (Equation (2)):

$$M = \sum_1^m \frac{1}{A_i} \tag{2}$$

where: A_i – pipe hydraulic resistance of i outflow pipe [s^2/m^6], m – number of outflow pipe from II° water pump station.

Table 2. Parameters of storage tanks (volume) and pipes (diameter, unit pipe hydraulic resistance and total pipe flowability)

Station	**Tank storage volume**		**Pipe diameter *d***	**Unit hydraulic resistance *A***	**Total pipe flowability *M***	
	m^3	%	mm	s^2/m^6	m^6/s^2	%
1	3 440	9.8	500	0.079322	25.21371	16.81
			500	0.079322		
2	3 000	8.6	500	0.079322	12.60686	8.40
3	3 550	10.1	600	0.031878	31.36989	20.91
4	19 000	54.3	600	0.31878	44.00738	29.34
5	3 860	11.0	600	0.031878	35.89487	23.93
			400	0.242071		
			250	2.538301		
6	1 000	2.9	250	2.538301	0.787929	0.53
			250	2.538301		
7	1 160	3.3	200	7.746281	0.129094	0.08
Total	**35 010**	**100.0**	**-**	**-**	**15.0009733**	**100.0**

2.2. Methods

The diversification evaluation of water sources was pursued by different coefficients:

- **Number of water sources *N*** – number of water sources in analyzed water supply system. Single source is defined as a water intake integrated with water pumping station suppling network with water.

- **Water demand reserve Q_{res}** – the ratio between summary daily water efficiency of all water sources ($\sum WP$) cooperating with water supply network and overall water efficiency (maximum daily D_{max} and average daily D_{aver}) (Equations (3–4)).

$$Q_{res\ max} = \frac{\sum WP}{D_{max}} \tag{3}$$

$$Q_{res\ aver} = \frac{\sum WP}{D_{aver}} \tag{4}$$

- **Pielou indicator *P*** – originally developed by Evelyn Cristal Pielou, is the indicator for assessment of diversity in different types biological collections (Pielou, 1966). Subsequently, this indicator was applied in other fields, including assessment of the degree of diversification of water supply (Rak and Boryczko, 2017). Pielou indicator is calculated in accordance to Equations (5)–(7).

In assessment of the degree of water sources diversification, the u_i means the ratio between daily efficiency of a selected source and total efficiency of all sources in the analyzed system. Pielou indicator $P = 1$ means equal efficiency of every station. The lower the indicator value, the bigger the difference of water efficiency. In cases, when number of elements $n = 1$, the Pielou indicator is undetermined. The exemplary calculations of Pielou indicator for $n = 2$, 3 and 4 elements are presented in Table 3.

$$P = \frac{d_{SW}}{d_{SW\,max}} \tag{5}$$

$$d_{SW} = -\sum_{i=1}^{n}(u_i \cdot ln\, u_i) \tag{6}$$

$$d_{SWmax} = ln(n) \tag{7}$$

where: d_{SW} – Shannon-Wheaver indicator, u_i – part of i-element in the whole system, possible values: <0,1>, n – number of elements.

Table 3. Exemplary values of Pielou indicator for $n = 2$, 3 and 4 elements (Rak and Boryczko, 2017)

***n* = 2**	u_i	0.5	0.6	0.7	0.8	0.9	0.95	0.99
		0.5	0.4	0.3	0.2	0.1	0.05	0.01
	P	**1.000**	**0.971**	**0.881**	**0.722**	**0.469**	**0.286**	**0.081**
***n* = 3**	u_i	0.333	0.4	0.5	0.6	0.6	0.7	0.8
		0.333	0.3	0.3	0.3	0.2	0.2	0.1
		0.333	0.3	0.2	0.1	0.2	0.1	0.1
	P	**1.000**	**0.991**	**0.937**	**0.817**	**0.865**	**0.730**	**0.582**
***n* = 4**	u_i	0.25	0.3	0.4	0.5	0.6	0.7	0.8
		0.25	0.3	0.3	0.3	0.2	0.1	0.1
		0.25	0.2	0.15	0.1	0.1	0.1	0.05
		0.25	0.2	0.15	0.1	0.1	0.1	0.05
	P	**1.000**	**0.985**	**0.935**	**0.843**	**0.785**	**0.678**	**0.511**

- **Modified (three-parametric) Pielou indicator P_{mod}** – in some cases, the application of Pielou indicator to assess the diversity of water sources, may lead to ambiguous, difficult to interpret results. That happens in cases of big differences in water efficiency between particular water stations. In order to avoid that problem, Rak and Boryczko (2017) proposed the modification of that indicator, in which three additional parameters should be taken into account: water intakes efficiency Q, volume of water in water network tanks V and flowability of pressured pipelines of a II° water pump station M. The value of modified Pielou indicator P_{mod} is a sum of indicators calculated in accordance to Equations (8)–(11).

$$P_{mod} = P_Q + P_V + P_M \tag{8}$$

$$P_Q = \frac{-\sum_{i=1}^{n}(u_i \cdot ln\, u_i)}{ln(n)} \tag{9}$$

$$P_V = \frac{-\sum_{j=1}^{m}(u_j \cdot ln\, u_j)}{ln(m)} \tag{10}$$

$$P_M = \frac{-\sum_{k=1}^{z}(u_k \cdot ln\, u_k)}{ln(z)} \tag{11}$$

where: Q – water station efficiency, n – number of stations, V – volume of all storage tanks, m – number of storage tanks, M – flowability of pressured pipelines, z – number of pressured pipelines.

Additionally, Rak and Boryczko (2017) proposed the classification of degree of diversification assessed by modified Pielou indicator, presented in table 4.

Table 4. The classification of a degree of diversification

Degree of diversification	$\boldsymbol{P_{mod}}$ **value**
None	0
Low	0.1 ÷ 1.5
Average	1.6 ÷ 2.0
Acceptable	2.1 ÷ 2.5
Very acceptable	2.5 ÷ 3.0

- **Reliability index value of a water intake system *K*** – presented indicators do not fully describe the operational reliability of water supply stations. The calculated values define only the certainty of water delivery to customers. It hinders the interpretation of obtained results. Taking that into account, we propose the additional application of reliability index value K (Equation (12)), commonly used during the assessing water supply object and systems reliability.

$$K_g = \frac{T_p}{T_p + T_o} \tag{12}$$

where: T_p – average time between breakages, T_o – average repair time.

We recommend to divide calculations into two steps. First, the values of reliability indexes of particular elements of water supply station should be calculated. Next, the reliability index value of all stations should be evaluated. In the second step, the method of complete decomposition (Kwietniewski et al., 1993) is used, in which all possible combinations of particular stations operations are considered.

3. Results and Discussion

Calculated values of water demand reserve Q_{res} indicator are presented in table 5. Basing on obtained results, it can be said that analyzed water supply system has a great water reserve, both in reference to average and maximum daily water demand. The results of calculated Pielou indicator P and modified Pielou indicator P_{mod} are presented in table 6.

Table 5. Water supply stations reserve

Production reserve Q_{res}		
Average	1.729	2.232
Maximum	1.340	1.730

Table 6. Results of calculated Pielou P indicator and modified Pielou indicator P_{mod}

	Average capacity		**Maximum Capacity**		**Storage tank volume**		**Flowability**	
	u_i	$u_i \cdot \ln(u_i)$	u_i	$u_i \cdot \ln(u_i)$	u_i	$u_i \cdot \ln(u_i)$	u_i	$u_i \cdot \ln(u_i)$
	0.104	0.235992	0.104	0.235448	0.098	0.227633	0.168	0.299677
	0.493	0.348808	0.491	0.349159	0.086	0.210993	0.084	0.208063
	0.119	0.253761	0.116	0.249429	0.101	0.231556	0.209	0.327173
	0.149	0.283897	0.150	0.284827	0.543	0.331581	0.293	0.359682
	0.097	0.226325	0.098	0.227985	0.110	0.2428	0.239	0.342079
	0.030	0.104822	0.029	0.102424	0.029	0.102673	0.005	0.026492
	0.007	0.036551	0.012	0.051562	0.033	0.112571	0.001	0.006908
Total	**1.000**	**1.490156**	**1.000**	**1.500833**	**1.000**	**1.459808**	**1.000**	**1.570073**
	$\boldsymbol{P_Q = 0.766}$		$\boldsymbol{P_Q = 0.771}$		$\boldsymbol{P_V = 0.750}$		$\boldsymbol{P_M = 0.807}$	

The calculated Pielou indicator was equal to water station efficiency P_Q indicator $P = 0.766$ for average daily efficiency and $P = 0.771$ for maximum daily efficiency. It means, that in the analyzed system there is a diversification of water sources, but it cannot be classified as high or low. The modified, three-parametric Pielou indicator, which includes water station efficiency, storage tanks volume and flowability of outflow pipes of particular water stations was equal to $P_{mod} = 2.323$ for average daily efficiency and $P_{mod} = 2.328$ for maximum daily efficiency of particular stations. In accordance to (Rak and Boryczko, 2017), that values responds with acceptable level of diversification degree.

Significantly more laborious was the calculation of reliability index value K for the whole water supply system. The most problematic was the collection of operational data from last 10 years, archived only in analogous form (over 200 exploitation tomes). More precise description of K index estimation for singular water pump station is presented in (Suchorab et al., 2017). The calculated values of reliability indexes *K* of all water stations are presented in table 8. The calculated value of reliability index *K* for all water stations were compared with references values from reliability classification evaluated by Kwietniewski et al. (1993) (Tab. 7). The calculated value of index *K* for all water stations represents the water supply system of the highest reliability category.

Table 7. Water supply stations reserve

Category of water supply system	***K* index value**
1. Big water supply systems, over 50 000 inhabitants	0.9917809
2. Average water supply systems, 5 000–50 000 inhabitamts	0.9835617
3. Small water supply systems, below 5 000 inhabitants	0.9671233

Table 8. Calculated reliability index *K*

Station	1	2	3	4	5	6	7	**Total**
K	0.87120	0.99232	0.98467	0.99790	0.98844	0.96872	0.99157	**0.99637**

4. Summary and Conclusions

Proper water supply safety audit requires the evaluation of the diversification level of water intakes. Individually, each of presented indicators do not enable to evaluate the diversification level. Currently, there is no appropriate guidelines to include all of the presented indicators, which hinders the objective diversification level evaluation.

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Green infrastructure in urbanized areas as the future of eco-engineering

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Abstract

Stormwater is the natural resource from economic and environmental point of view, i.e. the resource treated as natural resources, forces of nature and environmental values affecting the quality of human life. The constant change in relation to time is a distinctive feature of these resources. The direct possibility of usage by mankind and renewability of surface water and groundwater resources seems to be important. The development of cities, directly triggered by the progress of civilization (simultaneously its driving force) completely distorted the natural water cycle. It is predicted, that until 2050 nearly 70% of Earth’s population (approx. 6 400 000 000) will reside in the cities or urbanized regions. This means the nearly twofold increase in the number of urban residents. Thus, the increase in demand for natural resources, including water, may be expected. In many cities of the world, the stormwater is the only unpaid and easily accessible resource, which may minimize the negative effects of urbanization. The main challenge is to direct the development of the cities to positively affect all aspects of life, including the quality of natural environment. The city is an ecosystem, place for coexistence of people, infrastructure and nature. Until recently, stormwater in the cities was treated as a threat, so it was supposed to be removed as quickly as possible. Nowadays, the shaping of friendly public space, in accordance to the principles of sustainable development may be assured by the green and blue infrastructure. This infrastructure is an effective manner of stormwater management in the urbanized basins and allows to restore the natural processes occurring in the environment: infiltration – by increase in available permeable and retentive surfaces, retention – by increase in available surface open waters, application of surface and underground retentive reservoirs as well as application of bioretention; transpiration and treatment – by application of the properly selected species of plants. The green and blue infrastructure are firmly related one to another: the plants are the biological water reservoir, while water is necessary for the plants to growth. This work contains a precise description of green walls, known as live walls, allowing, besides to green roofs, to establish the green landscape of natural appearance, both, inside and outside the buildings, including their elevations.

Keyword

green infrastructure, green walls, green roofs, stormwater, biological water reservoir

Introduction

The expansion of the cities, which is an inevitable consequence of civilization development has thoroughly disturbed the natural water cycle. It is being forecasted that until 2050 almost 70% of entire population (i.e. around 6,4 billion people) will live in cities and their functional areas, which means that cities' population shall almost double.

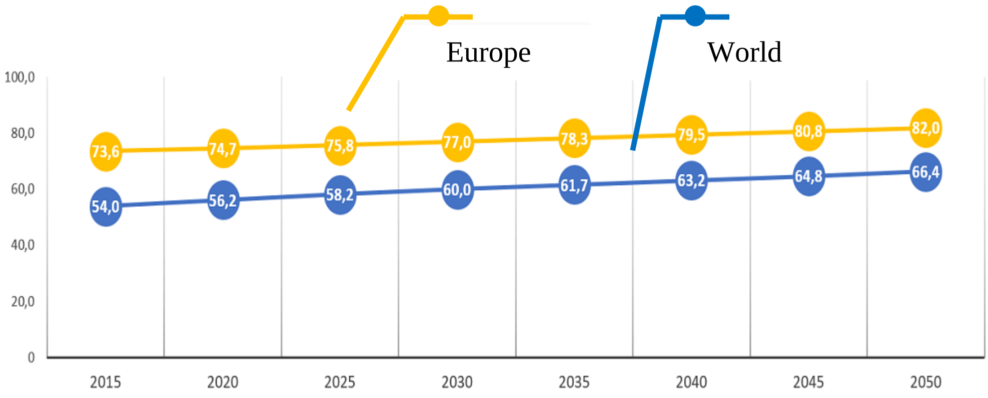

Fig. 1. Europe's and world's population growth until 2050 [http://www.ideologia.pl/smart-city-jak-inteligentne-miasta-poprawiaja-zycie-mieszkancow]

It will cause a huge increase in demand for natural resources, included water.

Municipal areas have always been exposed to natural threats which in turn have been causing social, economic and environmental losses, what is directly related to the climate, environmental conditions, and human actions occurring in a particular area. Currently, such threats in Poland and in the world have been intensified mainly due to global warming which stands in direct relationship with extreme weather conditions as well as natural disasters (Kundziewicz and Kowalczak, 2008; Mańkowska-Wróbel, 2014).

Cities are a unique ecosystem where people, infrastructure and environment should co-exist in harmony. The real challenge is to direct the development of a city in such a way that it has a positive influence on all areas of life as well as on the environment.

Meteoric water is, from economic and environmental point of view, a natural resource, i.e. a resource which can constitute a part of natural resources, powers of nature and an environmental value that influences the quality of human life. Main feature of such resources is their constant change in relation to time. Meteoric water is mainly the source (renewal) of water supplies (both surface and underground) but it can also be directly used by humans. Meteoric water is the basic part of water supplies which renews the surface and underground sources.

In majority of cities meteoric water is the only free and easily accessible water source that can minimize the negative consequences and costs of urbanization.

The way of treating meteoric and melting water in urbanized and industrial areas is therefore the key element to achieve a balanced city growth. They should be protected against pollution, managed and used in the place where they are created. Not long ago, meteoric water was considered a threat which should be immediately drained (Burszta–Adamiak, 2012; Januchta–Szostak, 2011).

A good way of dealing with unfavorable climate changes are solutions based on the powers of nature, blue and green infrastructure included.

Blue and green infrastructure is the key element in the following strategies:

- Protection and reasonable exploitation of natural resources as well as adaptation to climate change (Europe 2020).
- "Balanced, intelligent and favoring social inclusion growth is being prioritized" (Strategy…, 2012).
- Actions in favor of reasonable resources management, environmental conditions enhancement as well as adaptation to climate changes.

The article pays special attention to the so called vertical gardens or green walls which, apart from green roofs and green gardens, allow to create a floral countryside both inside and outside of the building as well as on its elevation.

1. *Characteristics of vertical gardens*

Vertical gardens are one of the most modern trends in landscape architecture. It is being assumed that they are vertical partitions, walls, grates, screens or even fences, covered with plants. Both complex constructions and simple solutions can be qualified under this label (Kania et al., 2013; Kolbek, 2014).

Vertical gardens design includes defining the climate conditions of a particular place, the choice of plants, construction and maintenance method.

The creation of the first green walls, which are becoming more and more popular nowadays not only outside but also inside buildings, was inspired by nature observation. In natural conditions, plants grow perfectly fine on steep rocks or cliffs with none or hardly any soil. They need only carbon dioxide and water to survive.

Living walls have been already known in ancient times. The history of green walls began in Babylonia 6th century BC with the Hanging Gardens of Babylon. They are the first, scientifically documented examples of green roofs (Burszta–Adamiak, 2012; Szajda–Birnfeld et al., 2012).

The greatest promoter of these wonderful solution is Patrick Blanc, a French botanist. One of his greatest achievements is the creation of vertical garden in Madrid, which consists of 18000 plants belonging to 250 species. It is located on 460 square meters, it is 24 meters high, 19 meters wide and 1 meter deep.

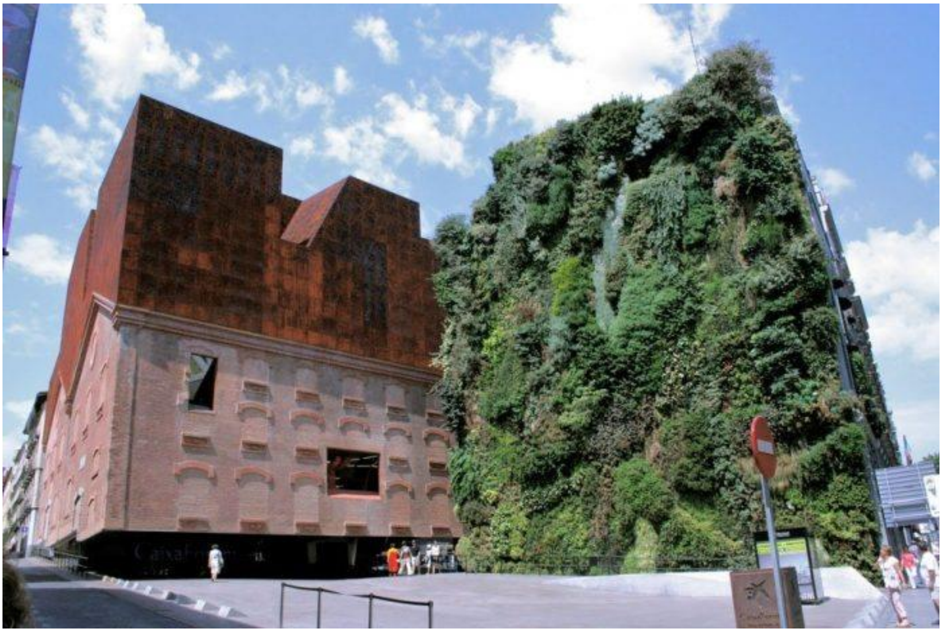

Fig. 2. Green Wall – Center of Art and Culture in Madrid – Caixa Forum (https://redesignreport.com)

Various methods can be used to green vertical surfaces – from cheap and simple solutions to complex and expensive projects requiring state of the art technology.

Most commonly used method is to lead the creepers, which are anchored in soil, on the wall. It takes time to let the wall grow green but it is the most effective an economic.

Green facades, which are the reflection of latest technology, use not only creepers, but also perennials, bushes and even small trees planted in special constructions, creating exceptionally high gardens.

Three types of establishing vertical gardens can be distinguished depending on the way of planting them, i.e. felt, module and container method. The system is chosen on the basis of material from which the building is created, climate conditions or area (Furmaniuk, 2010; Patro and Koper, 2016; Skarżyński, 2012).

The most well-known method is the hydroponic method used by Patrick Blanc. Undisputable advantage of such a construction is its lightness which allows to cover considerable area with plants as well as its free composition and living pictures creation (felt system).

Fig. 3. Green wall covered in creepers

Felt system is basing on hydroponics and uses the properties of mats constructed from plaited synthetic fibers, which can accumulate water. The mats are attached to one another in such a way that they create single pockets, where flora is planted. They way of plait allows the roots to grow between the fibers leaving them constant access to humidity and air.

Fig.4. Felt method used for vertical garden creation.
(https://architekturakrajobrazublog.wordpress.com)

Hydroponic systems which use felt as the only medium for the growth of root system are not appropriate for Polish climate conditions. Such solutions can be found in cities, where coldest average temperatures never go below 0 degrees centigrade and ground frost is non-existent or short lasting. The minus of such a technology is moderately large water demand which can reach between 5 and 7 liters per square meter per day. Such solutions usually use meteoric water, store it and re-use it to water the green wall or adjacent green areas again later (Patro and Koper, 2016).

Fig.5. Vertical garden on the wall of University of Life Scienes in Lublin (http://www.dziennikwschodni.pl)

Module system bases on easily constructed modules with containers where plants are placed like in simple flowerpots. Surface for rooting and providing nutrition can be soil, pearlite, leca, mineral cotton, etc. Frame construction is made of plastic, wood, stainless steel or aluminum (Fig.6).

Third type of green walls are the so called container systems also called hybrids. Contrary to Blanc's walls, the walls in container method are thick and heavy, they need much space, very solid constructions and huge amount of time to achieve the end effect. The system is composed of tightened steel ropes and boxes for soil. This cascade solution is achieved through placing containers with plants on different levels. Special scaffolding must be built in order to finalize such a solution. These types of constructions are much heavier than the hydroponic ones and weigh around 150–200 kg/square meter but it does not cause any technological problems during plants installation.

Correctly designed construction can easily bear such loadings. An advantage of such a system is reduction of water needed for maintenance (around 2 liters per square meter per day). Water loss is minimal and allows the plants to use the water fully.

Each system must be insulated from the wall with a waterproof material such as EPDM membrane or PVC slab. Insulation must be made very carefully and be 100% waterproof. Next important element is to keep a few inches of free ventilation space between the construction and the wall that enables good air circulation which shall protect the surface from fungus and allows thermal insulation to “breathe.” Necessary detail of every vertical garden is the watering system. It is a series of pipes transporting water with nutrition to the roots. The construction of vertical gardens should hide the roots of the plants as well as the watering system. It should also keep the pockets for plants at a given level.

Fig.6. Module method used for vertical garden creation (Furmanik 2010; http://gardenspot.pl)

Fig.7. Container system used for vertical garden creation (Furmanik 2010; http://blog.gcl.com.pl, http://www.gabiony.poznan.pl)

A few rules must be taken into consideration while designing green walls. Contrary to traditional gardening, plants in a vertical garden are not covered with snow or mulch during winter. They are exposed to weather conditions (such as wind) that can damage the leaves. That is why species which re-generate fast and are immune to weather conditions should be chosen. Another important aspect is to avoid water losses in winter season. Even during this season occasional plus temperatures may occur which can encourage the plants to seek water. In case of its loss leaves will die. This is the main reason for leaves exchange during vegetative period. In order to avoid this problem a vertical garden adjusted to climate conditions must be chosen. Hydroponic systems using only felt might not be appropriate for polish climate conditions.

Vertical gardens need nurturing. Appropriate nurturing consists of watering times' verification, cleaning the drainage system and eliminating water losses in the watering system. Depending on climate conditions and chosen species fertilizer needs to be restocked, and preventive sprayings done to eliminate potential sicknesses as well as pest. During autumn-winter period cleaning and maintenance of the watering system is of utmost importance in Polish climate conditions.

2. *Vertical gardens' influence on municipal water management system.*

Vertical gardens are multi-functional entities providing services such as supplying, regulatory or cultural. Living walls are environmentally functioning elements of cities' infrastructure which have positive influence on many aspects of life (Burszta–Adamiak, 2012; Energie Cites, 2013).

Green walls create a healing environment. Green speeds up the regeneration processes, which reduces the average time of hospital visits. They bring soothing, enhance our mood and create a healthier climate in the area where they are created. Illnesses such as eye sore, headaches, sore throat or fatigue are less frequent. In offices, which are full of plants, people take less sickness leaves.

Vertical gardens are not only an interesting decoration, but they also hide old, cracked walls.

They compensate the loss of green areas for investments letting the plants into the city without occupying much space.

As every green area they are "the lungs of the city" enhancing air quality and treating it. Plants on the green wall filter the dust from the air and convert carbon dioxide into oxygen. One square meter of green wall consumes 2,4 kg of carbon dioxide and produces 1,7 kg of oxygen per year.

Green walls are an ideal acoustic isolator that consumes 41% more noise than a traditional façade, reducing the noise by 8 dB, so by half. They create a fireproof layer and prolong the life of the façade.

Green walls provide isolation of the elevations of the buildings which helps in reducing the amount of energy needed to air-condition the rooms. They also help to eliminate the negative consequences of urban heat island.

Thank to vertical gardens biodiversity is increased; plants covering the green wall are a perfect hideout for birds, butterflies and insects.

In water management, both in excess and shortage situation, the green and blue infrastructure (vertical gardens included) is an effective way of managing meteoric water in urbanized areas. Green and blue infrastructure closely co-operate – plants are biological water containers, whereas water is essential for plants' growth and lets the following natural processes be renewed:

- Infiltration – by increasing the permeable and absorptive areas
- Retention – by increasing the share of open waters, using surface and underground retention tanks as well as using appropriate solutions based on bio-retention.
- Evapotranspiration – absorbing some amount of water from the root layer and evaporating it to the air.

Vertical gardens are a solution which slows down the drainage of meteoric water to the sewer system. Its fast and rapid drainage is currently the most important problem of municipal water management.

3. Conclusion

Meteoric water in majority of cities is one of the biggest challenges of water management system. Green and blue infrastructure allows introducing modern technology of meteoric water management system that is in accord with eco-hydrology and balanced growth.

Building retention and infiltration systems through blue and green infrastructure must be related to spatial planning as well as anti-flood protection with help of systems of spatial information.

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Influence of milling on the effects of co-digestion of brewery spent grain and sewage sludge

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Abstract

This study examined the influence of milling on the effects of co-digestion of brewery spent grain and sewage sludge. The experiments were performed in batch-mode to evaluate both the biogas potential and the biogas production rate. Three runs were carried out, one of them concerned the anaerobic digestion of sewage sludge (as control), whereas the others referred to the co-digestion of milled and un-milled brewery spent grain with sewage sludge. The runs were conducted under mesophilic conditions (temperature 37±1°C) and lasted for 21 days. Addition of brewery spent grain increased the biogas potential by 19%, while the biogas production rate raised for above 27% as compared to the control. A similar tendency was found for methane. However, there no influence of milling on the co-digestion effectiveness was observed.

Keyword

sewage sludge, brewery spend grain, milling, anaerobic digestion, biogas

1. Introduction

As far as environmental issues are considered, a widely accepted concept of sustainable development – which implies ecological, social and the economic responsibilities – should be taken into account. A sustainable sewage sludge management is a difficult task, both from the economic and technical point of view. Sewage sludge (SS) is characterized by a high content of organic compounds and this is the cause of its putrescibility (Cecchi et al., 1996). One possible method of stabilization and hygienization involves anaerobic digestion process (Hamzawi et al., 1998). The environmental regulations in the European Union are based on the concept of prevention and control of pollution; therefore, sludge management and the production of biogas from this residue seem to be a promising solution. Biogas is generated as a product of the anaerobic digestion process and usually contains about 40–70% of methane (Montusiewicz et al., 2008). It can be considered as a source of energy for producing heat or electricity. Taking into account the economic and ecological benefits related to mono-substrate anaerobic degradation, co-digestion (defined as fermentation of at least two types of different organic substrates) seems to be especially attractive. Due to the large amount of sewage sludge produced in municipal wastewater treatment plants and a high number of operating digesters, the co-fermentation of SS with other types of organic waste can be very appealing (You et al., 2007). Brewing residues include brewery wastewater, surplus yeast and brewery spent grain (BSG); the latter one represents about 85% of the total by-product of the brewing process with an annual global production of 30×10^6 tonnes. The BSG comprises a lignocellulosic residue obtained from the mashing and filtration of the malting grains to produce maltose from starch for fermentation (Zupancic et al., 2016) and this is primarily used for the preparation of animal feeds because of its high fibre (60%) and protein (20%) contents (Mussatto et al., 2006, Fillaudeau et al., 2006). The BSG remains largely un-tapped resource for anaerobic digestion. Nevertheless, Sezun et al. (2011) have shown that anaerobic digestion of BSG as mono substrate cannot be successful even using chemical pretreatment; they have determined a p-cresol inhibition. The BSG have also a high protein content (C/N ratio of 3–5 and total nitrogen (TN) of 11–13 g/kg of wet weight), which may lead to ammonia inhibition. Sung and Liu (2003) reported inhibition limit already at 4.92 g/kg of TN. Therefore, dilution of the substrate or using additional substrates is necessary to perform co-digestion, where BSG is mixed with a more carbon-rich substrate. Interestingly, some studies have recently been conducted involving BSG as a co-substrate. Malakhova et al. (2015) co-digested Jerusalem artichoke phytomass with BSG (5:1 total solids (TS) ratio of BSG vs. Jerusalem artichoke) and achieved specific methane production of 108 L/kg of wet weight BSG added in a batch experiment. Tewelde et al. (2012) used cattle dung as a co-substrate with BSG, achieving a specific biogas production of 410 L/kg VS of added mixture in a continuous experiment with a hydraulic retention time of 40 days.

In order to enhance the hydrolysis rate and omit the limitations of the bioconversion of BSG lignocellulosic structure, several pretreatments such as mechanical, chemical and biological have been proposed (Bochman et al., 2015; Severini et al., 2015; Wilkinson et al., 2014). These are applied for decreasing the size of particles and breaking down cell wall structures. Among them, milling also requires investigation in terms of improving the BSG digestibility.

The study examined the influence of milling on the effects of co-digestion of brewery spent grain and sewage sludge. The experiments were performed in batch-mode to evaluate both the biogas potential and the biogas production rate.

2. Materials and Methods

2.1. Substrate characteristics

Mixtures of sewage sludge and dried brewery spent grain (both non-milled and milled) were used for the co-digestion study. Sewage sludge was used for the anaerobic digestion as a control.

Sewage sludge was obtained from the Lublin municipal wastewater treatment plant (WWTP), Poland. This included two-source residues from primary and secondary clarifiers, both of them thickened. Under laboratory conditions, sludge was mixed at the recommended volume ratio of 60:40 (primary:waste sludge), then homogenized, manually screened through a 3-mm screen and partitioned. The sludge prepared in this manner was fed to the reactor as mixed sewage sludge (SS), with main characteristics presented in Table 1.

Brewery spent grain (BSG) was sourced from a small brewery in Lublin that uses barley as a raw material for beer production. The BSG sample of 2 kg weight was transported to the laboratory, then dried using oven at a temperature of 50°C to omit its rapid degradation and sub-divided. One of the sub-samples was milled using a stainless steel blender to reduce the BSG particle size to 1 mm (MBSG), the other was retained without milling. The BSG composition is given in Table 1.

An inoculum for the laboratory reactors was sampled from Lublin WWTP as a collected digest from a mesophilic anaerobic digester operating at a hydraulic retention time of about 30 days.

Table 1. Characteristics of co-digestion components during experiment (average value ± standard deviation)

Parameter	Unit	Sewage Sludge	Brewery Spent Grain
TS	g kg^{-1}	48.72 ± 0.75	223.9 ± 4.3
VS	g kg^{-1}	36.94 ± 0.72	217.2 ± 4.2
pH	-	6.39 ± 0.03	-

At the beginning of the experiment, each reactor was inoculated with digest from Lublin WWTP. Before starting the anaerobic digestion, the nitrogen gas was purged for 2 min in each reactor to remove the gases present inside and ensure anaerobic conditions. After 30 days of adaptation ensuring the inoculum to be deeply treated with residual daily biogas production of 0.01 Ndm^3 d^{-1}, reactors were fed with the substrates and washed out once again using nitrogen gas.

2.2. Experimental set-up and procedure

The specific biogas (BP) and methane potential (BMP) of inoculum, sewage sludge and co-digestion mixtures were studied in batch assays using the automatic biogas/methane potential test system BioReactor Simulator (Bioprocess Control AB, Sweden). The BRS consisted of two units: BRS-A and BRS-B (Fig. 1).

Fig. 1. BioReactor Simulator (BRS): anaerobic reactors (BRS-A, unit in the right), gas volume measuring device (BRS-B, unit in the left)

The BRS-A unit comprised of 6 reactors with the volume of 2 L. The media in each reactor were mixed by means of a slowly rotating agitator. Biogas was continuously produced and registered by the system. The volume of gas released was measured with unit BRS-B using a wet gas flow-measuring device with a multi-flow cell arrangement, which works according to the principle of liquid displacement and can monitor low gas flows (a digital pulse is generated when a defined volume of gas flows through the device). An integrated embedded data acquisition system was used to record and display the results.

After the adaptation of inoculum which lasted 30 days, three runs were carried out. One of them concerned SS anaerobic digestion (as control), the others referred to the co-digestion of BSG and MBSG with SS. The BP tests were conducted in duplicates, under mesophilic conditions (temperature 37±1°C). Each of these lasted 21 days. In the first run the reactors R1 and R2 were fed with 1.4 L of inoculum and 0.4 L of SS. The second run was conducted in the reactors R3 and R4 following the same schedule. However, the reactors were fed using mixture of SS and BSG (the influent consisted of a mixture of 1.4 L inoculum, 0.4 L SS and 5g BSG). The third run arrangement was the same as the previous one, but this time the influent consisted of a mixture of 1.4 L inoculum, 0.4 L SS and 5 g MBSG. The biogas produced by the inoculum was subtracted from the results obtained from the test samples. The biogas (BP_{21}) and methane (BMP_{21}) potential as well as the biogas (GPR) and methane production rate (MPR) were determined.

2.3. Analytical methods

In the samples of SS and BSG, total solids (TS), volatile solids (VS) and pH were analyzed in triplicates. Total solids were determined as the residue after water evaporation (24 h drying at 105°C to constant weight), volatile solids were measured by ignition at 500°C. TS and VS were determined according to Polish Standards Methods PN-75/C-04616/01, pH were analyzed according to PN-EN 12176. Biogas volume was measured on-line using BRS-B unit. The composition of the biogas was measured using a Shimadzu GC 14B gas chromatograph coupled with a thermal conductivity detector (TCD) fitted with glass packed columns. The Porapak Q column was used to determine CH_4 and CO_2 concentrations. The parameters used for the analysis were as follows – injector 40°C, column oven 40°C, detector 60°C, and current bridge 150 mA. The carrier gas was helium with a flux rate of 40 cm^3min^{-1}. Peak areas were determined by the computer integration program (CHROMA X).

3. Results and Discussion

The results of the study concerning the biogas/methane potential and the biogas/methane production rate are given in Table 2. The cumulative biogas production obtained for each run is presented in Fig.2. The data on the graph represent the average values of daily biogas production per g of VS fed the reactors for specified runs.

Table 2. The biogas/methane potential and biogas/methane production rate (average value ± standard deviation)

Parameter	**Unit**	**SS Run 1**	**SS + BSG Run 2**	**SS + MBSG Run 3**
BP_{21}	$Ndm^3g^{-1}VS$	0.405 ± 0.019	0.482 ± 0.036	0.480 ± 0.014
GPR	$Ndm^3dm^{-3}d^{-1}$	0.713 ± 0.022	0.910 ± 0.048	0.907 ± 0.032
BMP_{21}	$Ndm^3g^{-1}VS$	0.25 ± 0.011	0.30 ± 0.024	0.30 ± 0.010
MPR	$Ndm^3dm^{-3}d^{-1}$	0.44 ± 0.013	0.57 ± 0.023	0.57 ± 0.010

Concerning addition of BSG and MBSG, all the co-digestion parameters increased as compared to the control SS. The biogas potential was enhanced by 19%, while the biogas production rate raised over 27% using co-substrate. Such BP values were comparable to those noticed by Tewelde et al. (2012) in the anaerobic co-digestion systems supplied with cattle dung and BSG. The methane concentration was a little higher in the presence of brewery spent grain and reached the level of 62.38 ± 0.13 and 62.83 ± 0.14 for run 2 and run 3, respectively. In the case of SS, it was 62.07 ± 0.33. Thus, the biomethane potential as well as the biomethane production rate was found higher by 20% and 30% for BMP_{21} and MPR, respectively. The BMP in co-digested runs amounted 300 dm^3 kg^{-1} VS and exceeded the value of 108 dm^3 kg^{-1} of wet weight BSG (added as a co-substrate to anaerobic digestion of Jerusalem artichoke) suggested by Malakhova et al. (2015) as the maximum biomethane potential. The difference of the values could result both from using the other co-substrate and involving dry BSG instead of the wet one.

Interestingly, there was no observed influence of milling on the co-digestion effectiveness since the parameters specified revealed comparable values. Thus, adding both BSG and MBSG in dose of 5g did not alter the co-digestion results. This may be caused by insufficient grinding of BSG. According to Kaparaju et al. (2002), the biogas production efficiency is closely related to the reduction in particle size of lignocellulosic matter. Reduction of the sugar cane marc particles and coconut fibers size from 5 mm to less than 0.85 mm resulted in increased methane efficiency by 30% (Kivaisi and Eliapenda, 1994).

The cumulative biogas production was differentiated for the co-digestion and control runs. In the case of SS, the biogas was generated with first-order kinetics $V = V_{max}(1-\exp(-k\cdot t))$; where V – the biogas volume produced in time (t), k – the reaction rate constant and V_{max} – the maximum gas volume. However, the biogas production for co-digestion revealed the presence of two phases with different kinetics. The second one started after 12 days of the experiment and could be accomplished with degradation of lignocellulosic matter (specific BSG component). Such a complex structure required much longer time to be converted to the monosaccharides, easily degradable by microorganisms. In order to describe the cumulative biogas production phases in the co-fermentation process, detailed kinetics studies should be performed.

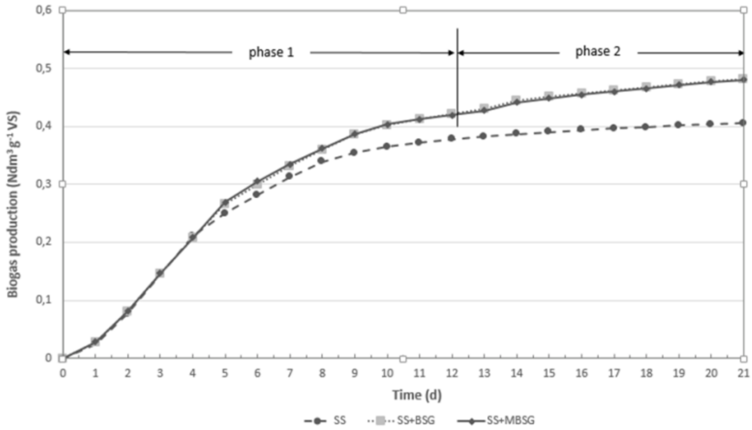

Fig. 2. Cumulative biogas production for specified runs

4. Summary and Conclusions

The results indicate that measured additions of BSG/MBSG as a co-substrate for the anaerobic digestion of sewage sludge enhanced both the biogas/methane potential and the production rate for above 19%. Two phases of biogas production occurred conducting co-digestion of BSG/MBSG and SS. The second phase is most likely accomplished with degradation of complex lignocellulosic compounds. No influence of BSG milling on the co-digestion effectiveness was observed, probably because of an insufficient grinding level.

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Use of the cumulative curve to calculate the efficiency of the roof emergency drainage system

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Abstract

Short-lived torrential rains are an increasingly common phenomenon. The amount of water accumulated in a short time on the roof of an object can lead to overload of the roof structure. This problem is particularly related to the large area of the warehouses, where drainage systems, under average conditions of use, use the Pluvia vacuum rainwater drainage system.

The article presents a calculation scheme using the cumulative roof volume curve set together with the cumulative volume of emergency spillways. The calculation assumes a 1 cm increase in the volume of water accumulated on the roof and combined with a 1 cm increase in drainage through the spillways system. The computational scheme was made for the actual roof of the warehouse, which was described in geodetic coordinates. Different elevations of spillway crest and lack of regularity in the ordinate roof surfaces (flat roof with thermo and hydro insulation on trapeze steel sheet) were included in the calculations. The analyzed example also provides a way to deal with the calculation of the emergency spillway, whose crest has an unusual shape in the plan and with the choice of the efficiency factor for atypical spillway, in the absence of physical measurements, confirming the validity of the assumed assumptions.

Key words

Pluvia, torrential rain, cumulative curve, emergency spillway

1. Introduction

Rainfall is less and less predictable in recent years. It is becoming increasingly difficult to drain the excess rainwater. The construction of a drainage system prepared for the discharge of excess rainwater in the case of torrential rains is practically impossible. Ensuring the discharge of excess rain water from roofs of warehouses is particularly important. Warehouse roofs are often drained using the Pluvia system (Joniec, 2008). The Pluvia system requires an emergency water drainage system. For this purpose most often the spillway system made in roof attics are used. The crest of emergency spillways must be properly defined. The crest ordinate of the spillway will decide on the size of the dead volume. The dead volume is the water that will accumulate on the roof when the Pluvia system fails before the water table reaches the crest of the spillway (Joniec, 2008). Dead volume is an extra load on the roof. Not every roof construction is able to carry extra load from accumulated water. Such a situation can lead to a construction catastrophe. In the article a volume cumulative curve was used to verify the operation of the emergency system. For regular and irregular roof surfaces and for different types of overflows, a calculation scheme was provided.

2. Types of the emergency spillways

Most of the emergency spillways made in the attic are treated as broad-crested weirs (Fig. 1).

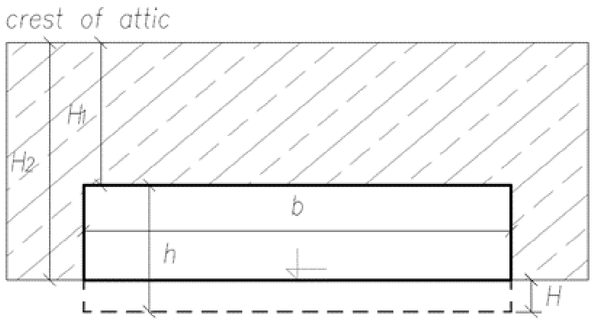

Fig.1. Broad-crested spillway calculation scheme

The first example "w1" assumes a constant value of coefficient m = 0.387 (empirically determined for a broad-crested weir without threshold), the second variant "w2" (less preferred scheme) takes into account the variable value of the coefficient m as a function of the thickness of the overflow layer of the water H and the width of the spillway b (Fig. 1). It was assumed that the maximum dam on the threshold could be equal to the height of spillway. If this height is exceeded, the spillway should be treated as large orifice. Because of the size of the roof surface in relation to the width of the spillways, it was assumed for calculation that $H_0 = H$, omitting the value of the velocity of the incoming water. For both variants, the calculation is as follows (Dąbrowski et al., 1982; Landau and Lifszyc, 2009; Mitosek, 2007):

$$Q = mb\sqrt{2g}H^{3/2} \tag{1}$$

where: m – taken coefficient: in variant 1 „w1" as constant m = 0,387

In variant 2 „w2" calculated depend on $m = 0{,}0474\, ln\left(\frac{H}{b}\right) + 0{,}4075$ (Dąbrowski et al., 1982).

After crossing over the upper edge over the spillway, the classification from the weir to the large orifice should be changed. Outflow from the orifices is calculated from the following formula [1,3,4]:

$$Q = \frac{2}{3}\mu b\sqrt{2g}\left(H_2^{\frac{3}{2}} - H_1^{\frac{3}{2}}\right) \tag{2}$$

Another example of a overfalls can be a shaft spillway (Fig. 2), with a polygonal crest in the plan. The flow rate of the shaft spillway is determined using the same formulas and flow coefficients as for the weirs (1), but instead of the actual length of the threshold L, the equivalent length L_e is applicable.

The equation for calculating the equivalent length is an empirical formula. Equivalent length is calculated from the following relationship (Dąbrowski et al., 1982):

$$L_e = mL\left(1 - 0.78072\frac{y}{L} - 0.81564\frac{H}{L}\right)\sqrt{2g}H^{\frac{3}{2}} \tag{3}$$

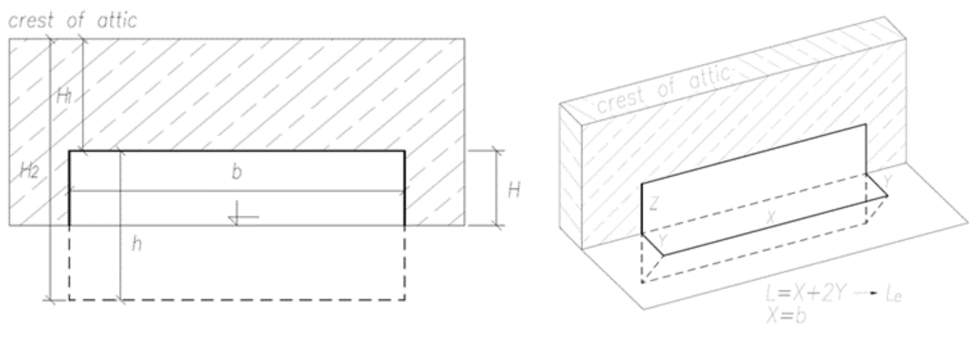

Fig.2. Shaft spillway calculation scheme

3. *Use of cumulative curves to calculate the flow capacity of emergency spillways for warehouse roofs for rain of 300 and 600 l/s ha*

If the roof has a regular structure (rectangular plan, symmetrical, gabled roof), volume growth calculations should not cause more problems. The volume of water accumulated on the roof will be a regular solid. In order to calculate the increase in water volume accumulated on the roof integration with a step of 1 cm was performed. The volume of water up to 2 cm of the dam is assumed to be dead volume. In case of an emergency (no Pluvia work), that volume will be retained on the roof without being able to drain it by the spillways. Further volume increments were calculated for 1 cm water layer. The letter symbols of the dimensions used for calculating the volume increase are shown in Fig. 3 and Fig.4. For subsequent values x and y, volume increment was calculated as a function of roof filling with water.

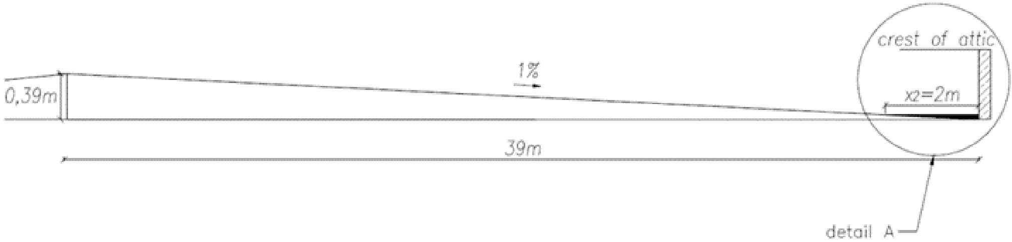

Fig. 3. Scheme of roof volume increment calculation - geometric dimensions of solid volume

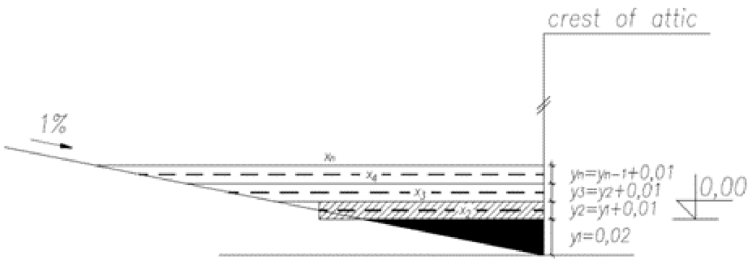

Fig. 4. Scheme of roof volume increment calculation - geometric dimensions of solid volume. Detail A.

Particular attention should be paid to the crest ordinate of spillways placed on the perimeter of the roof. Different ordinates of crests are very common. Flows can be summed only for the same ordinates. Tables 1 and 2 show the calculated flow capacity of the spillways for the two sides of the roof. Dark gray fields, it is the ordinate of the water table for which the spillways started work as large orifices.

Table 1. Summary of the calculated flow capacity of the spillways for both roof planes

No.	Crest	Spillway 1	Spillway 2	Spillway 3	Spillway 4	2∑Q
	cm	l/s	l/s	l/s	l/s	l/s
1	15,6	0,606	0,606	0	0	2,422
2	14,6	1,673	1,673	0,595	0	7,884
3	13,6	3,002	3,002	1,644	0	15,294
4	12,6	4,509	4,509	2,947	0,606	25,143
5	11,6	6,146	16,69	4,426	1,673	57,871
6	10,6	16,39	17,95	6,029	3,002	86,739
7	9,6	17,63	19,10	14,63	4,509	111,731
8	8,6	18,75	20,17	15,80	6,146	121,745
9	7,6	19,80	21,17	16,87	16,69	149,076
10	6,6	20,79	22,13	17,86	17,95	157,458
11	5,6	21,73	23,04	18,79	19,10	165,305
12	4,6	22,62	23,91	19,67	20,17	172,731

No.	Crest	Spillway 1	Spillway 2	Spillway 3	Spillway 4	Spillway 5	2∑Q
	cm	l/s	l/s	l/s	l/s	l/s	l/s
1	15,6	0,623	0	0	0	0	1,25
2	14,6	1,722	0	0	0	0	3,44
3	13,6	3,091	0,614	1,563	0	0	10,54
4	12,6	4,646	1,698	4,383	0	0,625	22,70
5	11,6	6,337	3,046	7,979	0	1,727	38,18
6	10,6	8,125	4,578	12,172	0	3,100	55,95
7	9,6	9,980	6,241	16,855	0,625	4,661	76,72
8	8,6	11,877	7,999	21,951	1,727	6,358	99,82
9	7,6	16,99	9,821	27,402	3,100	8,153	130,94
10	6,6	18,28	11,683	33,162	4,661	10,015	155,60
11	5,6	19,45	13,563	39,193	6,358	11,919	180,96
12	4,6	20,54	15,443	45,461	8,153	13,845	206,88

Then, knowing the surface of the roof A = 0.4446 ha and assuming a rainfall intensity of I = 300 and 600 l/s respectively, the rain flow was calculated:

I = 300 l/(s·ha) Q = 0.13338 m^3/s,

I = 600 l/(s·ha) Q= 0.26676 m^3/s.

On this basis, the time needed to fill another layer of roof volume was calculated. Calculations for both rainfall intensities are summarized in Tab. 2.

By applying a cumulative curve of accumulated rain on the roof, and a cumulative curve of drainage through emergency spillways to one plot, it is easy to determine the equalization of rainfall and outflow through spillways and to determine the dead volume of water, which in case of Pluvia failure will be gathered on the roof increasing its load.

Calculations will be more complex if the roof is not regular in shape and additionally spillways have crests on different ordinates. If this is the case, you can create a contour map of the roof, which will allow you to integrate the increase in water volume as a function of the water layer with a certain step (Fig. 5). The configuration of spillways remained unchanged as for a regular flat roof.

Table 2. Roof filling time for two different rain intensities

No.	y	x	V	ΣV	t_{300}	Σt_{300}	t_{600}	Σt_{600}
	m	*m*	m^3	m^3	*s*	*s*	*s*	*s*
1	0	0	0	0	0	0	0	0
2	0,02	2	2,28	2,28	17	17	9	9
3	0,025	2,5	2,85	5,13	21	38	11	19
4	0,035	3,5	3,99	9,12	30	68	15	34
5	0,045	4,5	5,13	14,25	38	107	19	53
6	0,055	5,5	6,27	20,52	47	154	24	77
7	0,065	6,5	7,41	27,93	56	209	28	105
8	0,075	7,5	8,55	36,48	64	274	32	137
9	0,085	8,5	9,69	46,17	73	346	36	173
10	0,095	9,5	10,83	57,00	81	427	41	214
11	0,105	10,5	11,97	68,97	90	517	45	259
12	0,115	11,5	13,11	82,08	98	615	49	308
13	0,125	12,5	14,25	96,33	107	722	53	361
14	0,135	13,5	15,39	111,72	115	838	58	419
15	0,145	14,5	16,53	128,25	124	962	62	481
16	0,155	15,5	17,67	145,92	132	1094	66	547
17	0,165	16,5	18,81	164,73	141	1235	71	618
18	0,175	17,5	19,95	184,68	150	1385	75	692
19	0,185	18,5	21,09	205,77	158	1543	79	771
20	0,195	19,5	22,23	228,00	167	1709	83	855
21	0,205	20,5	23,37	251,37	175	1885	88	942
22	0,215	21,5	24,51	275,88	184	2068	92	1034

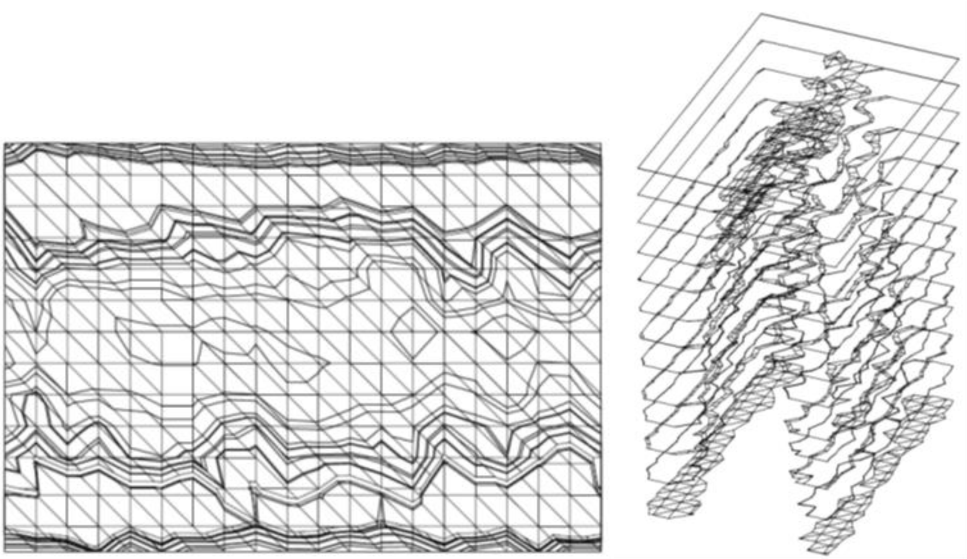

Fig. 5. A contour map of an irregularly shaped roof. Right: the integral step for the next layer shown graphically

Each volume layer corresponds to 1 cm increment of roof water. As the shape of the planes is irregular for each layer of volume increase of 1 cm, the average surface area Fz should be determined. Since the volume solids are irregular, the cumulative rainfall curve should be recalculated. Calculation of the cumulative roof volume curve is shown in Table 3. The cumulative curve calculation for the roof emergency system is summarized in Table 4.

Table 3. Roof filling time for two different rain intensities

No.	Rain	300			600		
	V	t	∑V	∑t	∑V	t	∑t
	m^3	s	m^3	s	m^3	s	s
	18,28	137	18,28	137	18,28	69	69
1	19,58	147	37,86	284	37,86	73	142
2	22,20	166	60,05	450	60,05	83	225
3	24,87	186	84,92	637	84,92	93	318
4	27,42	206	112,34	842	112,34	103	421
5	29,65	222	141,98	1064	141,98	111	532
6	31,73	238	173,71	1302	173,71	119	651
7	33,90	254	207,61	1557	207,61	127	778
8	36,07	270	243,68	1827	243,68	135	913
9	38,37	288	282,05	2115	282,05	144	1057
10	40,90	307	322,95	2421	322,95	153	1211
11	42,99	322	365,94	2744	365,94	161	1372

Table 4. Curved cumulative spillways and orifices for both surfaces of irregular roof

No.	F(z)	V	t	ΣV	t
	m^2	m^3	s	m^3	s
1	1828,00	18,28	7546	18,28	7546
2	1957,50	19,58	2483	37,86	10029
3	2219,50	22,20	1451	80,05	11480
4	2487,00	24,87	989	84,92	12469
5	2741,50	27,42	474	112,34	12943
6	2964,50	29,65	342	141,98	13285
7	3173,00	31,73	284	173,71	13569
8	3389,75	33,90	278	207,61	13847
9	3607,25	36,07	242	243,68	14089
10	3837,25	38,37	244	282,05	14333
11	4089,75	40,90	247	322,95	14580
12	4298,50	42,99	249	365,94	14829

No.	F(z)	V	t	ΣV	Σt
	m^2	m^3	s	m^3	s
1	1828,00	18,28	14677	18,28	14677
2	1957,50	19,58	5684	37,86	20361
3	2219,50	22,20	2106	60,05	22468
4	2487,00	24,87	1095	84,92	23563
5	2741,50	27,42	718	112,34	24281
6	2964,50	29,65	530	141,98	24811
7	3173,00	31,73	414	173,71	25 225
8	3389,75	33,90	340	207,61	25564
9	3607,25	36,07	275	243,68	25840
10	3837,25	38,37	247	282,05	26086
11	4089,75	40,90	226	322,95	26312
12	4298,50	42,99	208	365,94	26520

By applying a cumulative curve of accumulated rain on the roof, and a cumulative curve of drainage through emergency spillways to one plot, it is easy to determine the intersection point of these two graphs, which corresponds to the equalization of both volumes, and also give the equalization time.

4. Summary

The ability to determine the equilibrium point on the cumulative rainfall and drainage curves shows clearly whether the designed emergency water spillways system is able to drain excess water in the case of Pluvia failure. In addition, it is possible to determine the time after which the inflow will be equal to the outflow. This in turn allows to accurately calculate the increase in volume of dead volume. Calculating the dead volume is very important from the load capacity of the structure point of view. It is possible that an additional load (accumulated water) will cause the ultimate limit to exceed, which may result in a construction catastrophe.

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Scientific and technological aspects of a two-stage leachate pretreatment at Lviv municipal solid waste landfill

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Abstract

An analysis of the environmental hazards, caused by landfills of solid wastes in Ukraine, is presented. In particular, the problem of treatment of accumulated landfill leachate is considered. Ecological risks of Lviv landfill of municipal solid waste (MSW) are analyzed. A review of modern technologies of leachate treatment is fulfilled. The advantages and disadvantages of the most promising treatment technologies such as biochemical aerobic treatment and reagent physical-chemical treatment are considered in details. An analysis of the chemical composition of leachate, accumulated at Lviv MSW landfill, is performed. The prospect of leachate treatment in aerated lagoons is investigated. Effects of hydraulic retention time and temperature on leachate treatment efficiency of aerated lagoon have been obtained experimentally. A comparative analysis of the physical-chemical treatment efficiency of Lviv landfill leachate using different reagents is performed. The efficiency of reagent-coagulation treatment with use of Fenton reaction is proved. Different technological schemes for the realization of the reagent stage of leachate treatment have been investigated in laboratory conditions. The strategy of experimental and industrial testing of integrated two-stage technology of leachate treatment on a pilot plant is developed. Scheme of the pilot plant installation and technological equipment are proposed. The concept of in situ realization of integrated two-stage technology of leachate pretreatment for minimization of the ecological risks at Lviv MSW landfill is developed.

Keywords

leachate pretreatment, MSW landfill, aerated lagoon, reagent-coagulation method, Fenton reaction, ecological risk.

1. *Introduction*

Existing landfills in Ukraine are now a powerful sources of environmental risks. One of the most hazardous component is landfill leachate, accumulated in ponds or leaking out into the surface and groundwater sources as a result of the absence of anti-filtration screens, leachate drainage systems and treatment facilities at Ukrainian landfills. Methods of minimization of such environmental risk are investigated in this study on the example of Lviv municipal solid waste (MSW) landfill in Grybovychi, used till the May, 2016 as the main landfill of the Lviv city.

Lviv MSW landfill is located at a distance of 3 km from the northern border of Lviv, near the villages of Great Grybovychi, Zbyranka and Malekhiv. Landfill has been functioning since the 1960's and occupies an area of about 40 hectares (Fig. 1).

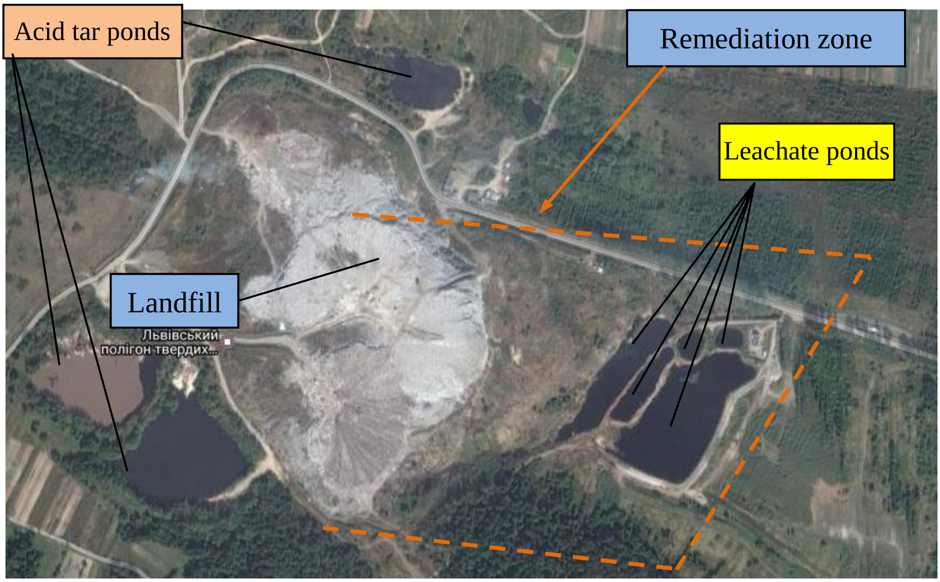

Fig. 1. An aerial view of Lviv MSW landfill

The thickness of the landfill's body reaches 50 m in the south-eastern part, while in the northwest part ranges from 1−3 m to 10 m (Voloshyn, 2012). Since the landfill has exhausted its resource, it is now at the closing stage. A prerequisite for technical remediation of the landfill is the slope grading (SBS, 2005). The normative value of landfill's slope depends on the subsequent using of the territory; the maximum slope is recommended for planting of forests, shrubs and trees − not more than 18°. The slopes of the Lviv MSW landfill are now much higher than the normative value. To obtain the necessary slope angle, slope reduction should be provided by filling the adjoining area with neutral solid materials (soil, clay, solid wastes, etc.). Since the remediation zone at Lviv MSW landfill is occupied by leachate ponds with total capacity of 100–150 thousand m^3 (Fig. 1), the priority tasks are now treating and removing of these leachate. This is the first obvious step allowing the landfill's slope reduction and hence the landfill's remediation process.

On the territory of Lviv MSW landfill there are also several ponds with accumulated dangerous industrial wastes – acid tars (Fig. 1). On the other hand, this situation is not typical one for other Ukrainian landfills, and therefore the methods of optimal strategy of acid tars utilization are not discussed in this study.

To resolve the problem of the environmental risks caused by the Lviv landfill leachate (typically as for most Ukrainian landfills), it is necessary to distinguish two different phases:

- Phase #1: short-term treatment of accumulated leachate in the context of landfill remediation;
- Phase #2: long-term treatment of leachate, which will be formed in the landfill's body for decades after the landfill's closing and remediation due to the biological processes of the decomposition of the organic part of the wastes.

These phase are quite different in the terms of the leachate discharge, it's physical and chemical characteristics and duration of implementation.

Therefore, it is ineffective to plan for the implementation of both these phases using a unified treatment technology: both from the technological point of view (the impossibility of providing full loading and efficient equipment's operation), and from the financial point of view (significant overexpenditure).

The most world widespread technologies for leachate treatment (Methodical recommendations, 2012):

1. reverse osmosis technology;
2. evaporation and drying technology;
3. technology of filtrate binding;
4. technology of biochemical treatment in anaerobic and aerobic conditions.

Technologies 1–3 are energy and resource-consuming ones; significant capital and operating costs are required for their implementation. Potentially attractive is anaerobic technology of leachate treatment (Jamalova, 2015), so as biogas production is an additional product of the treatment process. However, for successful implementation of this technology at practice, it is necessary to strictly observe the parameters of its implementation, that is complicated due to the alternating chemical composition of the leachate. A significant disadvantage of the anaerobic process of leachate treatment is sensitivity to changes in temperature and pH, as well as to various toxic substances contained in leachate (Gao et al., 2015; Hasa et al., 2009).

Preliminary biochemical leachate treatment in the aerated lagoons, that was proven successfully in Great Britain, Norway, Sweden and some other countries (Call et al., 2006; Sawaittayothin et al., 2007; Robinson et al., 1988; Mehmood et al., 2009;

Maehlum 1995), followed by reagent physical-chemical treatment, can be especially effective technology for implementation at Ukrainian landfills.

Call et al., 2006 and Sawaittayothin et al., 2007 founded that microbial communities of wastewater treatment systems are well adapted to the destruction of complex organic compounds in a wide range of waste flows. In landfill leachate treatment plants, after a while can be eventually activated specific aerobic biocenose, which is effective in oxidizing of organic compounds and ammonium ions even of the highly concentrated leachate in aerated lagoon conditions.

Practical aspects of the leachate treatment in aerated lagoons were investigated by a number of researchers, described in detail the results of the leachate treatment at various MSW landfills: Bryn Posteg Landfill in Wales, Great Britain (Robinson et al., 1988), Bell House Essex, Great Britain (Mehmood et al., 2009), Esval polygon in Norway (Maehlum, 1995). High required effects of landfill leachate treatment were achieved in all abovementioned case studies that allowed subsequent secondary treatment of pretreated leachate at municipal wastewater treatment plants (Essex, Wales) or at the artificial wetlands (Norway). Studies have shown that for each landfill leachate individual optimal conditions for the biochemical oxidation in aerobic lagoons are required.

Reagent methods, in particular coagulation, are widely used for preliminary or complete landfill leachate treatment in Germany, Great Britain and other countries. These methods allow to remove the biologically non-oxidable humic and fulvic acids from leachate, as well as other specific "heavy" contaminants, including heavy metal ions, organochlorine compounds, etc. All German leachate treatment plants use combined treatment methods. More than 60% of theese plants use the biochemical treatment as the first stage, including 15 treatment plants with the chemical treatment at the second stage (Wiszniowski et al., 2006).

Using the Fenton process and its varieties (photo-Fenton process, electro-Fenton process) at the reagent treatment stage in the schemes of preliminary or complete landfill leachate treatment are investigated systematically for the last two decades (Wiszniowski et al., 2006; Badawy et al., 2013; Bae et al., 1997; Deng et al., 2006).

Studies have shown that the efficacy of leachate treatment from organic contaminations with traditional or modified Fenton process depends on the type of leachate, the dosage of the reagents, the method of input and mixing of reagents, the initial and final values of pH, temperature, intensity of aeration and so on. It should be noted that oxidation and coagulation are both important in terms of the removal of heavy organic contaminants (Bae et al., 1997).

Thus, an analysis of the results of previous studies shows that an effective method for Lviv MSW landfill leachate treatment can be a two-stage integrated technology, where the first stage is biochemical treatment in aerated lagoons (Moroz et al., 2017), and the second stage is a reagent-coagulation treatment with use of Fenton reaction.

2. Materials and Methods

2.1. Lab-scale simulation of the landfill leachate treatment in aerated lagoon

Laboratory simulation of the landfill leachate treatment in aerated lagoon was carried out on the installation shown in Fig. 2. Sample of the Lviv MSW landfill leachate from the leachate pond #2 in quantity W=4.0 dm^3 was poured into a laboratory flask, which simulated the environment of the aerated lagoon. Initial parameters of the leachate: pH − 8.64; concentration of ammonium ions (NH_4–N) − 558 mg/dm^3; chemical oxygen demand (COD) − 4850 mg O_2/dm^3.

A laboratory aerator was installed at the bottom of the flask. Laboratory compressor supplied air with a constant flow rate of 0.042 dm^3/s. Leachate samples were periodically sampled from the flask and the values of COD and (NH_4–N) concentration were determined.

The pH value and the concentration of dissolved oxygen in the leachate were measured during the study. The concentration of dissolved oxygen was measured by a portable oxygen sensor sensIon6™, the concentration of ammonia nitrogen was determined by photometric method (LND 211.1.4.030-95, 1995).

COD was determined according to the standard method (ISO 6060: 1989), pH was determined by the potentiometric method using a portable pH/ISE/mV/°C-meter *sens*Ion™2.

Raw, untreated leachate was investigated. Leachate aeration was carried out in continuous mode at a constant temperature of 20 °C.

In the first mode of research (further − static research mode), the maximum possible effects of treatment by the COD and ammonium ions were achieved.

Fig. 2. Experimental lab-scale installation for the study of aerobic leachate pretreatment

In the second mode of research (further − dynamic research mode), real-time continuous treatment in aerated lagoon was modeled by means of periodical (once per day) outflow of some portion of pretreated leachate and the same inflow of the raw leachate. The volume of inflow and outflow portions corresponded to a certain value of the hydraulic retention time in the aerated lagoon. Study of the treatment efficiency in the lab-scale aerated lagoon for each value of the hydraulic retention time was performed until the three-day constant concentrations were reached; after that the study was proceed for the next value of the hydraulic retention time.

2.2. Lab-scale simulation of the reagent-coagulation stage of the landfill leachate treatment

Raw, untreated leachate from the pond #2 of Lviv MSW landfill was investigated. In the process of laboratory simulation of the reagent stage of leachate treatment, different types of reagents were gradually added into the leachate to obtain the best phase separation effect due to coagulation and flotation of contaminants. The researches were performed in measuring cylinders of 250 ml capacity. The testing reagents in portions and the consecutive reagent compositions were added to the leachate aliquots with volume of 200 ml. The modes of mixing and aeration of the medium after the dosing of reagents were studied. The control of the reagent leachate treatment was carried out by measuring the optical density of the medium. The efficiency of reagent stage of leachate treatment was determined by changing the concentrations of contaminants.

2.3. Chemical composition of raw leachate at Lviv MSW landfill

The average chemical composition of the raw leachate at Lviv MSW landfill is presented in Table 1. The maximum permissible concentrations (MPC) of contaminants in accordance with Ukrainian legislation are also indicated: MPC-S − for outputs into the municipal sewerage systems; MPC 1 − for outputs of the treated wastewaters into the surface water bodies of the domestic water-using type (type 1); MPC 2 − for outputs of the treated wastewaters into the surface water bodies of the fish-farming type (type 2).

Comparative analysis indicates that the principal contamination parameters that do not allow the output of the raw leachate into urban sewerage networks are ammonium nitrogen, BOD_5, Fe (total), TDS, chlorides and COD. It should be noted that analogical leachate composition (with slight pecularities, depending on local conditions) is a typical one for most Ukrainian landfills.

Table 1. The average chemical composition of the raw leachate at Lviv MSW landfill

N	Parameters	Units	Concentrations			
			Leachate pond #2	**MPC-S**	**MPC-1**	**MPC-2**
1	BOD_5	mg/dm^3	1330	350*	15*	*
2	Cd	mg/dm^3	0.047	0.01	0.001	0.001
3	Cl	mg/dm^3	7267	350	350	300
4	COD	mg/dm^3	4850	875*	80*	*
5	Co	mg/dm^3	0.227	1.0	0.1	0.005
6	Fe (tot.)	mg/dm^3	71.0	2.5	0.3	0.05
7	Mn	mg/dm^3	0.042	30	0.1	0.01
8	$NH_4 - N$	mg/dm^3	558.6	30	2	0.5
9	Ni	mg/dm^3	0.13	0.5	0.1	0.01
10	Pb	mg/dm^3	0.14	0.1	0.03	0.01
11	Sr	mg/dm^3	0.048	26	7	10
12	Suspended solids	mg/dm^3	320.0	500*	15*	*
13	Synthetic surfactants (anionic)	mg/dm^3	32.4	20	0.5	**
14	Total dissolved solids	mg/dm^3	13250	1000	**	**

* − individual requirements; ** − no requirements

3. Results and discussion

The analysis of the results of aerobic biochemical leachate treatment in the dynamic research mode has shown that the maximal asymptotic values of the treatment effects for COD and ammonium ions were obtained in 16 days.

Microbiological analysis of the leachate sludge (Figure 3) showed the appearance in the leachate of wide spectrum of microbiological aerobic culture, which differs from the the respective culture of active sludge at municipal wastewater treatment plants (WWTP).

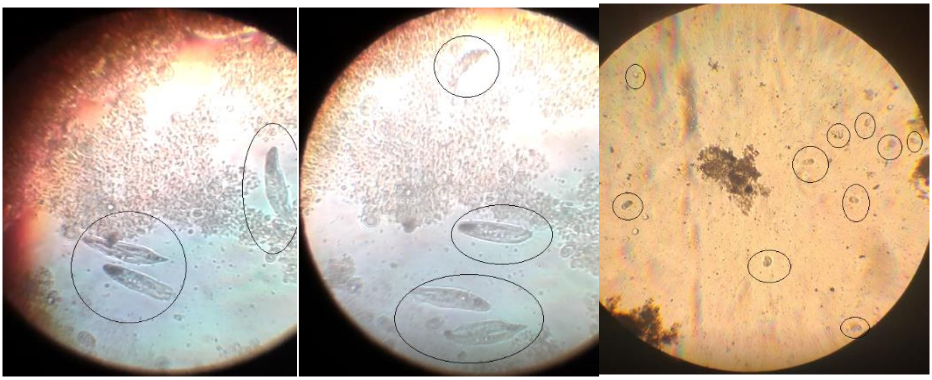

Fig. 3. Microorganisms detected in the leachate after 16 days of aeration

The concentration of dissolved oxygen was maintained in the lab-scale aerated lagoon at the constant level of 3.9 mg/dm^3 for all values of the HRT in the range 8−16 days.

The dependencies of the treatment effects in the aerated lagoon at different HRT (τ_d) are presented at Fig. 4.

Treatment effects $E_{NH_4^+}$ and E_{COD} were calculated by definition using equations (1)–(2):

$$E_{NH_4^+} = \frac{C(NH_4 - N)_{in} - C(NH_4 - N)_{fin}}{C(NH_4 - N)_{in}}\ , \tag{1}$$

$$E_{COD} = \frac{(COD)_{in} - (COD)_{fin}}{(COD)_{in}}\ , \tag{2}$$

$C(NH_4 - N)_{in}$, $C(NH_4 - N)_{fin}$ − respectively initial and final concentrations of ammonia nitrogen, mg/dm^3; $(COD)_{in}$, $(COD)_{fin}$ − respectively initial and final COD values.

The constant operating mode of the treatment installation in a dynamic mode was achieved on average after 9−11 days. The optimal HRT of the leachate in the lagoon in the case of the implementation of biochemical aerobic treatment in such study conditions is equal of about 11 days. In this case were obtained relatively high pretreatment effects of the leachate: 35% of ammonium ions reduction and 50% of the COD reduction. On the other hand, for a two-stage integrated pretreatment technology, the optimal HRT of the first stage should be defined on the basis of the technical and economic evaluation of the integrated technology as a whole.

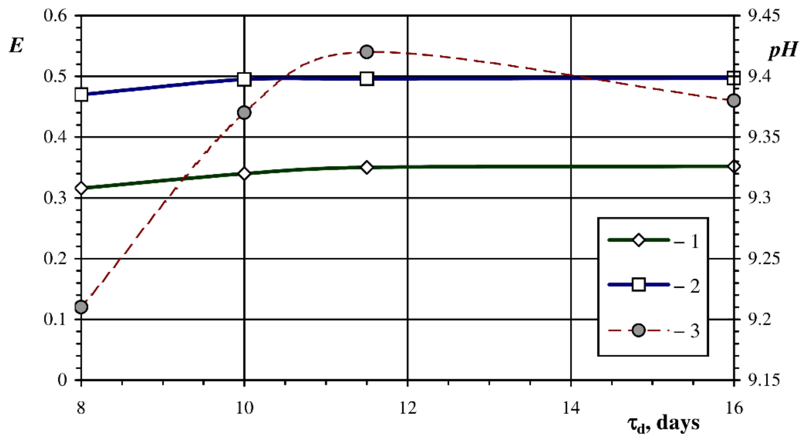

Fig. 4. The dependencies of treatment effects ($1 - E_{NH_4^+}$; $2 - E_{COD}$) and pH value (3) in the lab-scale aerated lagoon as a function of hydraulic retention time (τ_d)

At the first step of the study of the leachate reagent-coagulation treatment stage the most suitable reagents were selected, which ensure the maximum efficiency of the condensed contaminants separation from the leachate. For this purpose, reagents the most commonly used in the industrial wastewater treatment were investigated: crystallohydrates of aluminum sulfate and iron (II) sulfate, aluminum polyoxychloride, as well as Fenton's reagent. The kinetics of the optical density changing was investigated after the adding of reagents' solutions into the leachate. The best results were obtained in the case of using Fenton reagent (Fig. 5). This result can be explained by intensive oxidation of suspended and dissolved components of leachate by hydrogen peroxide. The ferrous salts that are the part of the Fenton's reagent also act as a coagulant in this case.

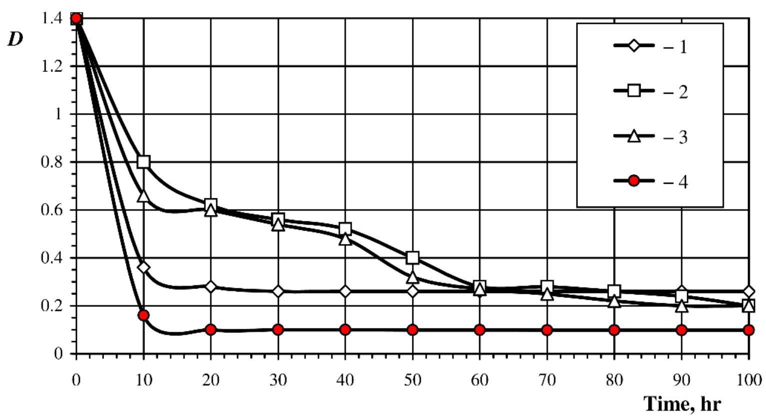

Fig. 5. Lviv MSW landfill leachate optical density changing using the different reagents: $1 - Al_2(SO_4)_3 \times 18H_2O$; $2 - FeSO_4 \times 7H_2O$; $3 - Al_n(OH)_{(3n-m)}Cl_m$; 4 – Fenton's reagent

At the second step of the study, different modifications of the Fenton's reagent composition were investigated, to achieve the maximum treatment effect at the minimum possible costs of the composition. In the testing process, the proportions of the reagent were changed, the effects of mixing and aeration and the effect of adding the polyacrylamide (PAA) solution as a flocculant were investigated (Table 2). The criterion for evaluating the technology's efficiency was the final treatment effect.

Table 2. Investigated reagent compositions for the Lviv MSW landfill leachate's treatment on the reagent stage

N	Reagent dosage, kg/m^3			Other options	
	$FeSO_4 \cdot 7H_2O$	$Al_2(SO_4)_3 \cdot 18H_2O$	H_2O_2 (60%)	PAA	aeration
1	3.2	1.9	1.6	–	–
2	2.8	1.0	0.51	–	–
3	3.2	–	1.06	+	–
4	3.2	1.9	1.6	+	–
5	2.8	1.0	0.51	+	–
6	2.4	0.9	0.5	+	+
7	2.1	0.7	0.45	+	+

The results of Lviv MSW landfill leachate's reagent-coagulation treatment using various variants of the process implementation are presented in Table 3.

Table 3. Results of reagent-coagulation treatment of the Lviv MSW landfill leachate by various reagent compositions

Parameters	Raw leachate, mg/dm^3	Leachate after reagent treatment, mg/dm^3						
		Number of reagent composition (see Table 2)						
		1	2	3	4	5	6	7
BOD_5	1330	1000	1400	450	395	420	396	354
Cl	7267	3828	4608	4608	4431	4431	4152	3899
COD	4850	1600	2400	1500	1800	1700	1200	800
Fe (tot.)	71.0	22	178	8.64	0.3	39.6	1.25	0.52
$NH_4 - N$	558.6	334	370	98	18	22	32	26
Suspended solids	320.0	389	1042	420	390	400	390	350
Synthetic surfactants (anionic)	32.4	4.68	7.2	7.0	6.5	6.5	8.0	7.2
Total dissolved solids	13250	5802	6564	2500	1450	1728	1558	1326

As can be seen from the results presented in Table 3, the highest effects of leachate treatment by the reagent method were obtained for the 4th and 7th reagent compositions.

At the same time, almost none of the variants of reagent processing does not allow to reach the normative value of MPC-S for COD (Table 1), which is especially important in terms of further discharging of pretreated leachate into the city sewer system. Optimistic results were obtained only with the application of a two-stage integrated treatment technology: by combining the aeration process over a long period of time with reagent-coagulation treatment.

The effectiveness of such two-stage treatment, optimal modes of realization of the biochemical stage and parameters of the implementation of the reagent-coagulation stage should be clarified as a result of appropriate feasibility study using the in-situ testing results.

Geotechnical integrated technology can be offered for the leachate pretreatment at the Lviv MSW landfill, using the natural reactors − existing leachate ponds. In-situ testing of proposed two-stage technology should be carried out on a pilot treatment plant in the continuously-periodic mode of the implementation of both stages of the integrated technology. This will enable a comprehensive study of both stages with the aim of optimizing technology and its industrial implementation. In the process of in-situ testing, periodical sampling should be done with the purpose to define the peculiarities of the implementation of proposed integrated technology under full-scale natural conditions.

An existing pond at Lviv MSW landfill, filled with leachate, 20×7×1.5 m (total volume of 210 m^3) was selected for the pilot plant's georeactor installation. Previously georeactor should be emptied and screened by two geomembrane layers, fixed to the bottom and the side walls of the georeactor. At the next step georeactor will be equipped with 2 jet-type aerator pumps, which will provide the required aeration in the testing process of the first aerobic biochemical stage of leachate pretreament of proposed integrated technology in natural conditions. Aerator pumps will also be used to mix reagents that will be fed into the jet fluid-air flow during the approbation process of the second reagent-coagulation treatment stage of the technology. The optimal parameters of realization of both stages of the integrated technology will be defined in the result of this approbation in experimental-industrial conditions at the pilot treatment plant. This will make it possible to implement the integrated technology industrially for the purpose of Lviv MSW landfill's leachate treatment that will allow to eliminate the leachate-storage ponds and to realize the project of physical and biological landfill's remediation.

At the full-scale industrial leachate treatment plant the georeactor of pilot plant is planned to be used to implement the coagulation stage of integrated technology. Implementation of the first biochemical stage is expedient in another existing leachate-storage pond with larger volume. To install an aerated lagoon on its basis, this pond should be equipped with a group of jet-type aerator pumps that would provide the required intensity of aeration. The overflow system should provide the required hydraulic retention time in the aerated lagoon before the leachate feeding to the coagulation-flocculation stage. After two stages of the proposed integrated technology, the pretreated leachate should be pumped through the existing pressure pipeline into the Lviv city sewer main and further to the final treatment at the Lviv municipal WWTP.

4. Summary and Conclusions

Landfills in Ukraine do not meet the requirements of environmental safety and require urgent measures for their remediation. An important part of this problem is the elimination of leachate-storage ponds.

Leachate treatment can be especially effective using complex technology approach that includes technologies of pretreatment and final treatment at municipal WWTP. An analysis of leachate pretreatment methods demonstrates the effectiveness of the integrated two-stage biochemical and reagent-coagulation treatment technology.

Biochemical treatment of raw leachate in aerated lagoons is substantiated as the first stage. The second stage involves the use of the reagent-coagulation treatment with use of Fenton reaction. Optimal process parameters of the lab-scale pretreatment of Lviv MSW landfill raw leachate have been defined.

Research on a pilot treatment plant is a necessary stage of approbation of the proposed technology in experimental-industrial conditions in order to determine the optimal parameters for the successful realization of the full-scale integrated two-stage pretreatment technology at the Lviv MSW landfill. The concept of carrying out the pilot industrial study and the subsequent realization of a full-scale industrial treatment plant for the purpose of eliminating the environmental hazard caused by accumulated leachate has been developed.

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Ways of increasing the efficiency of anaerobic-aerobic processes of biological wastewater treatment

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Abstract

The aim of the work is to substantiate the technological methods of increasing the efficiency of the process of biological treatment of urban household sewage.

Applying the method of physical modeling of the system of biological treatment of urban sewage, it was tried to use two types of active sludge recirculation in aerobic tank (internal and external) to improve the reliability of exploitation of treatment facilities.

During studies, the value of the coefficient of external recirculation of active sludge did not change and was maintained at a constant level, which corresponds to the value of wastewater discharge. With an increase in the coefficient of internal recirculation of sludge mixture from 1.0 to 2.6, an increase in the efficiency of purification by organic matter (BOD_f) was observed from 76.4 to 97.5%. At the same time, the value of the concentration of dissolved oxygen was provided at a sufficient level, which does not reduce the rate of aerobic oxidation of organic substances.

Scientific novelty of results: got subsequent development scientific principles in relation to the methods of intensification of process of biological treatment of sewage exactly taking into account the previous leadthrough of denitrofscation in anaerobic terms with the next stage of the aerobic sewage treatment in aerobic tank. This allows to ensure the high efficiency of wastewater treatment of suspended solids, organic and nitrogen-containing compounds for the established ecological safety standards.

The results of the conducted scientific research allow to carry out purposeful correction of technological regimes of the process of biological treatment of urban household sewage in order to increase its efficiency.

Keywords

sewage, active sludge, treatment facilities, aerobic tank, recycling, ecological safety

1. Introduction

Contamination of natural environment for today is a basic factor worsenings of health population, which are investigation of increase of level of ecological danger (Shmandiy et al., 2015).

Today in connection with worsening of quality of water in the opened reservoirs sharply the problem of diminishing of loading stands on a water pool. In a considerable measure this problem can be decided due to the increase of efficiency of work of treatment facilities of industrial and municipal enterprises that dispose of insufficiently purified water from the reservoir. The most important requirement for the operation of sewage treatment plants is the high efficiency of sewage treatment, which ensures the ecological norms of the maximum allowable discharge of pollutants into water objects (Yurchenko and Smirnov, 2015; Svyatenko et al., 2011). For the effective carrying out of the process of aerobic biological treatment of wastewater, a significant amount of air is required for aeration, which requires significant energy costs (Couras et al., 2015; Schlafer et al., 2002). In connection with the increase in the cost of electricity, in order to reduce the cost of wastewater treatment, it is advisable to implement such processes that require a lower cost of the above energy resource (Smirnov and Yurchenko, 2016). Therefore the purpose of our research is perfection of technology anaerobic-aerobic cleaning sewage in order to increase the efficiency of sewage treatment.

2. Materials and Methods

As a research object was chosen a system for the biological treatment of urban household wastewater.

For the receipt of necessary information research was carried out on two parallel operating laboratory models of denitrifikator and aerotank, having a volume of 10 dm^3, equipped with the necessary systems for the dispensing of sewage, forging sludge mixture, separation and pumping of active sludge, the removal of the cleared water.

The scheme of the laboratory installation is shown in figure 1.

Every setting included the followings basic elements:

1. A system of dispensing wastewater:

- calibrated capacities with variable working volume, from which waste water was used. The maximum volume of each container is 16 dm^3. Capacities can be connected to each other by 2–3 as conjugated vessels. The number of simultaneously used capacities depended on the required waste of wastewater;
- substrate dosing system with a special dispensing device.

2. Two parallel running aeronautical models, each of which included:

- model of aerobic tank, which is equipped the system of aeration air;
- model of reservoir for the species is spolked to nitrogen, but not, equipped a mechanical device for interfusion of sludge mixture;
- settling tank with variable working volume.

3. Membrane pump for pumping reverse sludge (autonomous for each model).
4. Air supply system with rotameters and hoses for supplying air in a container.

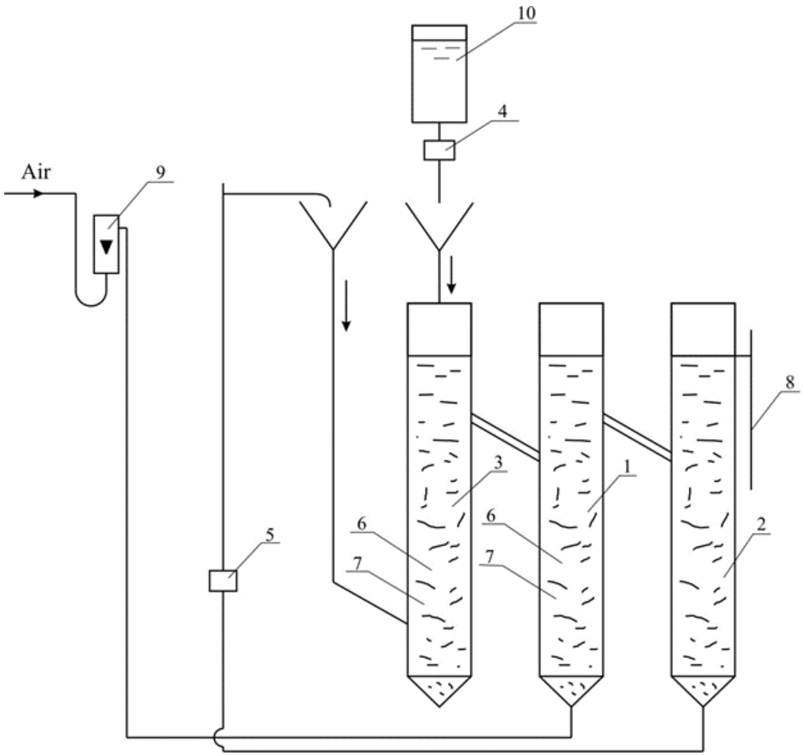

Fig.1 Chart of setting with an aerobic tank and anaerobic tank**.**
1 – aerobic tank; 2 – settling tank; 3 – anaerobic tank; 4 – wastewater dispenser; 5 – circulation pump; 6 – sensors of dissolved oxygen; 7 – pH sensors; 8 – a drain of purified water; 9 – air flow meter; 10 – capacity for sewage.

The installation operates as follows: the waste water from the container 10 through the dispensing device 4 continuously enters the anaerobic tank 3, which is equipped with a mechanical mixer. In the anaerobic tank, which is not used, an active sludge from the settling tank is also fed through the circulation pump 5. From the anaerobic tank**,** of sludge, the gravity flows into the aerobic tank. In an aerobic tank silt mixture is aerated air which is given through a flowmeter 9. From aerobic tanks, the sludge mixture with purified water is poured into the settling tank 2, where the active sludge is deposited in the conical part, and the purified water through the overflow 8 is removed from the installation.

The experiment consisted of several series, each of which changed one of the regulatory parameters. The following parameters were ensured: at a constant degree of external recirculation of sludge, permanent concentrations of dissolved oxygen and the initial concentration of organic contaminants in wastewater, the coefficient of internal recirculation of sludge mixture into aerobic tanks and the dose of sludge changed.

During the experiment, the following control parameters were monitored:

- degree of recirculation of sludge;
- amount of excess silt;
- air flow rate for aeration;
- flow of circulating sludge;
- wastewater for cleaning.

Out sighting was also carried out after parameters which are controlled:

- concentration of residual organic pollutants in cleared water,
- concentration of active sludge in aerobic tank,
- concentration of active sludge in anaeroic tank,
- pH,
- the temperature of the sludge mixture in the aerobic tank,
- ash-content of sludge
- coefficient of growth of sludge
- specific rate of oxidation of organic pollutants.

The experiment was carried out at a value of wastewater discharge in the installation of 1000 ml/h and the change in the coefficient of external recirculation of sludge mixture from 1.0 to 2.6.

3. Results and Discussion

As known, the mode of operations of aerotank is determined the values of technological and construction parameters: its volume, the value of the sludge recirculation coefficient, the concentration of dissolved oxygen, the dose of active sludge, the intensity of aeration, and others.

Sewage treatment was carried out in a laboratory model in anaerobic-aerobic conditions. Experimental works were carried out using actual sewage from the Kremenchug city and active sludge from urban waste water treatment plants. High-quality indexes of initial flow waters which were used in experiments resulted in a table 1.

Table 1. Qualitative indicators of wastewater used in experiments

Characteristic	**Unit measurement**	**Medium value indicator**
pH	pH unit	7,1
BOD_f	mg/dm^3	267
COD	mg/dm^3	486
Suspended matter	mg/dm^3	223
Ammonium nitrogen	mg/dm^3	25,6
Nitrates	mg/dm^3	1,2
Sulphates	mg/dm^3	33
Chlorides	mg/dm^3	46
Phosphates	mg/dm^3	15,4
Total Iron	mg/dm^3	1,6

At the beginning of the experiment in a reservoir for the species is spolked to nitrogen, but not device, a mixture of active sludge and sewage was mixed with a mechanical stirrer without aeration air.Time of the anaerobic cleaning changed from 4,4 to 6,9 hours. Correlation of organic contaminants and active sludge during the experiment supported thus, that the load on the active sludge ranged from 214 to 480 $BOD_{f.}$ mg/day. Ash-content an active sludge during the leadthrough of researches was range from 0,19 to 0,27. There was denitrofication at foregoing terms, during which the nitrogenous compounds recommence to molecular nitrogen and from a liquid phase transferred to atmospheric air. A degree of purification of sewage from nitrogenous compounds was 78–82%. Organic matters oxidize oxygen which was in composition nitrites and nitrates, with the most oxidized light oxidizing substances: carbohydrates, organic acids, alcohols. Bacteria that restore nitrogen compounds can not use high-molecular polymer compounds. The maximum intensity of the process is achieved at pH values of 7.0–8.2. The temperature of reactionary environment in aerobic tanks and reservoir for the species is spolked to nitrogen, but not during the leadthrough of researches was supported within range of 22–24 ° C.

After completion of anaerobic period of cleaning the preliminary cleared flows got in an aerobic tank, where constantly gave aeration air. For the increase of cleaning degree after organic compounds (for $BOD_{f..}$) in an aerobic tank sent an additional amount an active sludge, preliminary adapted to the certain level of loadings in a parallel acting aerobic tank. The coefficient of internal recirculation of the sludge mixture was gradually increased from 1,0 to 2,6 from the expense of flow waters on cleaning. Thus cleaning degree after organic compounds (for $BOD_{f.}$) constantly increased and on the final stage of experiment was 97,5%. The high values of cleaning degree after $BOD_{f.}$ testify to the presence in composition of active prokaryotic silt in microorganisms. Prokaryotes are optional anaerobes which at presence of oxygen pass to the ordinary breathing. Dependence of efficiency of cleaning of flows after $BOD_{f.}$ from the coefficient of external recirculation of the sludge mixture presented in Table 2.

It is set as a result of researches, that periodic change of terms of existence of microorganisms of active sludge leads results in diminishing of speed of their besieging, that worsening of sedimentation properties of the silt. A main factor which influences on a sludge volume index is loading on an active sludge (Svyatenko et al., 2011).

Loading on the active sludge of q was determined by the formula:

$$q = \frac{24 \cdot (L_н - L_к)}{a_i \cdot (1 - S) \cdot \tau_{at}}, \quad (1)$$

where: $L_н$, $L_к$ – concentration of $BOD_{f.}$, accordingly on an entrance and on an exit from an aerobic tank, mg O_2/dm^3; a_i – dose of sludge, g/dm^3; S – ash-content an active sludge, part of unit; τ_{at} – time of aeration, h.

Table 2. The dependence of the efficiency of sewage treatment according to $BOD_{f.}$ on the coefficient of external recirculation of sludge mixture

N	**Dose sludge, g/dm^3**	**Load on sludge, mg/g dose**	**Concentration dissolved oxygen, mg/dm^3**	**Duration aerobic phase, year**	**Coefficient exterior of recirculation**	**Efficiency cleaning of drainage for the $BOD_{f.}$**
1	3,0	241	1,62	3,33	100	76,4
2	3,2	272	1,79	2,86	150	83,2
3	3,4	261	1,68	2,5	200	86,8
4	3,5	342	1,74	2,38	220	90,9
5	3,7	276	1,86	2,27	240	93,2
6	4,1	283	1,61	2,22	250	95,1
7	4,4	261	1,66	2,17	260	97,5

It is thus possible to draw conclusion, that the greatest efficiency of cleaning of flows was got at a serve in an aerobic tank aerated (preliminary adapted to the necessary loading) a sludge in an amount, that in 2,6 times exceeds the expense of flow waters on cleaning.

For determination of dependence of sludge index from loading on an active sludge a few series of experiments, in which the parameters of process changed consistently, were conducted in aerobic tanks: concentration of initial organic contaminants in flow water and dose of sludge. The concentration of dissolved oxygen in an aerobic tank was supported within the limits of 1,8–2,1 mg/dm^3. The dependence of the sludge volume index on the load on active sludge in aerobic tanks is obtained, which is resulted in Table 3 and in Figure 2.

Table 3. **Dependence of the silo sludge volume index** on the load on active sludge in aerobic tanks

Load on active sludge, mg BOD_f/g /day	214	232	258	269	342	446	480
Sludge volume index, cm^3/g	180	165	161	168	171	201	216

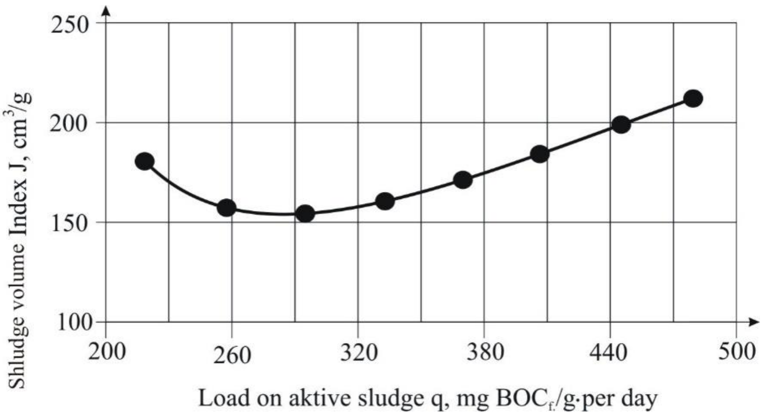

Fig. 2 – Dependence of sludge volume index on the load on active sludge in aerobic tanks

The results of the processing of experimental data indicate that there is a parabolic relationship between the sludge volume index characterizing the sedimentation properties of the sludge and the load on the active sludge. It is known that than less value of sludge volume index, the more effective carried out division of sludge and cleared water in second sedimentation, that is why it is necessary to support the certain loadings on an active sludge. The rational values of sludge volume index were certain at the optimum loadings on an active sludge. Such value of mule index for an aerobic tank makes 161 cm^3/g with an optimal loading of 258 mg BOD_f/g per day.

On the final stage of process of the wastewater treatment, when there is a negligible quantity of organic compounds in water, there is a process of oxidation of ammonium nitrogen to nitrites. Bacteria of the genus Nitrosomonas oxidize nitrogen (nitrogen) ammonium to nitrite. As a substrate of the bacterium Nitrosomonas can use ammonia nitrogen, urea, uric acid, guanine, but the organic part of the molecule is not consumed. At the second stage, bacteria of the genus Nitrobacter oxidize nitrites to nitrates.

Reactions of ammonia nitrogen oxidation:

1) $2\ NH_4^+ + 3O_2 \rightarrow 2\ NO_2^- + 2H_2O + 4H^+$

2) $2\ NO_{2-} + O_2 \rightarrow 2\ N0_{3-}$

The results of researches rotined that intensity of process of oxidation of nitrogen to ammonium was very low, thus part of nitrites does not exceed 3,6% from the general amount of nitrogen-containing connections, as a result of effective delete of nitrogen in the process of restoring.

4. Summary and Conclusions

It is thus possible to draw conclusion, that the greatest efficiency of sewage treatment, which is evened 97,5%, it was got at a serve in an aerotank aerated (preliminary adapted to the necessary loading) a sludge in an amount, that in 2,6 times exceeds the expense of flow waters on cleaning. In addition, according to the results of the performed work, it can be concluded that the main problem of anaerobic-aerobic purification is the unsatisfactory sedimentation properties of active sludge, which affects the deceleration of the process of sealing and dehydration of sewage sludge. This necessitates the construction of additional sludge sites at the treatment facilities.

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Characteristics of the main components of ecological safety of the Pokutsko-Bukovynian Carpathians

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Abstract

It is noted that in recent years there has been a tendency of deterioration of sanitary and hygienic indicators of water quality in the river basin of the Bukovyna Carpathians. The object of our researches were surface water streams of different functional zones of the National Nature Park "Vizhnitsky" (further - NPP). The NPP is divided into zones of various functional purposes, characterized by a well-developed network of recreational facilities, serving as a place of recreation and health improvement for residents of the region and visitors to the park. The results obtained by us testify to an increase in the value of the indicators of biological and chemical consumption of oxygen from the sources and to the mouth in all of our mountain water streams. In this case, there is also a decrease in the oxygen content and an increase in the value of the oxidation of water. In the mouthpiece there is also a clear seasonal condition of the index of oxidation. This is due to the rather intense pollution of river waters with organic substances, primarily wood waste and household waste of settlements located in their basins.

It has been shown that the epidemiological parameters of open surface water bodies can serve as a reliable criterion for monitoring observations of the state of mountain ecosystems and the level of environmental safety in the region. Among the epidemiological indicators studied, there is a circle-index, coli-titer and microbial number. The given data of microbiological researches, as well as calculations of recreational load carried out by us on the territory of the NPP, indicate certain perspectives for the development of this type of activity. It is shown that in the upper part of the current (protected area of the NPP) the watercourses investigated, in the vast majority, have excellent ecological and hygienic parameters. According to the indicators of the circle index and the total microbial number, the epidemiological indicators of these points do not exceed the existing norms, and in 1,5 times less than the indicators of the recreation zone and more than 2 times - the economic zone. The integrated assessment of the level of pollution of the river network by the epidemiological indicator has shown that the excess of the permissible level of microbial contamination in the economic functional zone of the object of the nature reserve fund is due to the traditional economic activity of the local population living in this territory. In most cases, there is a direct correlation (r = 0,95) between the biological biological and chemical consumption of oxygen and the amount of microbiological indicators.

Based on the state of ecological safety of the Pokutsko-Bukovynian Carpathians, the concept of ecological safety of the mountain region is proposed taking into account the sanitary-ecological and epidemiological indicators. The use of sanitary and hygiene indicators, as reliable elements of the monitoring system, can serve to model and predict the development of mountainous environmentally sensitive areas. The issue of the relationship between the health of the population of mountain inhabitants and the "health of mountain ecosystems" as a whole is discussed.

Keywords

ecological safety, mountain region, Pokutsko-Bukovyna Carpathians, sanitary-ecological condition, epidemiological indicators

1. Introduction

As a result of anthropogenic influence in the mountainous part of the Ukrainian Carpathians in recent years, the threat of violation of the ecological safety of the region has become acute. Despite the fact that for the territory of the Bukovynian Carpathians typical ecological problems are typical for the entire Carpathian region, its specificity is due to the transboundary situation, the peculiarity of climatic conditions, the conduct of traditional farming, etc. Bukovynian Carpathians occupy one third of the territory of the Chernivtsi region. In this mountainous part of the Chernivtsi region, the river Prut forms its basin, Cheremosh, Seret and others begin. rivers Catastrophic floods in 2008 and 2010 indicate violations of the norms and requirements of environmental safety and nature conservation technologies in the Bukovyna Carpathians. As a result of violations of agricultural technologies, in particular forestry, soil erosion and landslide processes have intensified in the region in recent years. Carpathian forests have a special influence on the formation of the hydrological grid and soil cover. In the process of violating the sustainability of mountain forest ecosystems there are significant economic losses, disturbing landscape and biological diversity. The accumulation in the upper mountainous areas of a significant part of the waste wood industry, the placement of livestock forms in close proximity to watercourses leads to deterioration of sanitary and hygienic indicators of water and soil, and is, most likely, a consequence of deteriorating health of residents in the region. The hygienic characteristic of the region serves as an indicator of the sustainable development of the region, in particular the harmonious development of the relations of its social and natural components.

That is why the timely development of the concept of ecological safety for the mountainous part of the Chernivtsi region is urgent.

2. *Materials and Methods*

In order to study sanitary and hygienic indicators of watercourses in the territory of the National Natural Park "Vyzhnytsky" (NPP), we applied the original approach (Gvozdiak PI, 1989) using synthetic fibers of the type "VIA" (TU 995990), made of textured plaid yarn (TU 6-06-C116-87, tex 350). for the accumulation of microorganisms and invertebrate hydrobionts. Water sampling points were selected taking into account the location of household objects and local communities (Fig. 1).

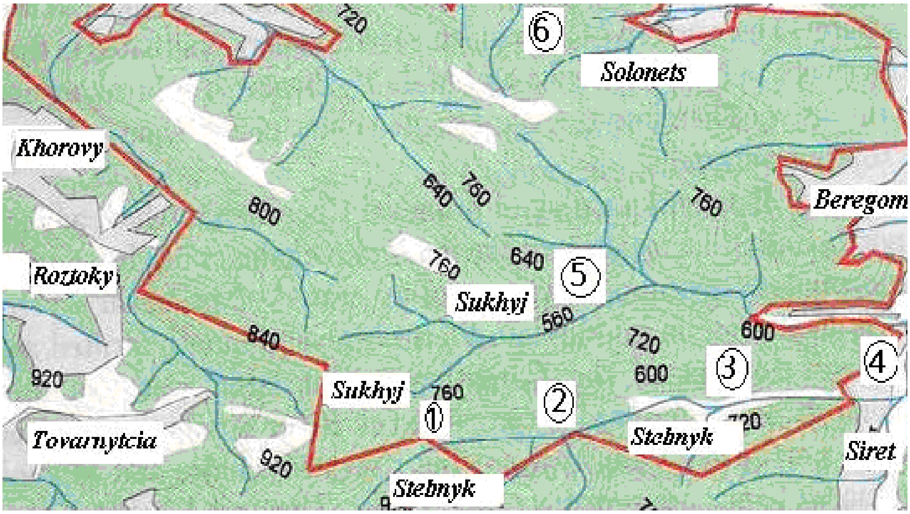

Fig.1. The scheme of sampling water in the territory of the Pokutsko-Bukovynian Carpathians (basin river Siret):
point 1 –upper part of the river Stebnyk, point 2 – midle part of the river Stebnyk (Oikos), point 3 – lower part of the river Stebnyk (hunting house), point 4 – Stebnyk river guidance in the Siret, point 5 - upper part of the river Sukhyj, point 6 - upper part of the river Solonets

In the presented scheme, the sampling points for water under Nos. 2, 3, 5, 6 are located in the zone of stationary and regulated recreation, point 1 - in the reserve area, item 4 - in the economic zone.

Selection, transportation and storage of water samples was carried out according to officially approved methods (DEST 17.1.5.01-80). The total microbial number and the circle-index were determined according to the procedures approved by the Order of the Ministry of Health of Ukraine No. 60 of 03.02.2005. Based on these results, the degree of pollution of surface water by the epidemiological criterion, which is based on the consideration of the danger of microbial contamination, was determined.

The results of analytical studies have been worked out statistically.

Fig. 2. In the time of monting traps of the “Vija” type for the accumulation of microorganisms and hydrobionts

The mounted "traps" were located on special structures "Kashitsah", which are widely used in the Carpathian region for water aeration (Fig. 2).

3. *Results and Discussion*

An important milestone in preserving the mountain ecosystems of the Carpathians was the adoption of the Framework Convention on the Protection and Sustainable Development of the Carpathians (2003). In particular, the Convention noted that during the last decade certain types of human activities arose and rapidly evolving, which led to significant changes in the biological and landscape diversity of the Carpathians, which is the natural habitat of many different species of flora and fauna. That is why the conservation, reproduction and sustainable use of the biological and landscape diversity of the Carpathians for the present and future generations is a matter of urgency and practical significance.

According to Kinal (2006), the Bukovyna Carpathians on the climatic background of the outer (Skybovyh) Carpathians are distinguished not only by the summer-morpho-tectonic features, but also by relatively dry and warmer climates. This is facilitated by the peculiarities of the geographical location, macroclimatic influence and regional morphostructural features. That is why the Bukovinian Carpathians are relatively densely populated and well-developed. This was largely due to the relatively comfortable landscape and climatic conditions. Natural conditions contribute to the formation of valuable biocenoses of forests and mountain meadows. Lowlands relatively recently (200–150 years ago) were everywhere covered with forests. Their intensive cutting in the conditions of a humid climate, high probability of repetition of catastrophic hydroclimatic phenomena, activates the course of unfavorable physical and geographical processes, in particular, plane flushing, erosion, landslides, gusts and the like. Accordingly, due to the manifestation of such phenomena, the mountainous landscapes will experience negative influence. Investigation of their properties, in particular climatic ones, has a scientific, cognitive and practical significance in the aspect of developing the scientific basis of rational nature management, conservation works, nature conservation work within mountain regions

According to the data (Hetman, 2001; Prikhodko and Prikhodko, 2011; Furdichko, 2002), the past 300 years, as a result of the anthropogenic factor, the area of forest geosystems in the Ukrainian Carpathians has intensively decreased (from 95% to 36%). On the site of natural beech and fir-beech forests on an area of 185 thousand hectares are created derivatives of spruce monocultures, which have low stability and now intensively dry. Excessive exploitation of forests, replacement of natural forests (virgin forests) with anthropogenically-modified, change in the age structure of forest stands led to a significant loss of the ecological potential of forest geosystems, degradation of water transformation and other functions, and lowering their environmental safety.

Bukovyna Carpathians are the richest biodiversity of the natural and territorial complex, which significantly affects the ecological status of the surrounding areas. The Framework Convention on the Protection and Sustainable Development of the Carpathians (hereinafter referred to as the Carpathian Convention) provides for the creation of a number of protected areas in the Carpathians, the development of the ecological network as a structural part of the pan-European ecological network, which guarantees the preservation of unique and typical natural landscapes as a basic element of ecologically balanced socio-economic development. Carpathian region (Solodky, Masikevich, etc. 2012; Solodkyj, Bezpalko et al., 2013).

Bukovyna Carpathians - first of all the region of the all-Ukrainian and European health resorts. The tourist and recreational potential of the Bukovynian Carpathians is significant, but until recently it was used inefficiently and not fully. NPP and adjacent unique territories will become the pivotal recreational axis of the region (Movchan, Shelyag-Sosonko, 2003; Golubets, 2007). Among the forest vegetation groups and mountain valleys, the types of vascular flora that are listed in the Red Data Book of Ukraine are growing, among them: Leontopodium alpinum (Cass.), Arnika Montana (L.), Gentiana acaulis (L.), Lilium martagon (L.), Carlina onopordifolia (Besser ex Szafer), Astrantia major (L.), Botrychium lunaria (L.), Leucorchis albida (L.), Listera ovata (L.), etc. The total number of species of flora subject to state protection, along with the species listed in the European Red List, is 54, which is almost half of the "Red Book" species of the Ukrainian Carpathians. This territory acquires a significant environmental value due to the high degree of saturation of its flora with endemic species as the Carpathian and East Carpathian forests, with almost 40 forests recorded here. There is also a rich fauna of the Bukovynian mountain forests. In this territory there are 28 species that are listed in the Red Data Book of Ukraine and 14 species from the European Red List. Among them: Ursus arctos (L.), Mustela erminea (L.), Meles (L.), Felis silvestris (Schreber), Lynx (L.), Sorex alpinus (Schinz), Circaetus gallicus (Gm.), Aquila chrysaetos (L.), Tetrao urogallus (L.), Aegolius funereus (L.) and many others. The territory of the NPP is also unique in that there are 70 species Cervus elaphus montanus (L.).

Kifiak (2015) without exaggeration believes that tourism development is one of the scenarios for improving the socio-economic status of the Chernivtsi region, convinces with the need to develop a strategy for tourism development in the region and the city of Chernivtsi by 2020 and to ensure the formation of an appropriate regional policy. Among the directions of such a policy, the author believes that the following should be done: inventory of sanatorium and resort establishments, tourist complexes, enterprises of accommodation and restaurant business, farmsteads of rural tourism and objects of tourist exhibition in the region, establishment of their condition and terms of restoration if necessary; definition and reservation of perspective sites for the construction of tourism objects. Important importance is given to the list of investment-attractive. Objects and dissemination of information about them; to promote the granting of the status of a sanatorium and resort area of national significance and the status of regional significance to the recreational territories of Vyzhnytsky, Putil, Storozhynets districts and other territorial-administrative units. Considerable attention is paid to the diversification of economic activity in the region and on this basis the establishment of the real impact of recreation and tourism on economic development, to determine the economic and social importance of the tourism industry in the region, and to develop a strategy for tourism development. However, it should be noted that the author (by the way, as well as many other economists) completely does not analyze the impact of tourism and recreation on the environment and its ecological component. And, even citing the classic of the environment of Reimers (1999), the author is limited only to the value of tourism and no more.

In some cases, not so much the impact of tourism on the environment is analyzed, on the contrary, the impact of environmental factors on tourism objects. For example, Pisareva (2014) believes that ecological factors in the form of global problems of the biosphere are one of the most important problems in protecting cultural and historical tourist resources, as there is a permanent destruction of the last acid rain, and as a result, it leads to economic losses for the tourism industry.

There are also thoughts (Arkhipova, 2014; Yakov, 2017) that the development of any kind of tourism brings harm to the environmental state of the environment. In order not to slow down the development of tourism and not to destroy nature, there should be restrictions on the number of tourist complexes in a certain territory, and the improvement of already existing ones.

According to Korabainikov (2013), the main impacts of tourism on the environment are primarily manifested in changes in primary landscapes, water pollution, air pollution. A pressing issue is the collection and treatment of waste in the tourist destinations. Therefore, the main directions of innovative activity in the field of tourism should be technological innovations, which will reduce the environmental impact of tourist destinations of tourist infrastructure and tourists.

An interesting approach to ecologization of tourism in terms of the development of noosphere consciousness and thinking offers Vorobyov (2009, 2012). According to the author, the turn to the noosphere civilization can begin only with radical changes in the spiritual life, from the formation, according to which a person should be able to fit into biosphere processes so that not only to preserve the biosphere in all its diversity, but also to ensure the spiritual growth of mankind, humanization of interpersonal and interethnic relations, preservation and development of cultural and ethnic diversity of the world. The development of ecological tourism, as a kind of balanced (sustainable) nature use in nature-protected territories, can and should contribute significantly to the formation of the noosphere consciousness, or thinking, by establishing harmony between man and nature.

According to Rusev (2006), ecological tourism, provided that infrastructure, service and advertising are properly developed, has rather good prospects in the nature reserves.

This point of view is developed in his research by Oliynyk, Getman (2002) when analyzing the issue of tourism development in the territories of national parks of Ukraine. According to the authors, the current state of development of the tourist sphere of national parks of Ukraine is relevant for the introduction of elements of sustainable (balanced) tourism on their territory. For parks with a dominant recreational and tourist function, the implementation of a sustainable development strategy involves the modernization of recreation and health facilities, the development of schemes for the dispersal of tourist flows and the more comprehensive introduction of forms of ecotourism, especially cognitive.

In our opinion, the advantages and disadvantages of ecotourism development in the Carpathian region in the scientific work of Pankov (2008) are sufficiently fully analyzed.

The author considers the main disadvantages of ecotourism: the lack of a single system of marking ecosystems, the poor quality of equipment for stops on eco-tags, the inaccessibility of instructors, the lack of comprehensive information on the possibilities of ecotourism in the region, violation of the rules of stay in protected areas, alienation of territories of national parks, which leads to shortening proposals from ecotourism.

The development of tourism and recreation is closely linked with the conduct of traditional activities by residents of the region. Ecologically clean products of livestock and crop production (milk, cheese, meat, mushrooms, berries) attract numerous groups of tourists. In this connection, in the last decades, the "green tourism" appears and is actively developing in the Pokutsko-Bukovynian Carpathians. According to a number of authors (School and others, 2007) Bukovina is an agrarian region: more than half of the land of Bukovyna region has agricultural purpose, more than 29% of the population are employed in agriculture, hunting, forestry and fisheries. Today in the Chernivtsi region there is an annual increase in the volume of foreign and domestic tourism. In the region there is a deterioration of the demographic situation, the process of negative natural population growth continues. The population of the region actively departs for earnings abroad (official data on the number of workers are absent) in search of work and high earnings. Mountains create specific, extremely difficult living and farming conditions. Therefore, the problem of employment of mountain areas is acute, and as a result, the low level of material well-being of the inhabitants of the mountains. Stabilizing these processes by expanding employment, infrastructure development will be facilitated by ecological tourism with the appropriate service and organization of service, which will involve the local population. In this context, alternatives to ecotourism simply do not exist.

According to Mishchenko (2010), today quite often is the identification of the concepts of ecotourism and environmental friendliness in tourism. It is clear that tourism affects the environment, changing the component structure of the landscape. One should agree with the author's opinion that the notion of environmental friendliness in tourism includes measures to protect the landscape and relates to each type of tourism.

An important element of the ecological safety of mountain areas can be the use of organic farming. Among organic farming areas are organic production of medicines (Sologub, 2017). It is the "green pharmacy" of the Carpathians that can serve as an important means of public health and provide a direction for economic enrichment. Organic production of medicinal plant growing is widely represented in the Bukovynian Carpathians: peppermint, chamomile, medicinal herbs, plantain large, dog nettles, medicinal sage, herring, thyme, etc. Organic medicinal plant growing, as noted by Sologub (2017), combines the multi-purpose functions: 1. Economic (economic efficiency, orientation to the domestic consumer, sustainable economic effect); 2. Ecological (conservation of biodiversity, natural ecosystem functioning, sustainable development); 3. Social (rural development, ecological consciousness of the population, satisfaction of local needs, use of own labor resources).

The object of our research was the various functional zones of the National Nature Park "Vyzhnytsky" (further - NPP). The NPP is characterized by a well-developed network of recreational facilities, serving as a place of recreation and recreation for the residents of the region and visitors to the park. During the year, the park is visited by 15 to 20 thousand adults and children. Today, the recreational network of the park is represented by two recreational points ("Luzhky" and "Velyka Rozchyshch") with a wide range of recreational infrastructure and 19 recreational areas in the tracts of Stebnik, Sukhyj, Solonets, Small Styzhok and others. This activity is regulated by the order of the Ministry for Environmental Protection of Ukraine dated March 14, 2011, No. 76. The tourist-recreational sphere is an important economic component of the development of the NPP. The experience of European countries shows that the regions where national parks are developing more actively. That is why considerable attention is paid to the development of recreational infrastructure to create some comfortable conditions for holidaymakers.

A good example of the creation of proper conditions for organized rest in the natural environment with the adherence to the regime of protection of protected complexes is the summer health camp for children and youth of the Chernivtsi City Council "Oikos", which was established in 2005 in the zone of stationary recreation NPP. In the camp since the creation were able to improve and rest about 5 thousand students. It should be noted that any recreation area is easily accessible due to the good access roads, recreational areas equipped with rest rooms, caretaking, regular garbage removal, etc. At the same time, it should be noted that the issue of constant control over the quality of the environment remains relevant. First and foremost, sanitary and hygienic indicators of water, air, and soil. Our research on the sanitary-ecological state of the territory of recreation zones (points 2, 3, 6) of the NPP "Vizhnitsky" (Fig. 3, Table 1) shows that for sanitary-hygienic indicators, surface waters of the I-II class pollution occurring on in these territories, according to SanPiN 4630-88, belong to the category of "permissible" contamination (<1 x 10^4 lactose-positive coliforms in 1 dm^3). Downstream (point 4) there is a contamination of watercourses with flushing of organic nature from coastal and water protection zones, which are decomposed using dissolved oxygen in water, as evidenced by a sharp decrease in the dissolved oxygen content in the investigated samples. This tendency is seasonal.

The degree of indigenization of water by organisms is taken to express the saprobity, which is understood as the totality of living beings living in waters with a large accumulation of animals or plant remains. Our previous studies (Masikevych et al., 2016) show that the water sources analyzed by us belong to the mesosample zone, where the mineralization of organic matter with intense oxidation and pronounced nitrification takes place. The given data of microbiological researches, as well as calculations of recreational load carried out by us on the territory of the NPP, indicate certain perspectives for the development of this type of activity.

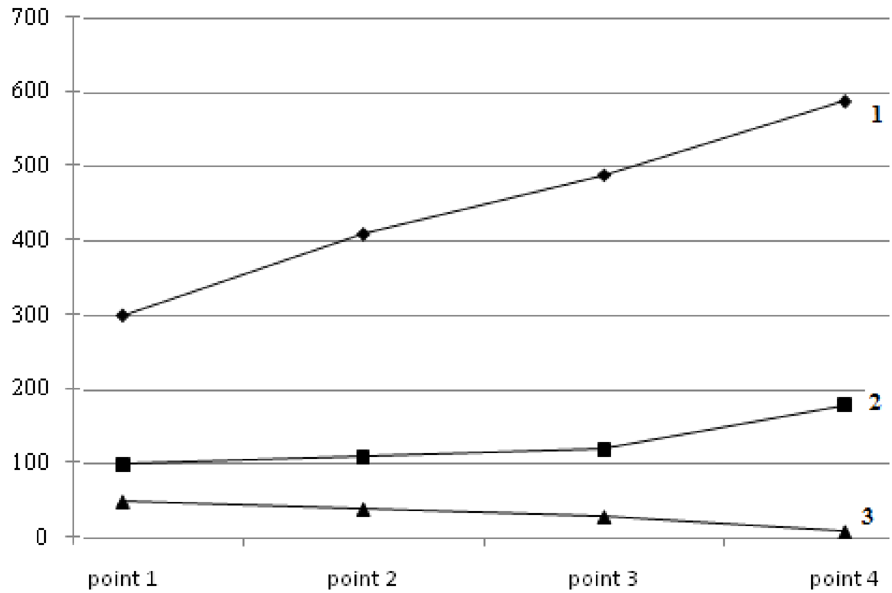

Fig. 3. The main microbiological indicators of open water sources on the trritory of the Pokutsko-Bucovynian Carpathians:
1 – total microbial number, 2 – coli-index, coli-titer, point 1 –upper part of the river Stebnyk, point 2 – midle part of the river Stebnyk (Oikos), point 3 – lower part of the river Stebnyk (hunting house), point 4 – Stebnyk river guidance in the Siret river

For comparison, we give the research data obtained at the points of sampling water 1 and 5, located in the reserve area of the NPP. In the upper part of the flow, the studied watercourses have, in the vast majority, excellent ecological and hygienic indices. According to the indicators of the circle index and the total microbial number, the epidemiological indicators of these points do not exceed the existing norms, and in 1,5 times less than the indicators of the recreation zone and more than 2 times - the economic zone. A comparative analysis of the results of our studies showed (Table 1, Table 2), that in the summer, there is a rise in the value of microbiological parameters under study. The reason for the sharp increase in the size of the microbial number in the summer (subject to the provision of a temperature optimum) can be explained by floods, which lead to a sharp increase in surface runoff and demolish impurities from mountain slopes and slopes, as well as partially fecal contaminations. The sharp change in the value of the indicator of the circle-index during the summer-autumn season testifies to the growth of pathogenic microflora in surface waters, which can be the cause for infectious and non-infectious diseases of the population of the region. The integrated assessment of the level of pollution of the river network by the epidemiological indicator has shown that the excess of the permissible level of microbial contamination in the economic functional zone of the object of the nature reserve fund is due to the traditional economic activity of the local population living in this territory.

Table 1. Main sanitary-microbiological indicators of water of open water sources on the territory of the Pokutsko-Bukovynian Carpathians (month of July, water temperature 19 ° C)

sampling points	**Sanitary-microbiological indicators**							
	coli-index			**coli-titer**			microbial number	pH
	warm-bloodeE.coli	amphi-bians E.coli	general	warm-bloode E.coli	amphi-bians E.coli	general		
1	26,0	49,0	75,0	38,5	20,4	13,3	2104,3	7,3
2	34,0	60,0	94,0	29,4	16,6	10,6	3020,7	7,4
3	39,0	78,0	117,0	25,6	12,8	8,6	3597,6	7,5
4	61,5	105,0	166,0	16,4	9,5	6,0	4191,6	7,5
5	16,0	34,0	50,0	62,5	29,4	20,0	1425,5	7,3
6	37,0	65,0	102,0	27,3	15,4	9,8	2902,0	7,4

- the tables are based on the data of three years of research (2015–2017), the research data is reliable for $p < 0,05$;
- point 1 - the upper part of the river Stebnyk;
- point 2 - the middle part of the river Stebnik (Oikos);
- point 3 - the lower part of the river Stebnyk ("Hunting House");
- point 4 - the fall of the Stebnik River in the Siret River;
- point 5 - the upper part of the river Sukhyj river;
- point 6 - the upper part of the river Solonets.

In accordance with existing international requirements, there is a need for the creation of environmental passports for the objects and territories of the natural reserve fund. An important criterion for such passports, as Mudrak (2014) notes, should be precisely the sanitary-ecological assessment of the status of protected areas. Our studies in this regard indicate an increase in the values of the indicators of biological and chemical oxygen consumption (BOC, ChOC) from the sources and to the mouth of the specified watercourses. It should be noted that in recent years there has been a tendency for deterioration of sanitary and hygienic indicators of water quality in the river basin of the Bukovyna Carpathians. The results obtained by us indicate the increase in the value of the parameters of the BOC from the sources and to the mouth of the watercourses we investigated. Downstream is the contamination of river waters with natural fl ows from the coastal and water protection zone, which decompose using oxygen dissolved in water (Fig. 4).

The results obtained by us testify to an increase in the value of the parameters of the BOC from the sources and to the mouth in all of our mountain water streams. In this case, there is also a decrease in the oxygen content and an increase in the value of the oxidation of water. Increased oxidation in rivers is a direct indicator of its pollution. If the upper part of the flow of watercourses, the oxidation was an average of 1,2 mg/dm^3 (at a rate of 5–6 mg/dm^3), in the middle part of this indicator increases to 9,3 mg/dm^3, and in - the mouth part it reaches 12,4 mg/dm^3. In the mouthpiece there is also a clear seasonal condition of the index of oxidation. This is due to the rather intense pollution of river waters with organic substances, primarily wood waste and household discharges of settlements located in their basins. At the same time, we conducted a study of the microbiological state of water of the above-mentioned objects of research. Among the investigated indicators is the circle-index, the circle-titer and the microbial number. In most cases, there is a direct correlation (r = 0,95) between the values of biological BOC, ChOC and the magnitude of microbiological parameters.

In order to concentrate microbial organisms that are in open water, we used synthetic fibrous material of the type "Vija".

The results obtained are presented in tables 2–3. In sufficiently clean areas of watercourses (points 1, 2, 3.5, 6) there is an active accumulation (almost 10 times) of bacteria on a fibrous material. Somewhat less adsorbed bacteria in more polluted parts of reservoirs (point 4).

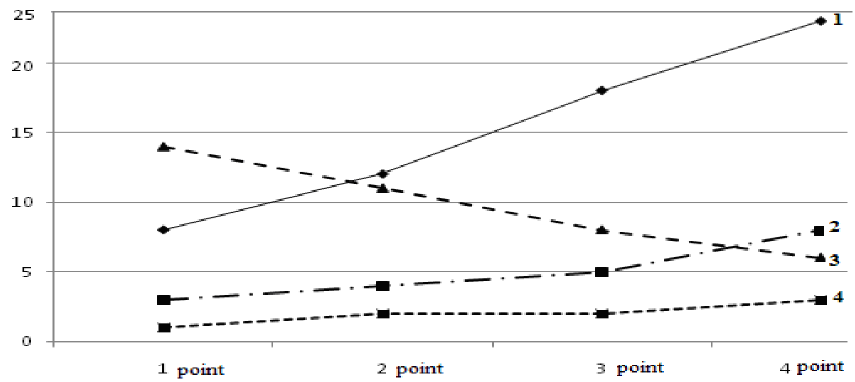

Fig. 4. Sanitary and microbiological parameters of different water intake points in the territory of the Pokutsko-Bucovinian Carpathians:1 - chemical oxygen consumption (COC), 2 - biological oxygen consumption (BOC), 3 – coli-titer, 4 - nitrites

Table 2. Comparative sanitary-microbiological parameters of flow water and the cumulative capacity of a synthetic sorbent (type "Vija") for open water sources in the territory of the Pokutsko-Bukovynian (August, water temperature + 23° C)

sampling point		**Sanitary-microbiological indicators**							
	sampling material	pH	**coli-index**			**coli-titer**			microbial number
			warm-bloode E.coli	amphi-bians E.coli	general	warm-bloode E.coli	amphi-bians E.coli	general	
1	water	7,3	26,1	55,9	82,0	38,3	17,9	17,8	2307,0
	1 g of sorbent		-	-	2,1 x 10^6	-	-	4,7 x10^{-6}	31216,0
2	water	7,4	43,0	71,9	114,9	23,3	13,9	18,6	3617,0
	1 g of sorbent		-	-	1,9 x 10^6	-	-	5,2 x10^{-6}	33168,0
3	water	7,6	74,0	119,0	193,0	13,5	8,4	11,0	4309,0
	1 g of sorbent		-	-	3,1 x 10^6	-	-	3,3 x10^{-6}	47286,0
4	water	7,6	47,0	96,0	143,0	21,3	10,4	19,1	38098,0
	1 g of sorbent		-	-	2,4 x 10^6	-	-	4,1 x10^{-6}	39132,0
5	water	7,3	19,0	36,0	55,0	52,6	27,8	40,2	1596,0
	1 g of sorbent		-	-	1,3 x 10^6	-	-	7,9 x10^{-6}	17216,0
6	water	7,4	47,0	29,0	126,0	21,3	12,7	17,0	3116,0
	1 g of sorbent		-	-	2,4 x 10^6	-	-	4,2 x10^{-6}	36148,0

the tables are based on the data of three years of research (2015–2017), the research data is reliable for $p < 0,05$; points for sampling of analogues, in tables 1.

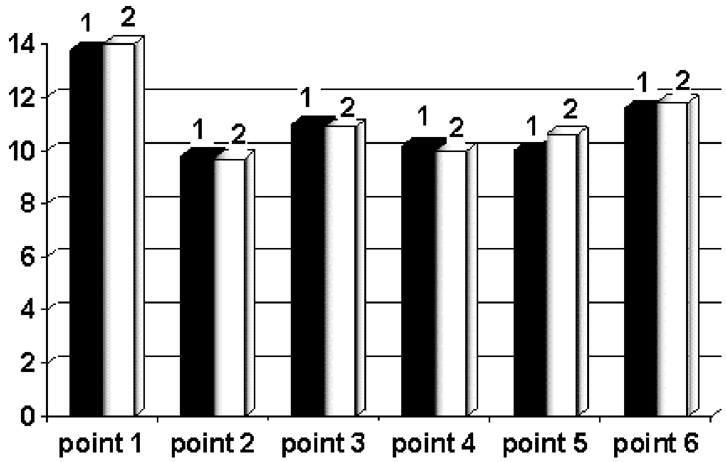

Fig. 5. Coefficient of absorption of bacteria by synthetic meterial type “Vija”:
1 – tempereture 23 °C, 2 - tempereture 6 °C

Studies have also shown that the seasonal variation of water temperature in the rivers does not have a significant impact on the process of bacterial accumulation on a fibrous material of the type "Vija" (Fig. 5).

Based on the comprehensive research carried out, the main elements defining the ecological safety of the mountainous region (Fig. 6) suggest a concept of balanced development of these territories. As shown in the diagram. An important role in ensuring the technological and environmental safety of the region, along with the rational use of natural resources, belongs to the sanitary and ecological state of the environment, which ultimately determines the population's health of the mountain riders. In this case, the importance of traditional management of economic activity, which serves as the material basis of development in the region of the recreational and tourist industry, primarily the so-called "green tourism". An ecosystem approach to the development of mountain areas can ensure the preservation of the landscape and biodiversity of the region, create conditions for the development of traditional types of farming, become the basis for the safe and healthy existence of the population. The use of sanitary and hygiene indicators, as reliable elements of the monitoring system, can serve to model and predict the development of mountainous environmentally sensitive areas.

Table 3. Comparative sanitary-microbiological parameters of flow water and the cumulative capacity of a synthetic sorbent (type "Vija") for open water sources in the territory of the Pokutsko-Bukovynian Carpathians (October, water temperature + 6 ° C)

sampling points	**sampling material**	**Sanitary-microbiological indicators**							
		pH	**coli-index**			**coli-titer**			microbial number
			warm-bloode *E.coli*	amphi-bians *E.coli*	general	warm-bloode *E.coli*	amphi-bians *E.coli*	general	
1	water	7,3	22,6	52,3	74,9	44,3	19,1	13,4	2193,0
	1 g of sorbent		-	-	995,8	-	-	1,0	29126,0
2	water	7,4	35,5	68,7	104,2	28,2	14,6	9,6	3426,0
	1 g of sorbent		-	-	970,2	-	-	1,0	31916,0
3	water	7,5	55,7	82,7	138,4	17,9	12,0	7,2	4137,0
	1 g of sorbent		-	-	1512,5	-	-	0,7	45193,0
4	water	7,5	36,6	75,9	112,4	27,4	13,2	8,9	3596,0
	1 g of sorbent		-	-	1163,6	-	-	0,9	37216,0
5	water	7,3	16,2	35,76	51,9	61,8	28,0	19,3	1391,0
	1 g of sorbent		-	-	534,2	-	-	1,9	14319,1
6	water	7,4	34,3	69,6	103,9	29,2	14,4	9,6	2879,0
	1 g of sorbent		-	-	1199,0	-	-	0,8	33216,0

the tables are based on the data of three years of research (2015–2017), the research data is reliable for $p < 0{,}05$; points for sampling of analogues, in tables 1.

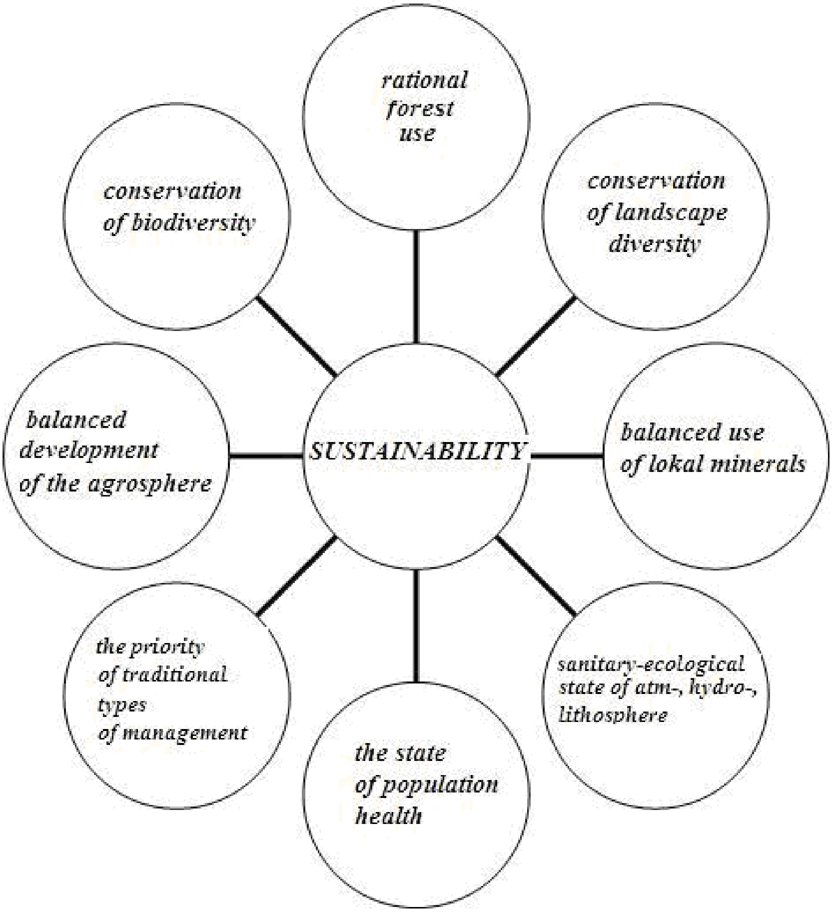

Fig. 6. The main elements of the ecological safety of the mountain ecosystem of the Pokutsko-Bucovynian Carpathians

4. Summary and Conclusions

Established that:

- in recent years there has been a tendency for deterioration of sanitary and hygienic indicators of water quality in the river basin of the Bukovyna Carpathians;
- In the upper part of the current course (the protected area of the National Park), the index of the circle and the total microbial number do not exceed the existing norms;
- for sanitary-and-hygienic indicators, surface waters in the territories of the recreational areas of the national natural park, in accordance with SanPiN 4630-88, are classified as "permissible" level of pollution;

- exceeding the permissible level of microbial contamination in the economic functional
- zone, the object of the nature reserve fund, due to the traditional economic activities of the local population living in this territory;
- in most cases there is a direct correlation (r = 0,95) between the indicators oxygen consumption and microbiological values;
- water sources analyzed by us refer to the mesosaprob zone where mineralization of organic matter with intense oxidation and pronounced nitrification takes place;
- the ability of the synthetic fibrous material of the "Vija" type to adsorb microbial organisms, can be used for monitoring studies of the sanitary and ecological state of surface waters; the seasonal fluctuation of water temperature in rivers does not have a significant impact on the process of bacterial accumulation on a fibrous material of the "Vija" type.

Our studies have shown that the ecological and hygienic characteristics of the river network of the region can serve as one of the important indicators of changes in the ecosystem, can be used successfully for the certification of objects of the nature reserve fund and further development of the recreation area in these territories. Based on the comprehensive research carried out, the main elements defining the ecological safety of the mountainous region, the concept of balanced development of territories of the nature reserve fund is proposed**.**

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Environmental Risk of Surface Water Resources Degradation

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Abstract

Water is a natural resource and an integral part of the existence of all living creatures on the planet. The problem of human supplying beings with drinking water is extremely important nowadays, since the available water resources in many areas are not sufficient to provide all consumers not only in the future but also today. Objective is to carry out a qualitative and quantitative assessment of the ecological status and to determine the level of environmental risk of surface water resources in the Mykolaiv region on the basis of the appropriate methods. Ecological quality classification of surface waters and estuaries of Ukraine is built on ecosystem principle. The necessary completeness and objectivity of the characteristics of surface water quality are achieved a fairly wide set of indicators that reflect the characteristics of abiotic and biotic components of aquatic ecosystems. The base of the study is selected organoleptic and sanitary-toxicological indices of water quality, because its most fully reflect the ecological condition of water resources. Assessment of ecological status according to the organoleptic properties of water provides for evaluation in terms of color, pH index and suspended solids. Based on sanitary-toxicology data includes the assessment of COD and, nitrates (NO_3^-), total hardness, chlorides, sulphates, phosphates, total ferrum and manganese. The assessment of the status of water in terms of ecological risk coincides with the assessment of environmental quality. Individual points are class II quality "good". Quite often water is "unsatisfactory", class 4 quality. 5 class of water quality is "bad", separately found almost for each item of observations, due to excessive concentrations as a result of an anthropogenic impact on water objects. This situation indicates that water bodies in the study area have somewhat disturbed the ecological parameters of their ecological status is estimated as "ecological regression". The research presented that water objects of the area are unsuitable for drinking water supply. The ecological condition is characterized by ecological imbalance due to excess concentrations of pH, suspended solids, color, COD, BOD_5, total hardness, sulfates, chlorides, total iron and manganese.

Key words

surface water pollution, the maximum permissible concentration, environmental risk

1. Introduction

Ecological problems of today create a danger to human existence at all levels – from local to global. Particularly acute, these problems are acquired in areas that are experiencing significant anthropogenic pressures. Currently, the problems of environmental assessment of water resources are particularly relevant.

Water is a natural resource and an integral part of the existence of all living things on the planet. The problem of human supplying beings with drinking water is extremely important nowadays, since the available water resources in many areas are not sufficient to provide all consumers not only in the future but also today.

In addition, it is necessary to take into account also the fact that Ukraine is one of the least secured in Europe for fresh water supplies. So, for one inhabitant, there are only 1 thousand m^3 of water, while in Sweden and Germany – 2,5 thousand m^3, France – 3,5 m^3, in the UK – 5 m^3. Almost 1,300 settlements in Ukraine live on imported water, which are almost 1 million inhabitants. The greatest shortage of water is on the steppe areas of Donbass, Polissia, Podillia and Crimea. Meanwhile, the volume of water consumption in Ukraine over the past 20 years has doubled. All these sources, according to official data, are polluted. Virtually no water surfaces that meet to the first class of hygienic requirements and this means that there is no qualitative drinking water in Ukraine (Klymenko, et al., 2006).

The part of water that suitable for use by the population and industry is very limited. Quantitative and qualitative composition of water have experienced by anthropogenic impact. Indicators of the qualitative composition of water are one of the determining factors during assessing of environmental situation.

The most promising method for identifying areas of high environmental hazard is the assessment of the environmental status. This allows determining the permissible anthropogenic pressure in order to save equilibrium in the natural environment.

Environmental assessment of surface water quality is the basis for establishing ecological standards for water quality, both for individual water bodies and their parts, groups of water bodies and river basins. It is also the basis for environmental risk management of anthropogenic pressure on environmental objects.

For the Mykolaiv region of Ukraine the problem of water resources pollution due to wastewater discharges is significant, which is greatly hampered by the lack of centralized drainage networks and qualitative cleaning of domestic and industrial discharges (Pohrebennyk et al., 2017).

It is important to do the monitoring the water status in the direction of the state national policy in the field of improving the quality and efficiency of water resources management. The scientific substantiation of carrying out water protection measures, development of the further strategy of using water resources for the purpose of improvement of the management of the Southern Bug river basin are urgent (Rybalova, 2011)].

Objective: to carry out a qualitative and quantitative assessment of the ecological status and to determine the level of environmental risk of surface water resources in the Mykolaiv region on the basis of the appropriate methods.

The formulated purpose is realized in the research by solving of the following **tasks**:

- assessment and characterization of the water resources state in Mykolaiv region on the basis of literary sources;
- study of legislative normative acts about water resources management;
- determination and assessment of the water resources quality on the basis of hydrochemical information;
- on the basis of environmental assessment of the water resources status justification the relevant conclusions and proposals.

2. Materials and Methods

The total area occupied by surface water objects of the Mykolaiv region is 150,5 thousand hectares, which is 6,1% of its territory. In the area, there are 121 rivers, channels and 26 lakes, 45 reservoirs, 1153 streams, 7 swamps and estuaries. Water resources of the region are very limited and depend mainly on inflows from other regions.

Governance has registered 47water users that discharge wastewater into surface water bodies. In the course of 2015, the total wastewater discharge amounted to 105,2 million m^3, of which 24,3% (25,55 million m^3) – contaminated wastewater (Mykolaiv, 2017).

Since 2010, on the territory of the Mykolaiv region not recorded discharges of wastewater without treatment (emergency wastewater discharges). Large wastewater discharges without any treatment were observed from 1995 to 2003, discharges of insufficiently treated wastewater are annually decreasing in comparison with 1991 when discharge was equal to 69,4 million m^3 (Mykolaiv, 2016).

The discharge of industrial waters into surface water objects is carried out by the energy companies and the engineering industry. In these discharges include heat exchangers waters that the composition is classified as normatively clean without treatment. The discharge of normatively clean without treatment of wastewater from said water user in 2016 amounted to 22,20 million m^3, which in comparison with the corresponding volume of water discharged in 2015, more to 1,2 million or 1,05% (Mykolaiv, 2016).

The greatest discharge of normatively clean without treatment of wastewater in the Mykolaiv region is made by South Ukraine Nuclear Power Plant which includes

Alexandrivka hydroelectric power station and Tashlyk pumped storage power plant. The volume of discharge specified enterprise is located 47, 4% of the total volume of wastewater discharges in the region, and is 39, 76 million m^3. Exceeding the established norms of wastewater discharges is undertaken by enterprises and public utilities.

Dynamics of discharged wastewater to surface waters in the region is on the fig. 1.

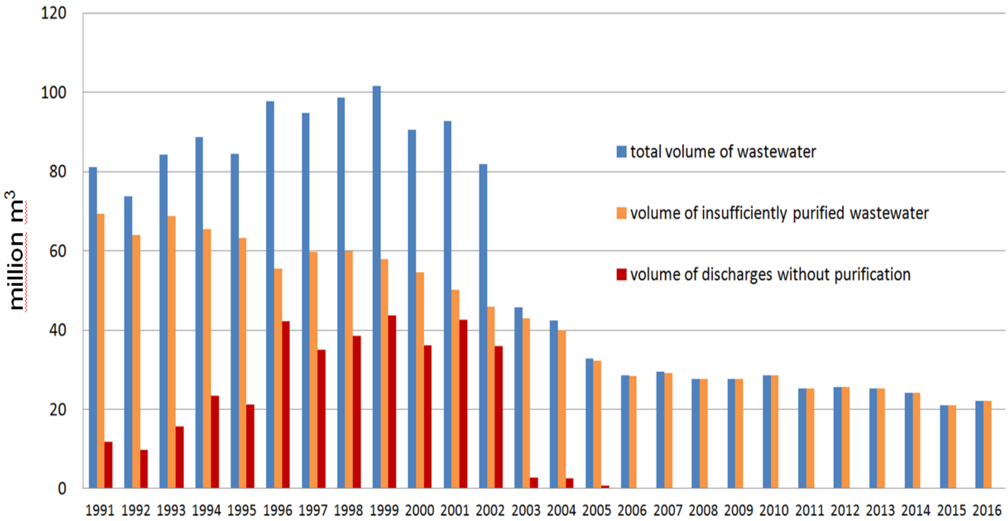

Fig. 1. Dynamics of discharged wastewater to surface waters in the region, million m^3

A marked decrease of the water amount discharge is connected with its reduction of enterprises and population. Water has become a very expensive resource. Businesses and people become more frugal with water.

For inefficient cleaning of sewer drains of Mykolaiv, for the past eight years "Mykolaivvodokanal" is the main polluter of water resources in the region. Over the past year to the water bodies dropped 26, 23 million m^3 of wastewater, of which insufficiently treated – 23,83 million m^3. 91% of the total number of reset mentioned utilities are contaminated with sewage, which in its own turn, affects the state of water resources (Mykolaiv, 2016, 2017).

The principle of quality management of the environment is currently expected the requirement to ensure hygienic standards of maximum permissible concentrations (MPC) of polluting substances in natural components (air, water, soil) and physical factors (noise, vibration and the like). Therefore, according to conservation methodology, the assessment degree of pollution of the environment is performed by comparing the concentration of the contaminant with the MPC. However, the hygienic standards are inherent in the anthropocentric approach to the assessment of the environmental state, i.e., under safe conditions of living are not taken into account peculiarities of functioning in the actual ecosystems (Rybalova, 2011).

Actual is the need to develop more comprehensive criteria for assessing the quality of the environment.

According to the Water code of Ukraine (№ 213/95-VR, 1995) assessment of water quality is based on regulations of environmental safety of water use and environmental standards of water quality of water bodies.

It is established that now the total number of methods of evaluation and water quality classifications is sufficiently large, but none of them is widely used in water protection practice, because it does not take into account the integral display quality of water, that is, the total effect of hydrophysical, hydrochemical and other data (Bharti and Katyal, 2011; Cordoba et al., 2010; Holodenko and Kosyak, 2014; Ridey and Zaharkevich, 2008; Yidana and Yidana, 2010).

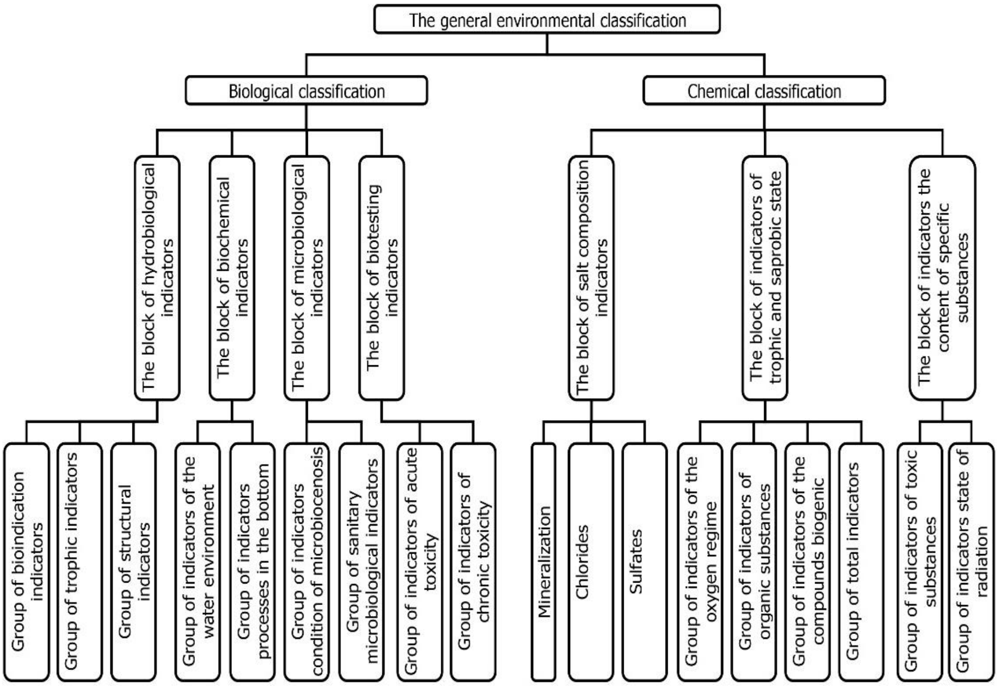

Fig. 2. Structure of the general environmental classification of surface waters

The system of environmental quality classification of surface water and estuaries of Ukraine has two subordinate classifications, namely: on biological parameters and physical and chemical and chemical indicators. The complex of indicators of environmental quality classification of surface water includes general and specific indicators. The general indicators characterize salt composition and trophic and saprobic of waters (ecological and sanitary), that are inherent to water ecosystems ingredients, whose concentration can vary under the influence of economic activity (Holodenko and Kosyak, 2014). Specific indicators characterize the content in water pollutants and toxic radiation exposure (fig. 2) (Mykolaiv, 2017).

But analysis of biomonitoring of surface waters of Ukraine shows that this component of the state monitoring system of the environment is in poor condition. This is reflected in the limited number of points observations, the practical absence of the expedition survey of water bodies of the country, low use of results of biological monitoring of water quality in water protection practices.

The specified demonstrates the need for a complex of works on improvement of biomonitoring of the country, above all to improve its efficiency and harmonization with similar systems in developed countries (Ridey and Zaharkevich, 2008).

On the basis of "Methods of environmental quality assessment of surface waters by corresponding categories" (Alymov and Tarasov, 2005; Mykolaiv, 2016), we developed a more complete and more accessible methodology that includes the definition of environmental assessment of surface water quality and ecological risk for the water bodies.

Ecological quality classification of surface waters and estuaries of Ukraine is built on ecosystem principle. The necessary completeness and objectivity of the characteristics of surface water quality are achieved a fairly wide set of indicators that reflect the characteristics of abortion and biotic components of aquatic ecosystems (Mykolaiv, 2016).

A complex of indicators of ecological quality classification of surface waters includes the General and specific indicators. The overall performance, which includes indices of the salt composition, trophy-saprobes waters (ecological and health), which characterize usual water ecosystems, the ingredients, the concentration of which can vary under the influence of economic activity. Specific indicators characterize the content in water of polluting substances of toxic and radiation action.

3. Results and Discussion

The assessment of the water quality and the ecological state of Mykolayiv region surface waters (namely, Southen Bug, Ingul, Mertvovid and Synyukha) was performed in the period from 1990 to present for 13 sections, in which the specialists of the Southen Bug Basin Water Resources Department perform monitoring.

The base of the study is selected organoleptic and sanitary-toxicological indices of water quality, because its most fully reflect the ecological condition of water resources. Assessment of ecological status according to the organoleptic properties of water provides for evaluation in terms of color, pH index and suspended solids. Based on sanitary-toxicology data includes the assessment of COD and, nitrates

(NO_3^-), total hardness, chlorides, sulphates, phosphates, total ferrum and manganese (Mykolaiv, 2016).

Environmental indices and categories of water quality are calculated using the functions of the software package MS Excel and are the average values at each point of the selection in fig. 3.

Risk for water body was determined by the formulas 1-3 (Alymov and Tarasov, 2005; Gritsenko et al., 2012) and represented on fig. 4.

$$R = -\ln(P), \tag{1}$$

where:

$$P = \frac{\sum n_i}{N}, \tag{2}$$

where:

$$\sum n_i = \sum \frac{C_i}{MPS}, \tag{3}$$

where: C_i is the concentration of the pollutant substances (PS),
MPC (PS that do not exceed the MPC in the formula (3) is substituted);
N – is the total number of analyzed PS.

The total environmental risk of deterioration in status of water bodies is determined by the rule of multiplication of probabilities, where the multipliers are not the values of the risk, and values characterizing the probability of its absence:

$$ER = 1 - (1 - R_1) \times (1 - R_2) \times \ldots \times (1 - R_n), \tag{4}$$

where: *ER* is the total environmental risk of deterioration of water objects;
R_1, ...*Rn* – environmental risk of each pollutant.

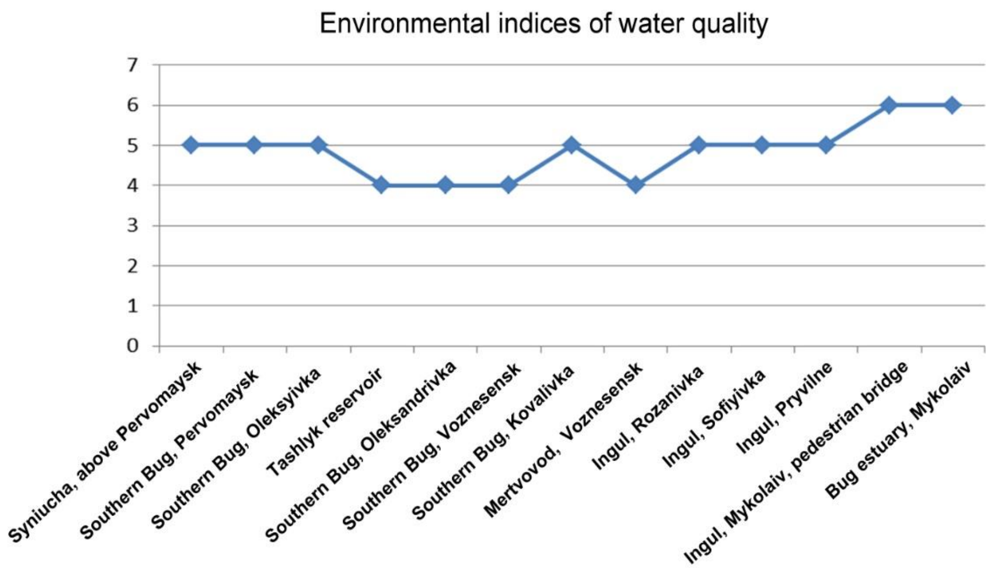

Fig. 3. Environmental indices of water quality.

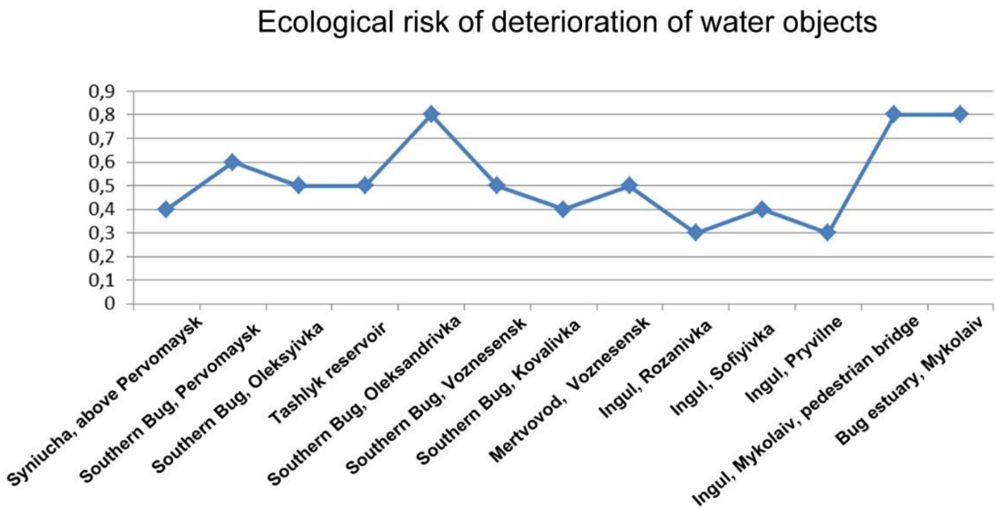

Fig. 4. Ecological risk of deterioration of water objects.

The data indicate that the predominant class of water quality is 3 quality categories with 4 and 5, that is, water in most rivers and reservoirs of the Mykolaiv region are "satisfactory" and "satisfactory mediocre", that is, weakly and moderately contaminated.

In general, the assessment of the status of water in terms of ecological risk coincides with the assessment of environmental quality. Individual points are class II quality "good". Quite often water is "unsatisfactory", class 4 quality (r. Southern Bug in the city of Pervomaisk, and within Mykolaiv). 5 class of water quality is "bad", separately found almost for each item of observations, due to excessive concentrations as a result of an anthropogenic impact on water objects. This situation indicates that water bodies in the study area have somewhat disturbed the ecological parameters of their ecological status is estimated as "ecological regression".

The ecological status of the region water resources is the most degraded by the substances that included in the chemical trophus-saprobiological criterion of pollution, namely: pH, suspended solids, color, COD, BOD_5, total hardness and also by the salt composition criterion: sulfates and chlorides. Pollution of water by components of toxic and radiation action (total iron and manganese) is moderate.

4. Summary and Conclusions

The upgraded method has represented the correct and reliable results of environmental risk calculations.

The research presented that water objects of the area are unsuitable for drinking water supply. The ecological condition is characterized by ecological imbalance due to excess concentrations of pH, suspended solids, color, COD, BOD_5, total hardness, sulfates, chlorides, total iron and manganese.

The main problem of Mykolaiv region is the question of polluted water resources due to unsatisfactory condition, namely, the deterioration of sewage treatment plants. This issue can be solved by expanding and reconstructing wastewater treatment facilities in Mykolaiv, Pervomaysk and Voznesensk; expand and reconstruct sewerage network in these cities; to implement a system of local treatment facilities and disinfection stations, to expand and reconstruct rainwater drainage in the city; to introduce the system of alternative drinking water supply of high-quality purified water with the use of modern technologies; create a system of filtering forest plantations for the treatment of wastewater from the territories of settlements, industrial objects, farms, landfills and fields of filtration.

All of this requires from environmental protection institutions and production organizations that are involved in the field of drinking water supply, to take appropriate measures to improve the situation. The main objective of these measures should be the desire to reduce concentrations of priority pollutants, the list of which was established during the calculations, namely, suspended solids, total iron, manganese, sulphates and chlorides. Of course, these measures will not be able to solve fully the problem of the lack of quality drinking water in the region, but it will be a decisive step forward to improving the environmental situation in the region.

In order to ensure balanced use and protection of water, it is expedient to develop integrated programs of protection and use of water supply sources and drinking water quality in the region; to introduce low-water and water-saving technologies, new modern means of treatment and decontamination of water at water supply facilities; to strengthen the managerial support of the efforts of entrepreneurs to create domestic water treatment equipment.

Due to the limited inventory of fresh groundwater, the presence of load on the ecosystem of water facilities as a result of discharges of poorly treated return water from enterprises and utilities, the decision of the issue of quality drinking water supply of the population is a priority for the oblast.

5. Acknowledgments

We would like to express our gratitude to the Southen Bug Basin Water Resources Department for the information about quality indicators of the water objects of the region.

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Metabolic activity tests in organic matter biodegradation studies in biologically active carbon filter beds

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Abstract

Filtration through a biologically active carbon filter (BAF) is one of the more frequently used advanced water treatment technology. It removes from the water organic compounds responsible e.g. for the color, odor and formation of toxic disinfection by-products including trihalomethanes (THMs). It also increases the biological stability of the water entering the water supply network. Removal of organic pollutants in BAF beds occurs in two ways - by adsorption on the bed and by the process of biodegradation occurring due to microorganisms forming biofilm on bed's surface. Control of the development of biofilm and biochemical processes occurring in the filter bed with the involvement of microorganisms is essential to ensure proper operation of the filters. In this study, the heterotrophic plate count method (HPC), esterase activity assay (FDA) and Eberhardt, Madsen and Sontheimer (EMS) tests were used to evaluate the microbiological activity of GAC filters operated on a pilot scale. There were differences in the number of psychrophilic bacteria and enzymatic activity in two parallel filters working under the same conditions, differing only by the method of the filter bed activation. Moreover, the effectiveness of organic compounds elimination from water was determined using the following parameters: COD_{KMnO4}, dissolved oxygen, total organic carbon (TOC), pH, alkalinity and UV_{254} absorbance.

Keywords

metabolic activity tests, biologically active filters (BAF), organic matter, biodegradation

1. Introduction

The legal regulations in force in Poland and in Europe impose an obligation on water treatment plants to supply recipients with water which meets a number of microbiological, organoleptic, physicochemical and radiological criteria (Council Directive 98/83/EC; Journal of Laws of 2015, item 1989). In order to meet them, more and more advanced water treatment technologies are used. One of them is filtration through a activated carbon biofilter (BAF) using granular activated carbon (GAC), which is designed to remove organic compounds responsible for, among other things, the color, smell and the formation of harmful disinfection by-products including trihalomethanes (THM). It also reduces the concentration of organic compounds in the water introduced into the water supply network, increasing its biological stability (Włodyka-Bergier and Bergier, 2011; Wolska, 2014; Szuster-Janiaczyk, 2016).

Removal of organic pollutants in BAF occurs in two ways: by adsorption on the GAC filter bed and by the process of biodegradation triggered by the microorganisms developing on the bed. In the initial period of the filter operation, the main process is adsorption on the bed. GAC is an adsorber with a very high porosity and specific surface area (600–1000 m^2/g). During the operation of the filter bed, the bacteria naturally occurring in water begin to settle on the porous surface of the grain. They are mainly heterotrophic bacteria for which the environmental conditions in the bed are suitable. After about 2–3 months, biofilm forms on the grain surface. It is composed of microorganisms attached to the substrate by extracellular polymeric substances (EPS) (polysaccharides, proteins, nucleic acids, lipids). It is a porous, tangled matrix that stabilizes inside the microbial cell (Simpson, 2008; Kołwzan, 2011). Therefore, due to reduced contact with the clean bed, biodegradation becomes the predominant carbon removal process.

It is very important for the proper functioning of the BAF to control the development of biofilm and the biochemical processes occurring in the bed with the involvement of microorganisms. The most commonly used methods for determining the activity of BAF include: the bacterial culture – heterotrophic plate counts (HPC) method (Stewart et al., 1990; Łebkowska et al., 1997), Eberhardt, Madsen and Sontheimer (EMS) tests (Papciak et al., 2016), the determination of dehydrogenase activity using TTC (Reczek, 2000), the quantification of ATP (Magic-Knezev and van der Kooij, 2004; Velten et al. 2007), the determination of esterase activity (FDA) (Seredyńska-Sobecka et al., 2006; Pruss et al., 2009), the use of glucose-labeled ^{14}C (Servais et al., 1991; Servais et al., 1994), the observation of bacteria using various microscopy techniques (Surman et al., 1996; Gibert et al., 2013) and the determination of compounds contained in biofilm (phospholipids, polysaccharides and proteins) (Elhadidy et al., 2017). Most of the analyses are expensive or require specialist equipment.

In the presented article, three of the above mentioned methods were used to determine microbial activity in the BAF deposit profile: the culture method (HPC method), esterase assay and the EMS test.

The HPC method is a basic and relatively simple method of determining the number of heterotrophic microorganisms that can be carried out in microbiological laboratories with basic equipment. By this method, the number of different groups of bacteria growing under specific conditions (e.g. thermal or oxygen) on the appropriate growth medium can be determined.

In practice, when examining the microbial activity of bed-settling bacteria, the total number of microorganisms incubated at 22°C for 72 hours for saprotrophic psychrophilic bacteria is most commonly determined. The method for the determination of psychrophilic bacteria in drinking water is described in PN-EN ISO 6222:2004.

The HPC method allows for the assessment of changes in the number of bacteria settling the bed grain during its use. However, that it is not possible to establish the total amount of all the bacteria that develop on the bed, because of their inability to grow on microbiological culture media. It is not possible to reproduce the natural conditions for the bacteria growth in the laboratory and, additionally, that the bacteria are capable of transitioning into the "viable but nonculturable state" (VBNC state). Bacteria in VBNC state are living but do not show growth and colony formation on microbiological culture media on which they normally develop. The transition of cells into this state is the result of various types of environmental stress: starvation, temperature out of range of growth, elevated or lowered osmotic concentration, oxygen concentration, heavy metals or exposure to white light. Bacterial cells, probably by reducing metabolic activity, protect themselves from excessive stress that could lead to their death (Byrd et al., 1991; Oliver, 2005; Oliver, 2010). It is estimated that only 0.25% of freshwater bacteria are capable of growth on culture media (Jones, 1977) and up to 15% in the case of activated sludge (Wagner et al., 1993).

The determination of microbial activity using the FDA test involves the use of esterase enzymes present in living cells. The test is based on the measurement of the activity of non-specific esterases that convert fluorescein diacetate (FDA) to fluorescein, a green fluorescent chemical compound. The test consists in adding a solution of fluorescein diacetate to the sample containing the suspension of the tested microorganisms. Fluorescein diacetate, as a non-polar compound, easily passes into the cell via the cell membrane by way of passive diffusion. In the cell, it is converted to polar fluorescein, which in turn cannot be secreted from the cell by diffusion. Only live cells are capable of showing fluorescence. Furthermore, it has been shown that the ability to absorb fluorescein diacetate by cells depends on the species of bacteria and the ability to produce fluorescein changes in in the cell growth phases. Due to the presence of the outer membrane in Gram-negative bacteria, the penetration of fluorescein diacetate into the cytoplasm is impeded. In some species of Gram-negative bacteria, fluorescence is very weak or absent (Chrzanowski et al., 1984). Hence, this method may not give correct results when the content of Gram-negative bacteria in the biofilm is significant.

The amount of fluorescein can be determined by measuring luminescence intensity using a luminescence spectrometer (excitation wavelenght λ_{ex} = 405 nm or 485 nm, wavelength emission λ_{em} = 520 nm) or a spectrophotometer (absorbance at wavelength λ_{max} = 490 nm) (Chand et al., 1994; Clarke et al., 2001). Modifications of this method can also be used to count cells using an epifluorescence microscope (Breeuwer and Abee, 2000) or flow cytometry (Jepras et al., 1995).

The advantages of this test include simplicity, low cost and short time of execution. The method has been widely used to measure the activity of bacteria in soil (Adam and Duncan, 2001), in freshwater (Battin, 1997) and in activated sludge (Leszczyńska and Oleszkiewicz, 1996; Ziglio et al., 2002). It is also used to determine the viability of algae (Franklin et al., 2001), plant cells (Chen et al., 2015) and fungi (Schading et al., 1995), and to test the antimicrobial activity of natural products (Chand et al. 1994).

An indirect method for determining biological activity in BAF is the Eberhardt, Madsen and Sontheimer (EMS) test. It is based on the determination of the coefficient S, which is calculated as the quotient Δ[COD] by $\Delta[O_2]$, where Δ[COD] denotes the reduction of the chemical oxygen demand of water and $\Delta[O_2]$ is the loss of dissolved oxygen in water. The test is helpful in determining the relationship between the adsorption and the biodegradation process on the BAF bed, assuming that organic compounds are removed both by way of sorption and biodegradation, and oxygen is consumed by aerobic microorganisms to oxidize carbon. If S=1, both adsorption and biodegradation proceed with the same intensity in the filter bed. If S>1, adsorption predominates, and if S<1, biodegradation is predominant. When S and Δ[COD] equal 0, the sorption and biodegradation processes are stopped. If Δ[COD]>0 and $\Delta[O_2]=0$, sorption is present and biodegradation does not occur. In turn, when both Δ[COD] and $\Delta[O_2]$ are equal 0, neither of the processes takes place (Wolborska et al., 2003; Papciak et al., 2016).

2. *Materials and Methods*

The technological investigation was conducted on a pilot scale. The experimental stand consisted of two filtration columns with the diameter of 100 mm and the height of 300 cm, filled with WG-12 granulated activated carbon (Fig. 1.).

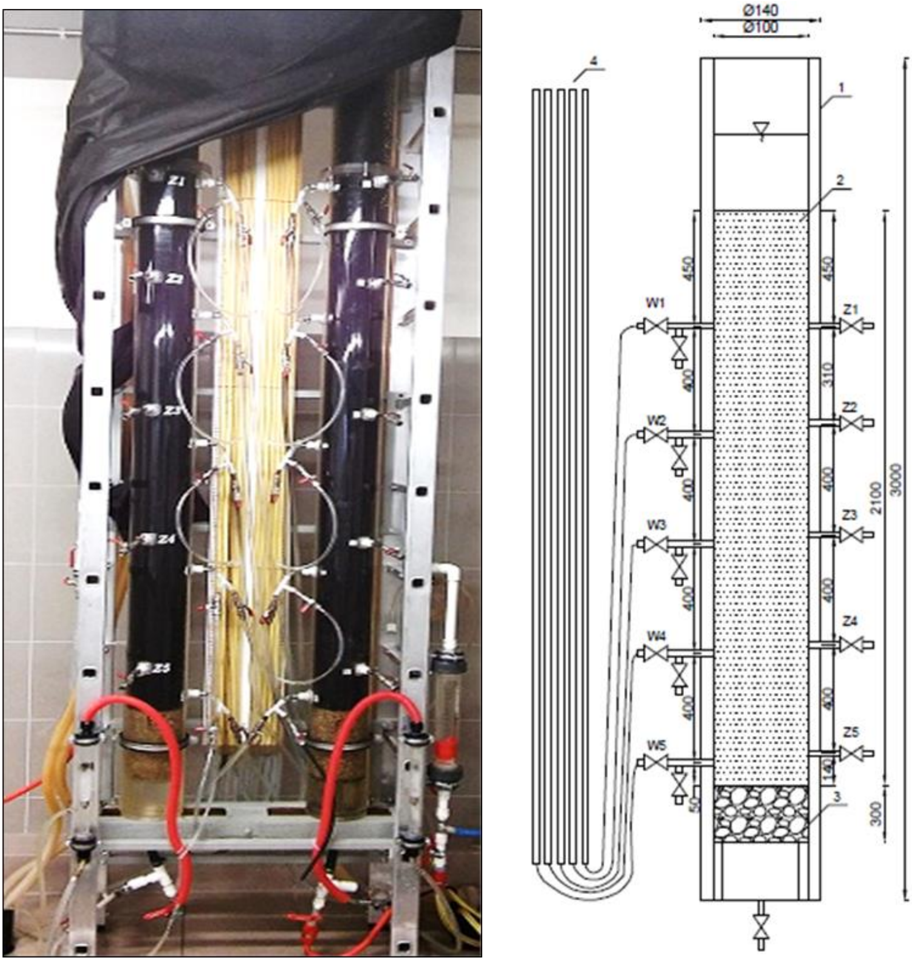

Fig. 1. Test station consisting of two identical filtration columns (photograph and scheme)
1– filtration column, 2 – WG-12 active carbon bed, 3 – supporting gravel layer,
4 – piezometers, W1-W5 – water sampling, Z1-Z5 – filter bed sampling

Stable temperature was maintained across the entire filter bed height thanks to a water jacket - a pipe with the inner diameter of 140 mm filled with water. The water flowing through the pipe had a temperature equal to the filtered water temperature. In order to prevent the growth of algae, the filters were covered with black geotextile. The water supplied to the filters was dechlorinated tap water. Across the entire filter height, stub pipes were located for the sampling of water and the filter bed. The system was activated in April 2015 and it is still being operated. The filtration columns differed with regard to the method of the filter bed activation (Holc et al., 2016a) and the microorganisms found there (Holc et al., 2016b). In the first filtration column, the microorganisms residing in the filter bed came from the water passing through the filter i.e. from the network water. The filter bed of the other filtration column was inoculated with the wastewater from the backwashing of carbon filters operated in a selected water treatment plant. The water for analysis were collected on the supply line (W1-W5) from the filtration columns and from the filter bed cross-section, at the depth of 45 cm, 85 cm, 125 cm, 165 cm and 205 cm. Firstly, microbiological samples were taken to 100 ml sterile bottles, then water samples for physical and chemical tests were taken to 2 liter bottles. The filtration velocity was 5 m/h. The filter beds were backwashed with a one-week delay due to a high number of determinations.

The effectiveness of organic substances elimination from water was assessed using the following parameters: COD_{KMnO4}, dissolved oxygen concentration, total organic carbon (TOC), pH, alkalinity and UV_{245} absorbance.

The amount of psychrophilic bacteria was determined after their deep inoculation and growth on enriched agar (HPC method) and the metabolic activity of biomass by the FDA method (Kijowska et al., 2001).

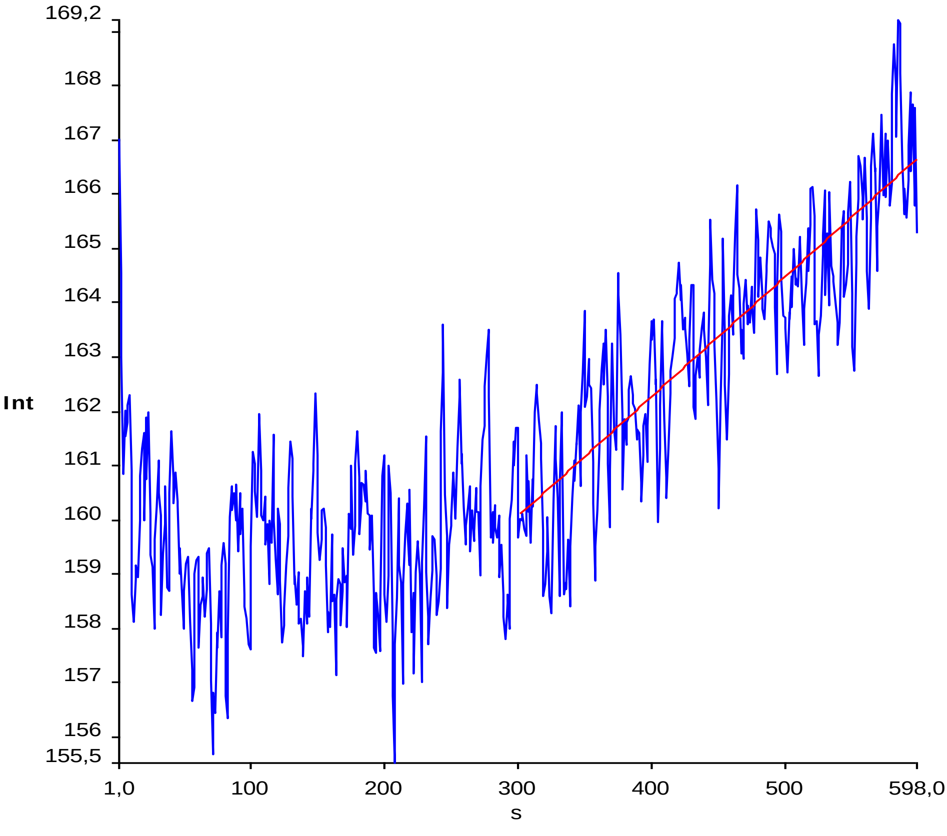

Fig. 2. Changes of the luminescence intensity - sample dated 31.05.2017

To measure the enzyme activity, samples of 3 mL of water were collected and transferred to a cuvette. 120 μL of fluorescein diacetate solution was added to the sample and measurement was started. The measurement of the FDA hydrolysis rate (fluorescein luminescence intensity) was performed using the PerkinElmer Instruments LS55 luminescence spectrometer, at the excitation wavelength of 433 nm and the emission wavelength of 525 nm. The spectrometer was compatible with a computer equipped with the appropriate FLWinLab software. Continuous measurement was performed over the period of 10 minutes. After the adaptation time, in all samples FDA hydrolysis occurred. The metabolic activity of microorganisms was read from the slope of the straight line representing the relationship of the fluorescein luminescence intensity and time, for a period of 5–10 minutes. The value was expressed in relative units per second [r.u./s]. Figure 2 shows an example of the luminescence intensity measurement.

3. *Results and Discussion*

Table 1 shows the quality of the water supplied to the BAF model. The small differences between the inflows resulted from the different filter backwashing dates.

Table 1. The quality of the water flowing into the filters during the analyzed filter cycle (up to the measurement day).

Parameter	**Unit**	**Filter 1**	**Filter 2**
Temperature	^oC	15.8	15.6
pH	-	7.70	7.58
Alkalinity	mg $CaCO_3$/L	187.5	188.0
Dissolved oxigen	mg O_2/L	4.58	4.45
COD $_{KMnO4}$	mg O_2/L	3.87	3.89
Absorbance 254 nm	cm^{-1}	0.0065	0.0067
Psychrophilic bacteria	CFU/mL	30	41

The effectiveness of organic pollutants removal on BAF filters changes during the filter cycle (Pruss et al., 2009; Gibert et al., 2013; Holc et al., 2016a; Holc et al., 2016b; Kołaski et al., 2017). This is primarily due to the development of biofilm on the bed and its break-up and entry into the drain. Figures 3–5 show the results obtained from the samples collected from both filters on 13th day of the filter cycle. This day of filter work was chosen because the analysis of the results obtained from the beginning of the BAF model exploitation (Holc et al., 2016a; Holc et al., 2016b) and from presented filter cycle indicated that in this time stable state of filter work was achieved.

The water collected from the filter bed profile contained mainly free-flowing bacteria and fragments of the biofilm (microscopic observations). In spite of very slow water flow during the sampling procedure, it cannot be ruled out that the local surge in the flow velocity increased the amount of washed out bacteria. This thesis is confirmed by the fact that the amount of bacteria in the effluent from both filters was similar to that in the inflowing water (Fig. 3). The sampling method used in the bed profile was the same, hence the higher number of bacteria washed out of the filter is an indirect indicator of their greater number at a specific depth of the bed. As expected, the highest number of bacteria was found in the upper layer of the bed, which is related to the availability of oxygen and organic compounds.

There was no relationship between the metabolic activity and bacterial amount at a specific depth of the bed. This may be due to the fact that the FDA is a compound whose microbes are not hydrolyzed in the beds, because it does not penetrate the cell membrane. Another reason may be the content of carbon microparticles (visible in the microscope) in the water, which are capable of FDA adsorption.

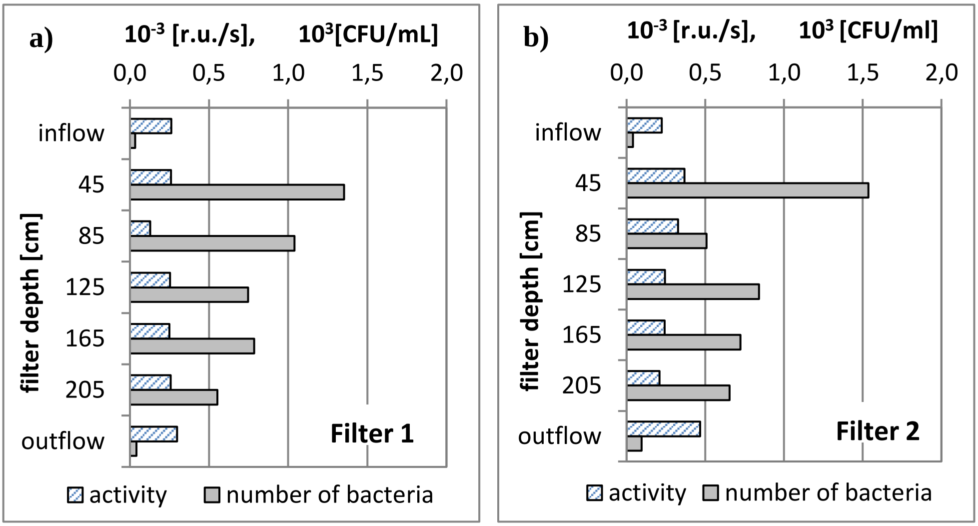

Fig. 3. Changes of the biomass activity and number of bacteria in filter 1 (a) and 2 (b)

The metabolic activity measured in filter 1 remained at a similar level throughout the bed. Its averaged value was $0.2294 \cdot 10^{-3}$ r.u./s. Although its increase in the effluent relative to the inflow was $0.0366 \cdot 10^{-3}$ r.u./s. In filter 2, an increase in the activity at the depth of 45 cm was detected and then the activity measured in the samples collected from lower parts of the bed decreased. The average activity in bed of filter 2 bed was $0.2781 \cdot 10^{-3}$ r.u./s. The increase in the activity was observed only in the effluent. The increase in the activity at outflow from filter 2 in relation to the inflow was $0.2443 \cdot 10^{-3}$ r.u./s.

The number of bacteria in the water samples collected from the bed profile of both filters (W1 to W5), was comparable and was 895 CFU/mL in filter 1 and 853 CFU/mL in filter 2 (Fig. 3). There was no relationship between the amount of bacteria and activity and also between the changes in COD and oxygen concentration in the bed (Fig. 4). Despite the lower activity in the bed, on the measurement day, the filter 1 removed more organic compounds in terms of COD, i.e. 0.63 mg O_2/L versus 0.56 mg O_2/L in filter 2. On the other hand, oxygen consumption by microorganisms was lower in filter 1 and equaled 1.00 mg O_2/L, compared to 1.42 mg O_2/L in filter 2. This indicated that the biodegradation process was more advanced in the latter filter. The smaller amount of carbon removed by filter 2 versus the inlet value was caused by the oxidation of the biomass accumulated in the filter.

Eberhardt, Madsen and Sontheimer (EMS) test is often used to determine the biological activity of biologically active carbon beds. By performing the test for the entire bed, it was found that the biodegradation was a predominant process in both filters, because the S coefficient for filter 1 was 0.63 and for filter 2 was 0.39.

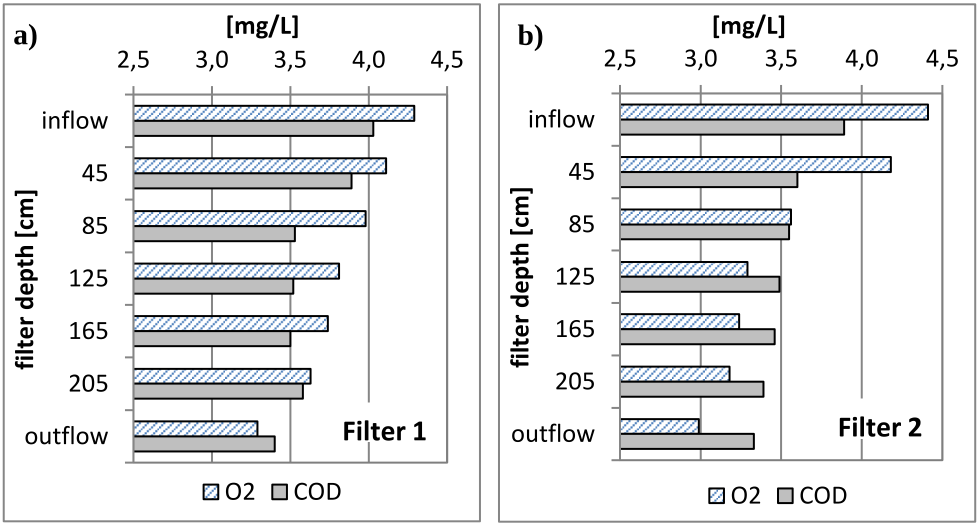

Fig. 4. Changes of the oxygen concentration and COD in filter 1 (a) and 2 (b)

The share of biodegradation and sorption varied in the individual bed layers (Fig. 5). There was a high proportion of sorption in the upper layers of the bed, which may indicate biosorption, because the working time of the beds was longer than two years. S values <0 were also obtained. This may be the case when carbon (biomass) is removed from the filter and the dissolved oxygen concentration decreases as the biodegradation process takes place.

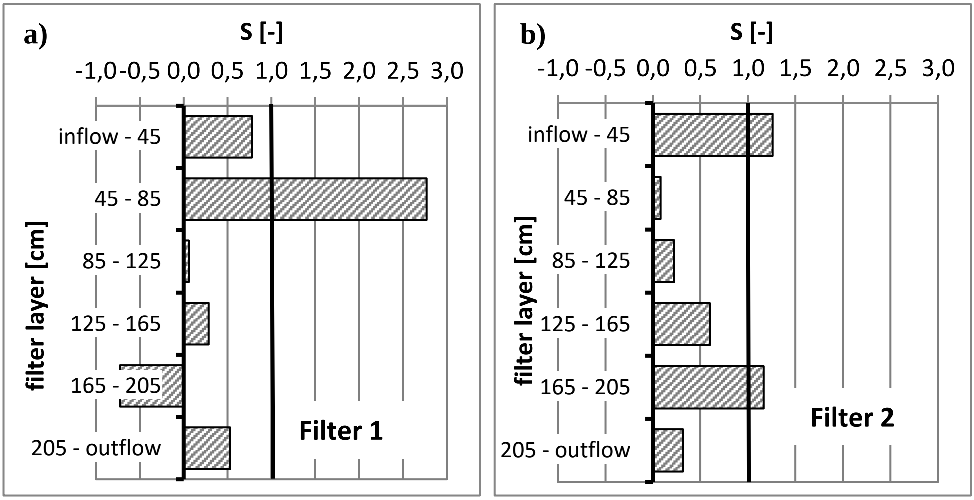

Fig. 5. Changes of the S coefficient (results of EMS test) in filter 1 (a) and 2 (b)

4. *Summary and Conclusions*

Biologically active filters are devices in which several processes occur in parallel: sorption, biodegradation, the formation of new bacterial cells and the dying of the old ones. After the two-year operation period it can be concluded that the analyzed filters are biologically active all over their depth and remove organic compounds (COD) from water efficiently.

Based on the investigation of the water samples collected from the two filter columns working under the same conditions, differing only by the method of the filter bed activation, the following conclusions were made. The number of the psychrophilic bacteria in both filters was comparable, albeit different in particular layers of the filter bed. The highest amount of bacteria was noted in the upper part of the filter beds, which was undoubtedly due to more favorable oxygen conditions and better availability of nutrients. No relationship was found between the metabolic activity and amount of bacteria at a specific depth of the bed, and also between those parameters and changes in the COD and oxygen concentrations in the filter bed. The EMS test showed that the biodegradation was predominant in both filters (in filter 1 S=0.63, while in filter 2 S=0.39). The share of biodegradation and sorption varied in the individual bed layers. The high proportion of sorption in the upper layers of the bed may indicate biosorption.

The studies have proven that the number of psychrophilic bacteria in the water samples collected from the filter bed does not accurately reflect the microbial activity. In turn, the FDA test may not give precise information about the metabolic activity of microorganisms in the filter bed, due to the application of this method to specific groups of microorganisms. Therefore, in order to accurately characterize the bacteria developed in biofilms and to study their activity, a range of different research methods including advanced taxonomic identification e.g. molecular biology techniques should be used.

Acknowledgments

The study was financed under the project 01/13/DSMK/0864 "Using of modern microorganism identification methods for the study of biologically active carbon filters".

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Analysis of water losses on the example of the selected water distribution system

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Abstract

One of the major problems of water supply companies around the world is the high level of water losses in the water supply networks. This problem concerns both small and large water supply systems. In Poland, the problem of water losses was not observed until the 1990s, when the market economy and the individual measurement of water consumers were introduced. The type and amount of losses is influenced by many various factors, which are different in each distribution system. Consequently, on the one hand, there is a need for in-depth individual analysis of the work of specific water supply networks, so that a roadmap for reducing water losses in the system can be developed. However, on the other hand, a uniform approach to the problem not only within individual countries, but also on an international scale is needed, which will enable to compare distribution systems in terms of water losses. In recent years, in Poland, similarly as in many other countries, balancing and loss measurement methodology proposed by the International Water Association (IWA), and indicator method have been used.

The aim of this paper is the analysis of water losses in the years 2008–2016 in the selected municipal water distribution system in Mazowieckie Voivodship. The analyzed system has an average water network intensity indicator of approximately 38 [$m^3/(d \cdot km)$]. The analyses based on standard methods (IWA method and indicators method) have shown relatively small water losses (from 7.34% to 13.24%) compared to the literature values (on average 21.4%) for systems comparable to the analyzed one in terms of load. The conducted research allowed for the assessment of the technical condition of the network and proposing actions aimed at reducing water losses in the system, which results in economic benefits and improved the quality of services provided by the water supply company.

Keywords

water balance, water losses, indicators method, IWA method

1. Introduction

Rational water management, both by suppliers and consumers, is closely connected with the quality of water distribution management in water supply systems. However, the quality of the distribution system management is determined by the water losses and the network failure related to them. Both factors mainly resulted from the poor technical condition of the fittings and water pipes. The problem of identifying and estimating the water losses in water supply systems affects almost all the water supply in the world and has been undertaken in numerous national and international scientific studies (Lambert, 2003; Kowalski and Miszta-Kruk, 2003; Kutyłowska and Hotloś, 2014; Kutyłowska, 2015a,b; Iwanek et al., 2015, 2016, 2017). Water losses are undesirable, but inherent in the operation of all water supply networks. The multiplicity of their causes and their individual character require the analysis of specific distribution systems. A uniform approach to the problem of losses in all networks is also recommended, so that mutual comparisons can be made as well as the reference to the values of indicators considered as standard.

In Poland, the problem of water losses was strongly reflected in the changes in the political system and the introduction of a market economy, which forced the water companies to take measures aimed at reducing excessive water losses. Water losses defined as the difference between the measured volume of water pressed into the network and the invoiced amount of water supplied to the recipients are divided into actual and apparent. Losses, regardless of their type, affect the quality and cost of water supplied to the consumers (Siwoń et al., 2004), as well as the financial condition of the water companies (Siwoń et al., 2004; Kowalski et al., 2012; Żaba and Langer, 2012; Zimoch and Szymura, 2013; Piechurski, 2014).

Water losses are one of the basic elements of the technical condition of the water supply system. The analysis of these losses should therefore be the basis for undertaking modernization and corrective actions and thus reducing the costs associated with the production and distribution of water. Water losses can be estimated by the indicators method and method of balancing the amount of water recommended by the International Water Association (IWA).

The purpose of this paper is to present an example of water losses analysis in the selected municipal water supply system. The obtained results of the calculations allowed for assessing the state of network and propose actions aimed at reducing water losses.

2. Materials and Methods

The first stage of the research was to obtain the operational data from the water supply company managing the analyzed network. The data that has not been measured by the network operator has been estimated based on the literature or experience of the network manager. The data was used to create a simplified IWA balance and to calculate selected water losses indicators. The analysis of the calculation results enabled to assess the state of the water supply network.

2.1. Description of the network

The analysis of water losses was carried out for one zone of the selected municipal water supply system, located in the southern part of Mazowieckie Voivodship. The entire network is powered by 7 water supply stations which supply the separated zones. Due to the lack of connection between the areas, the analysis included one of them, supplied from two water supply stations located in the middle of the city, for the purpose of the article called Zone A. The total length of the pipes and the connections in the Zone A under the analysis is 102.23 km. Network pipes are made mainly of PE and PVC; however, all house connections are made of PE. The average pressure in the zone is 0.4 MPa. The area served by the water supply network in the analyzed zone in the years 2008–2016 was inhabited by an average of 13 373 people. The total average daily water production in Zone A is approx. 2 831.88 m^3/d.

The characteristics of the analyzed zone are shown in Table 1. The average value of the area's water network intensity indicator in 2008–2016, calculated according to the equation (1) (Dohnalik and Jędrzejowski, 2004).

$$WNII = \frac{SIV}{L_m} \tag{1}$$

where: SIV – System Input Volume [m^3/day], L_m – length of mains [km].

Table 1. Characteristic of water well zone A

Parameter	**Zone A**
Length of mines [km]	75.78
Number of house connections [-]	1 669
Length of house connections [km]	26.45
Water Network Intensity Indicator (*WNII*) [m^3/day/km]	38.89

2.2. Calculations of water loss performance indicators recommended by IWA

Water balance according to IWA consists in dividing the water introduced into the network into types and determining the number of individual types (Lambert, 2010; Kutyłowska, 2015a; Musz-Pomorska et al., 2017). System Input Volume (*SIV*) is divided into: Billed Authorised Consumption (*BAC*), Unbilled Authorised Consumption (*UAC*), Apparent Losses (*AL*) and Real Losses (*RL*).

The values of *SIV* and *BAC* were obtained from the water companies. *UAC* was estimated as 6 000 [m^3/yr], according to the information obtained from the network operator. *AL* were calculated as a sum of Unauthorized Consumption (*UC*) and Customer Metering Inaccuracies (*CMI*). *UC* and *CMI* were calculated according to the literature (Bergel, 2012 a, b; Choma et al., 2014; Merlo, 1992; Musz-Pomorska et al., 2016) as 2% and 3% of *SIV*, respectively. *RL* was calculated as a difference between *SIV* and the three components of the balance (*BAC*, *UAC* and *AL*).

In order to evaluate water losses, performance indicators recommended by IWA and commonly used in Poland were calculated, including Real Losses Level per connection per day (*RLB*), Infrastructure Leakage Index (*ILI*), Non-Revenue Water Level (*NRWB*), Percentage Indicator of water losses (*WS*), Water Losses Index per person (*WSM*) and Water Losses Index per kilometer (*WSL*) (Tab. 2, 3) (Lambert, 2010; Bergel, 2012a, b; Choma et al., 2014).

Table 2. Indicators recommended by IWA (Lambert, 2010).

Indicators	**Formula**	**Abbreviations**
RLB [m^3/km/d]	$RLB = \frac{CARL}{N_c}$	*CARL* – Current Annual Real Losses corresponding to *RL* from an IWA water balance [dm^3/day], N_c – number of service connections [-].
ILI [-]	$ILI = \frac{CARL}{UARL}$	*UARL* – Unavoidable Annual Real Losses [dm^3/day].
UARL [dm^3/day]	$UARL =$ $= [18 \cdot L_m + 25 \cdot L_c + 0.8 \cdot N_c] \cdot P$	L_m – length of mines [km], L_c – average distance from property line to a customer meter [km], in Poland usually corresponding to the length of connections, *P* – average operating pressure head [m H_2O].
NRWB [%]	$NRWB = \frac{SIV - BAC}{SIV} \cdot 100\%$	

Table 3. Indicators commonly used in Poland (Bergel, 2012a, b; Choma et al., 2014).

Indicators	**Formula**	**Abbreviations**
WS [%]	$WS = \frac{WL}{SIV} \cdot 100\%$	*WL* – water losses [m^3/day] calculated as the sum of Real and Apparent Losses according to an IWA water balance.
WSM [dm^3/person/day)]	$WSM = \frac{WL}{N_p} \cdot 1000$	N_p – number of population [person].
WSL [m^3/h]	$WSL = \frac{WL}{24 \cdot L_m}$	L_m – length of mines [km].

The obtained performance indicators results for system in question were analyzed and used to evaluate the condition of system according to World Bank Institute Physical Loss Assessment Matrix (Tab. 4) (Liemberger et al., 2007).

Table 4. Components of a simplified IWA water balance (Liemberger et al., 2007).

Technical Performance Category	***ILI***	***RLB* [dm^3/connection/day] at an average pressure of:**			
		0.2 [MPa]	**0.3 [MPa]**	**0.4 [MPa]**	**0.5 [MPa]**
A	1–2	<50	<75	<100	<125
B	2–4	50–100	75–150	100–200	125–250
C	4–8	100–200	150–300	200–400	250–500
D	>8	>200	>300	>400	>500

3. *Results and Discussion*

Within this work, complete water balances have been prepared for the entire distribution system in question for the years 2008–2016. The selected components of these balances are summarized in Table 5.

During the analyzed period, the amount of *SIV* was kept at a constant level (1 046 435 [m^3/year], on average). The volume of *SIV* from 2008 to 2013 has been declining. In 2013, the amount of water lower by nearly 170,000 [m^3], in comparison to 2008, was pressed into the system. The greatest amount of water (11.2% more than the average) was added to the network in 2015. In 2015, the highest value was also recorded for real losses (*RL*), which were more than 52% higher than the average value (79805 [m^3/year]). Apparent losses (*AL*) and billed authorized consumption (*BAC*) were highest in 2008.

Table 5. Components of the water balance over the period 2008–2016, expressed in m^3/yr

Year	*SIV*	*BAC*	*UAC*	*AL*	*RL*
2008	1132516	1021255	6000	32672	72589
2009	1084059	944266	6000	30209	103584
2010	1044094	918284	6000	29377	90433
2011	997396	889350	6000	28452	73594
2012	968081	886069	6000	28347	47665
2013	962679	886052	6000	28346	42281
2014	989533	884798	6000	28306	70429
2015	1163852	1003734	6000	32112	122006
2016	1075703	943844	6000	30195	95664
Avarege	**1 046 435**	**930850**	**6 000**	**29780**	**79805**

Calculation of water balances enabled to determine the value of indicators of water losses for the analyzed water supply system in the years 2008–2016 (Tab. 6). The average value of *UARL* for the considered period is 118 456 m^3/year. Figure 1 shows the change of *UARL* value in 2008–2016.

Table 6. Indicators of water losses in distribution system in 2008–2016

Year	WS	WSM	WSL	RLB	ILI	NRWB
	[%]	[dm^3/person/d]	[m^3/h/km]	[m^3/km/d]	[-]	[%]
2008	9.29	21.56	0.22	139.76	1.86	9.82
2009	12.34	27.41	0.28	199.43	2.66	12.90
2010	11.48	24.55	0.25	174.11	2.32	12.05
2011	10.23	20.91	0.21	141.69	1.88	10.83
2012	7.85	15.57	0.12	86.31	1.05	8.47
2013	7.34	14.47	0.11	75.91	0.92	7.96
2014	9.98	20.23	0.16	126.28	1.53	10.58
2015	13.24	31.57	0.24	216.77	2.61	13.76
2016	11.70	25.78	0.19	157.04	1.95	12.26
Avarege of literature data*	**21.40**	**30.40**	**0.15**	**154.10**	**1.90**	**24.00**

* - (Bergel, 2009; 2012a, b)

When analyzing the calculated values of indicators of water losses (Tab. 6), it should be noted that the tested water supply system is characterized by relatively small water losses compared to the systems of similar size (Fig. 2). For all calculated indicators of water losses, the lowest values were obtained between 2012 and 2013. The average water losses in the water supply system amount to 10.38% and they are twice lower (Fig. 2) than the average value found in literature (Bergel, 2009; 2012a; b).

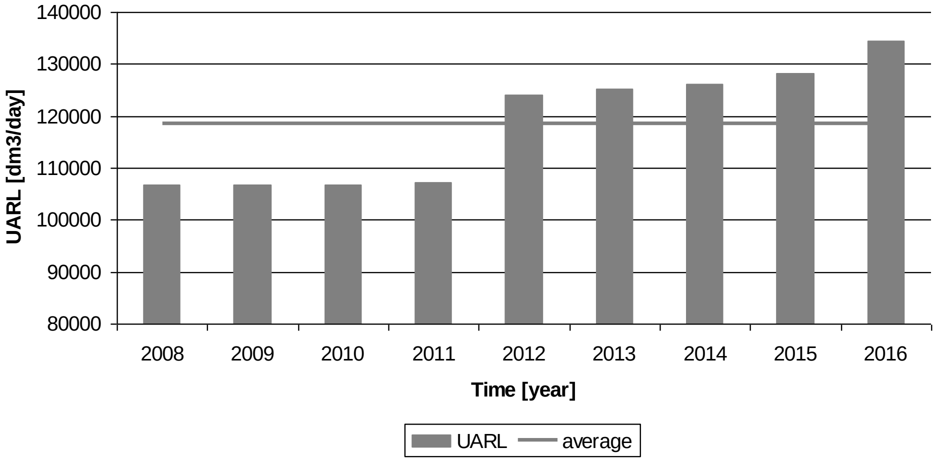

Fig. 1. *UARL* value change in 2008–2016

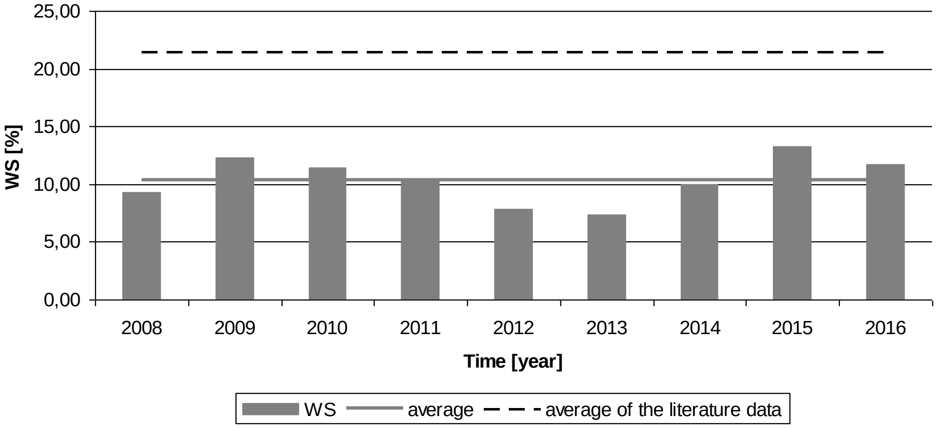

Fig. 2. The percentage water loss

The value of *WSL* varied widely. A minimum value of 0.11 $m^3/(hr \cdot km)$ occurred in 2013 and the highest equal 0.28 $m^3/(hr \cdot km)$ in 2009.

The values of *ILI* shown in Figure 3 correspond to the values of *RLB* shown in Figure 4. The average *ILI* value calculated for the years 2008–2016 is 1.86 and is slightly lower than the average value found in the literature. The highest *ILI* values above 2.0 were observed in 2009, 2010 as well as 2015 and were equal to 2.66, 2.32 and 2.61, respectively, while the lowest value of *ILI* – 0.92 was recorded in 2013.

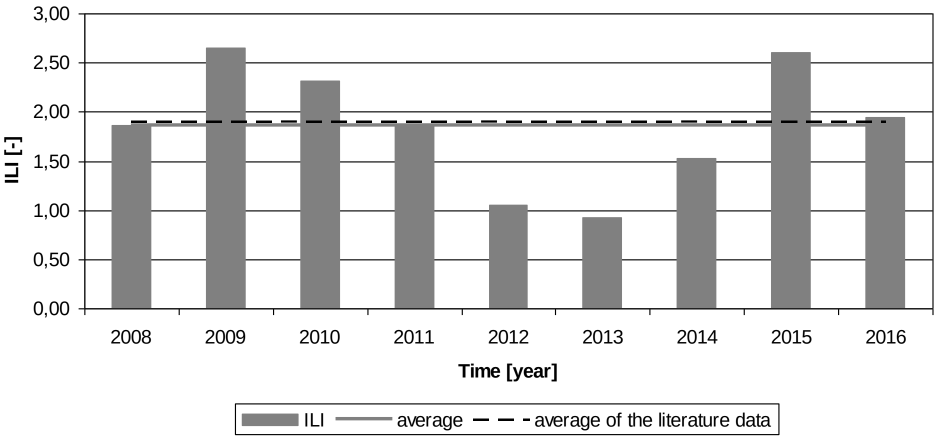

Fig.3. Operational indicator *ILI*

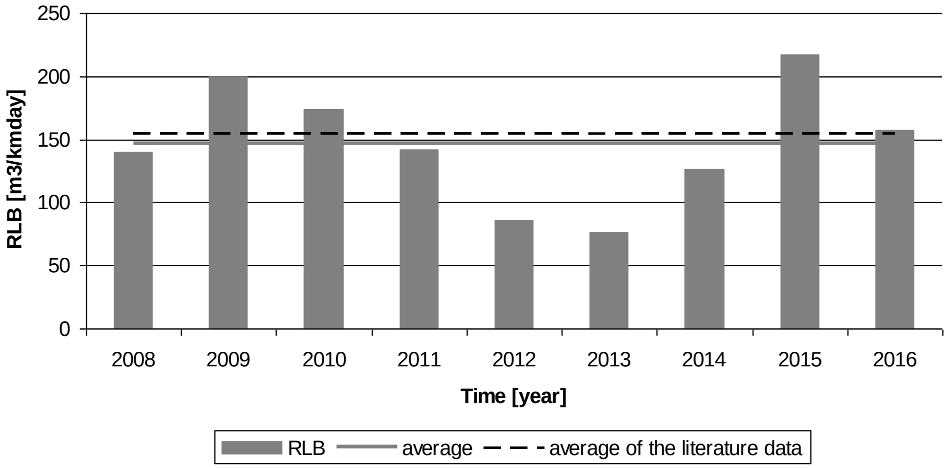

Fig.4. Operational indicator *RLL*

The *ILI* value below 2 indicates Category A according to the WBI Target Matrix (Table 2) in 2008, 2011–2014, and 2016. The *RLB* indicator according to the WBI Target Matrix (Table 4) indicates Category A only in 2012 and 2013. Taking into account both indicators, the analyzed water supply system in 2012 and 2013 could be classified to Category A and in the remaining years to B. It should be emphasized that *UARL* used in the calculation of *ILI* is a reliable predictor for a system with more than 5000 service connections, density of connections greater than 20 per km of the network and average operating pressure greater than 25 m H_2O (Winarni, 2009; Lambert and McKenzie, 2002; Lenzi et al., 2014) . The first of these conditions is not met by the system in question. However, on the basis of tests and the analysis, the guidelines for New Zealand (McKenzie and Lambert, 2008) recommend reducing this limitation by replacing 3 conditions by a single one: *UARL* calculation should be reliable, if (Lm · 20 + Nc) exceeds 3000. This condition is met by the system in question (Musz-Pomorska et al., 2017).

The *NRWB* value in the period under consideration is slightly different (Fig. 5).The lowest value of 7.96% occurred in 2013, while the highest of 13.76% in 2015. The average value of the index calculated for the years 2008–2016 is 10.96%.

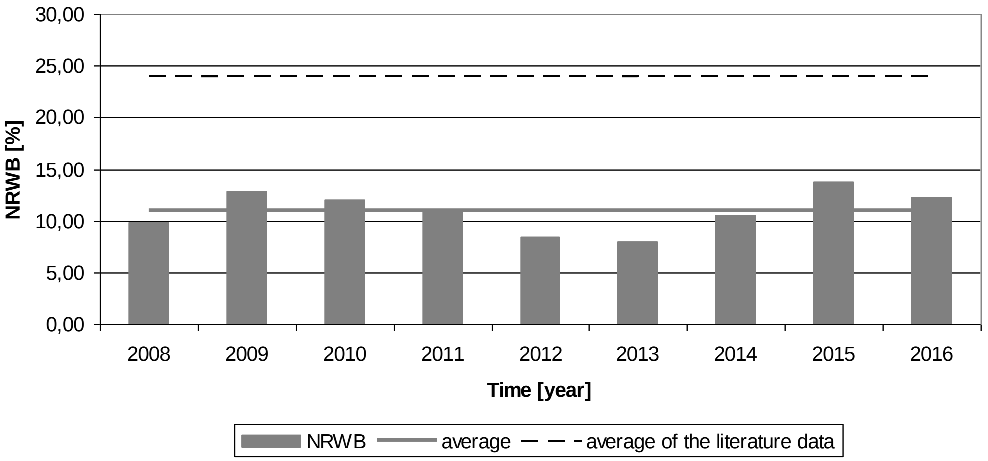

Fig.5. Financial indicator *NRWB*

Over the last decade, there has been an increase in awareness of water loss management in Poland. Numerous water supply companies use the calculated of water losses indicators for efficient and conscious management (Pietrucha-Urbanik, Tchórzewska-Cieślak, 2015; Iwanek et al., 2016; Musz-Pomorska et al., 2016). Table 6 shows the published mean of values of water losses: *RLB, ILI*, *NRWB*, for systems that are similar to those analyzed by the *WNII* indicator.

The values given in Table 7 indicate that the water losses in the tested water supply system are the lowest in comparison to other cities, which may confirm its good state. On the other hand, such low index values as compared to those reported in the literature may imply inaccuracies in the estimation of water balance components.

Table 7. Water losses in water distribution systems in selected Polish towns

Town	***WNII***	**Population**	***RLB***	***ILI***	***NRWB***	**Literature**
-	**[m^3/day/km]**	**-**	**[dm^3/(conn./day)]**	**[-]**	**[%]**	**-**
Sanok	40.35	39 569	625.83	10.28	40.2	Piegdoń and Tchórzewska-Cieślak (2012)
Krosno	38.30	47 307	889.3	7.3	-	Pietrucha-Urbanik and Studziński (2012)
Myszków	30.68	32 830	159.78	2.26	27,2	Kędzia and Ociepa (2015)
Jasło	47.01	36 640	362.9	5.0	20.3	Rak and Sypień (2013)
Zone A	**38.89**	**13 373**	146.00	1.86	10.96	-

4. *Summary and Conclusions*

The analysis of the water loss indicates the good technical condition of the analyzed network. According to the WBI Loss Target Matrix, the calculated average values of *RLB* and *ILI* indicators allow for classifying the analyzed water supply system in 2012 and 2013 into category A (systems that do not require further reductions of losses) and in other years, into the category B (systems requiring further repair and reduction of losses). Very good technical condition of the water supply system in 2012–2013 is also confirmed by the values of other water losses indicators (*WS, WSM, WSL, NRWB*).

Comparison with other Polish water distribution systems also indicates the good condition of the analyzed network. However due to the certain assumptions for the calculations, the values obtained should be regarded as indicative.

Real water losses are strongly connected with leakages occurring during breakages or failures of a network. Thus, the analysis of the water losses, apart from the analysis of indicators proposed by IWA and commonly used in Poland, should be supplemented by an analysis of the failures occurring on the analyzed network.

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Tannery wastewater treatment

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Abstract

Tannery wastewater is highly contaminated by organic and inorganic pollutants and highly turbid, colored and foul-smelling. The major components include sulfide, chromium, volatile organic compounds, large quantities of solid waste, suspended solids like animal hair and trimmings. The tannery wastewater characteristics are varied due to changes in cycling daily discharge. The kind of tanned skin influences wastewater characteristics. The characteristics of the wastewater are also dependent on the stage of production. The quantity and quality of tannery wastewater are discussed in the article. The environmental protection regulations stipulate that industries be not allowed to emit sulfide and chromium in the wastewater. Thus, removal of sulfide and chromium from the wastewater is significant. Tannery wastewater contains the significant amounts of sulfides, which hamper biological treatment process. In the article shows catalytic oxidation as the best method of remove sulfides. Chromium can accumulate in bottom sediments, which negatively affects the life and development of aquatic organisms and plants. The article shows the traditional chemical chromium treatment methods: chromium precipitation and chromium sorption onto activated carbon. Possible is support these processes by biopreparation, using closed-circuit or use wet-white. Chromium precipitation is also wastewater pretreatment, so activated sludge is protected from the toxic effects of chromium on further stages of the biological treatment. Very important in tannery wastewater treatment is using membrane techniques. In the case of the use of membranes essential are chromium recovery and repeated chromium using for tanning hides. The article discusses the processes of microfiltration and ultrafiltration, nanofiltration, reverse osmosis and separation with liquid membranes. The search for additional traditional wastewater treatment is indicated. Biological wastewater treatment is more favorable and cost-effective as compared to other physiochemical methods.

Keywords

tannery wastewater, tannery wastewater treatment

1. Introduction

The leather is a post-production waste from the food industry. It is an environmentally burdensome product. However, after proper preparation processes, it is the ideal raw material used in the clothing and shoe industries. The leather industry is one of the oldest known. The skin can be obtained from practically every mammal, reptile or bird, but most commonly used are porcine and bovine skins. (Religa and Gierycz, 2010)

Leather tanning consists of proper preparation and transformation into a product which has much better physical and mechanical properties. It should be suitably tensile and abrasion resistant. The appropriate technological process can provide such properties. The leather processing includes many activities such as preparation of raw material, tanning and finishing (Bartkiewicz and Umiejewska, 2010).

Beam house operations and tanning process are two parts of leather manufacturing. The first step in beam house operations is the removal of dirt and blood by washing. After that, the skins are soaked in water for softening and removal of salts. The next step is the removal of fatty tissue by fleshing. To swell the skins for the better penetration of tanning agents and hair removal the liming is used. The next operation part is the chemical dissolution of the hair and epidermis with an alkaline medium of sulfide and lime. During liming, a high concentration of sodium sulfides, organic matter and lime are in the effluent and after that skins are neutralized with acid ammonium salts and treated with enzymes. Enzymes are used to remove the hair remnants and to degrade proteins, what results in a significant part of the ammonium load in the effluent. Pickling is often used for the preparation of skins for tanning. The pH-index is adjusted mainly by the addition of sulphuric acid. Some salts are added for preventing the skins from swelling (Buljan and Kral, 2011).

Next part of leather manufacturing is tanning. It is the reaction of the collagen fibers in the skins with chromium, tannins, alum or other chemical agents. One tone of raw skin requires about 300–450 kg chemicals for technological processes of tanning. Based on the tanning agents, tanning operations are further divided into chrome tanning and vegetable tanning (Midha and Dey, 2008). After the tanning, the skin is immune to rotting, keratosis and swelling (Bartkiewicz and Umiejewska, 2010).

The final finish depends on the destination, the brand and the type of skin. The last stage is its staining, greasing and planning. Between each process, the skin should be rinsed. Skin after greasing is subjected to wringing and drying (Bartkiewicz and Umiejewska, 2010). Tanning agents could help in the permanent stabilization of the skin matrix against biodegradation, but the tanning industry has gained a negative image in society concerning the potential of pollution and therefore is facing a severe challenge (Buljan and Kral, 2011; Midha and Dey, 2008).

2. *Tannery wastewater characteristics*

In the tannery industry could be distinguished several places in which wastewater are generated. Tannery wastewater tends to vary the composition. This is related to the need to discharge wastewater from specific steps of technological operations during the day. There is also no daily cyclicality when wastewater is discharged (Bartkiewicz and Umiejewska, 2010). An example of a wide range of pollutant concentrations is shown in Table 1.

Table 1. Parameters of tannery wastewater

Parameter	**Tanning**	
	chrome	**vegetable**
pH-index	7.5–12.5	4.4–12.2
Suspended solids [mg/l]	452–20645	390–5048
Chlorides [mg/l]	2140–3950	2100–3750
Sulphides [mg/l]	9–140	15–99
Chromium[mg/l]	11–3226	-
Total nitrogen [mg/l]	177–470	144–424
COD [mg O_2/l]	232–4208	1111–7440
BOD_5[mg O_2/l]	235–2700	278–3556

Swine-skin wastewater has a higher concentration of pollutants than bovine-skin wastewater. This is because swine and bovine skins vary considerably in chemical composition (Myjak, 2007). Significant differences in BOD and COD are due to different fat content in the skin of these animals. According to Myjak (2007), the fat content in the bovine skin is about 1–3%, whereas in pigs it reaches up to 16%. Swine skin is easier to connect with a tan than bovine skin in tanning process (Myjak, 2007). Therefore, in some cases, chromium pigments may be lower in chromium pigment effluents. Table 2 shows loads of pollutants in tanning wastewater (Klepaczewski, 2007).

Table 2. Loads of tannery wastewater impurities

Parameter	**Load [kg/tone of raw skin]**
Chlorides	170
Sulfides	7
Suspended solids	116
COD	188
BOD_5	68
Chromium (III)	5

After preparation of the raw skins, the wastewater from the rinsing and soaking are generated. These wastes contain fat, coat, sand, protein, meat and blood residue. There are also mineral salts used for skin preservation. The color of the wastewater is gray, the smell is unpleasant, rotting, hydrogen sulfide is noticeable, but there is a few suspended solids. At this stage, the highest risk to humans may be the presence of anthrax bacteria. The source of infection is sick animals or products. However, at a later stage of the technological process, the microbes are destroyed (Bartkiewicz and Umiejewska, 2010).

After the liming, the effluent contains significant amounts of both dissolved and suspended calcium compounds, moreover large amounts of sodium and potassium sulfide, fur residues, fat and protein degradation products. The pH-index of this type of wastewater is in the range of 9–10. Wastewater after liming contains large quantities of suspended solids. This type of wastewater is considered as the most dangerous because of high concentrations of toxic substances: sulfur compounds, phenols, chlorides, chromium compounds (Bartkiewicz and Umiejewska, 2010).

After unhairing, the skin is rinsed. Wastewater after rinsing contain large amounts of insoluble matter, trace amounts of lime, sodium sulfide (used for skin preservation) and proteins. After the descaling (next stage of the process) the wastewater is characterized by the presence of soluble organic compounds and their pH is acidic and becomes neutral. Wastewater is gray, contains trace amounts of calcium, protein and ammonium compounds (Bartkiewicz and Umiejewska, 2010).

Wastewater after vegetable tanning is characterized by dark brown or gray-brown color. It can be noticed the presence of tannin compounds as well as organic acids. Such wastewater also contains a bacteria. The pH-index is acidic (4.5–5). Other non-tannin organic compounds, such as carbohydrates, may be noted (Bartkiewicz and Umiejewska, 2010). During chrome tanning, the wastewater has a slightly acidic pH, high concentrations of dissolved salts (mainly chromium (III)) and a significant amount of organic matter. After chrome tanning, the skin is neutralized and rinsed. Wastewater is greenish (exceptionally colorless), wide, variable pH range, slightly acidic to slightly alkaline. They may contain minor amounts of chromium salts and low amounts of suspended solids (Bartkiewicz and Umiejewska, 2010).

It should be highlighted, that except the wastewater generated during technological processes there are also wasted water from washing machines and devices. Also, drainage outflows from auxiliary departments should be added to the total volume of effluent from the production departments. This includes the glue factory and the hair cleaning unit (Bartkiewicz and Umiejewska, 2010).

3. Chromium - main pollution from the tannery industry

As the tanning industry progressed, the methods of tanning began to improve. According to Żarłok (2004), chromium compounds have been shown to be the best tanning reagent. In 1858 German scientist Knapp discovered the tanning properties of these compounds. In 1893 this idea had been patented in the USA and started using chrome baths in the tanning industry. At present, it is estimated that about 90% of tanning production is the leather from the chrome tanning.

Chromium in water occurs in two oxidation stages + III and + VI. Reducing and oxidation processes occur rapidly, so the total chromium concentration is determined. Chromium is rapidly precipitated, for example, in the form of hydroxide. In pure river waters, chromium concentrations should not exceed 0.5 μg/l. In groundwater, the chromium content is about 0.07–2 μg/l. Concentration over 100 μg/l in surface waters harms biological activity. It is estimated that about 35,000 t/yr of chromium goes to rivers and seas. The concentration of chromium in bottom sediments is about 50 mg Cr/kg (Bojanowska, 2000).

Problems with using chrome in tannery are wasted baths and solid waste containing this element. The wasted chrome solution cannot be discharged directly into the wastewater treatment plant. It could interfere with the biological processes. Solid wastes are generated during mechanical processing such as punching, molding of skin pieces. Protein solutions of chrome shavings are obtained, but their use is limited, and use as a fertilizer is forbidden (Żarłok, 2004). Due to the increase in the price of chromium tannins and the need to reduce the chromium concentrations in the sewage sludge, tannery industry is forced to use chromium salts sparingly and rationally. For the most efficient use of the chromium bath, the tanning process can be conducted at a higher temperature, in higher pH and with the prolonging of the process time. Another way is to support tanning with various special preparations. One of them is an aldehyde-based substance. Such substances are added in the tanning process or earlier during pickling. The same effect is obtained by adding magnesium oxide to the tanning bath, which increases the pH of the bath. Another way is the reuse of tannins in a closed circuit. After cleaning the bath from solids and replenishing it to the appropriate chemical composition, it is used again. Another solution is to collect all the chrome-containing baths from which the $Cr(OH)_3$ precipitates. Then the hydroxide is separated from water and dissolved before reuse. There is also a wet-white system to reduce chrome tanning. Raw skin is tanned with non-chromatic solutions with poor tanning effect (Żarłok, 2004).

4. Traditional methods of tannery wastewater treatment

4.1. Removal and recovery of chromium

Environmental quality standards should be achieved through the use of efficient methods of chromium removal from wastewater. This method including biological and physicochemical processes. Conventional methods for removing Cr(VI) include a chemical reduction to the Cr(III) by precipitation under alkaline conditions or removal by ion exchange and also adsorption reverses osmosis, electrochemical precipitation, bio adsorption, foam separation, separation by freezing and evaporation (Lkhagvadulam et al., 2017).

Chemical precipitation of chromium (III) is one of the traditional wastewater treatment methods for chromium removal. The most effective way is to collect wastewater which contains chrome (Religa and Żarłok, 2006). The most commonly used precipitate is sodium hydroxide. Chromium (III) hydroxide is then dewatered on filter presses. The sediment mass is subsequently dissolved in sulfuric acid (VI) and thus regenerated tannin, which can be reused (Religa and Gierycz, 2010). The appropriate pH and refill with a fresh tannin should be done before proceeding to further tanning (Religa and Żarłok, 2006). The highest influence on chromium precipitation efficiency has its concentration in wastewater. Therefore, it is a good solution to concentrate chromium compounds above 10 g/l (Religa and Gierycz, 2010). The chemical precipitation method removes chromium from the effluent to a value lower than 20 mg/l. The most significant problem is the sediments. Its composition is about 10% Cr (III), more than 70% water as well as various additives. In Poland, it is assumed that annually 18–20 thousand tons of this type of sediment are generated (Religa and Żarłok, 2006).

One of the most promising methods for removal of chromium from wastewater is adsorption. Sludge free treat operation is the undoubted advantage of this method, but from the other hand commercial available adsorbents are very expensive. To find efficient low-cost and locally available filter media as an adsorbent (e.g., pumice) tests are carried out (Birhanie et al., 2017).

Using of nanomaterials for removal of chromium from wastewater is very promising because some kinds of nanomaterials (such as carbon nanotubes, iron oxide, aluminum oxide, and titanium oxide) have showed excellent adsorption capacity for chromium and high surface area-to-volume ratio, surface modifiability, excellent magnetic properties, high biocompatibility, ease of separation using external magnetic field, reusability and comparatively low cost (Nirmala Ilankoon, 2014; Lkhagvadulam et al., 2017).

Another way to remove chromium from wastewater is the sorption on activated carbon. Utilization of sorbent consists of burning it with alkaline and then processing the chromium (zeolites, e.g., chondrites, flipsides). They remove chromium (III) by ion exchange. The process of the ion exchange system consists on the recovery of Cr (III) in an acidic cationite column, and next anionite one. In this way, chromate ions are stopped on the anion. The recovered chromium in both forms is suitable for the technological process after regeneration. The problem associated with this type of recovery of Cr (III) is the significant amount of saline that is strongly salted during regenerating columns (Bojanowska, 2000).

4.2. Sulfides removal

Wastewater after rehairing contains except the organic substances also significant amounts of inorganic compounds including sulfides. Consequently, it interferes with the biological wastewater treatment process (at values above 25mg/l Religa and Gierycz, 2010). Sulfides are also inconvenient due to odor nuisance. (Bartkiewicz and Umiejewska, 2010). The best way to remove sulfides is catalytic oxidation. The catalyst is manganese and the oxidant oxygen from air (Religa and Gierycz, 2010).

Another method can be oxidation with hydrogen peroxide. The duration of this type of process is estimated to be 6–10 days (Religa and Żarłok, 2006).

Another method of sulfides removal from wastewater is electrochemical oxidation. Wastewaters are in the iron electrode chamber, and effectiveness of this method is about 92% (Bartkiewicz and Umiejewska, 2010).

4.3. Coagulation

The coagulation process prepares tanning wastewater before biological treatment. BOD, COD and suspended solids can be removed in this process. The most commonly used coagulants are iron or aluminum sulfate. Then the whole is neutralized with lime milk (Religa and Żarłok, 2006). Another type of coagulant may be used, e.g., iron (III) chloride. It can be used for chrome water treatment. This coagulant influences the reduction of alkalinity of sewage, which influences the coagulation of colloidal protein substances. Where a vegetable coagulant such as iron sulfate (II) is used is not suitable for use. It forms with gallic acid (plant component) black coloring - ink (Bartkiewicz and Umiejewska, 2010). According to Religa and Żarłok, (2006) in wastewater after coagulation, up to 90% of impurities are inorganic substances, including but not limited to chlorides, sulfates and sulfides. These substances are not only a source of nutrients for active sludge microorganisms, but also endangers its development (Felicjaniak and Przybiński, 2003).

New types of coagulants were also developed, such as poly aluminium chloride (PAC), polyaluminium silicate (PASiC) and polyaluminium ferric chloride (PAFC) ($[Al_2(OH)_nCl6_{-n}]_m$. $[Fe_2(OH)_nCl6_{-n}]_m$) for improvement the coagulation efficiency and for minimizing residual coagulants in the effluent (Gao et al., 2004). For example, coagulation with PAFC in optimal conditions has resulted in high removal rates of COD (> 75%) (Lofrano, 2006).

4.4. Biological methods

Pre-purified effluents are easily biodegradable during biological treatment. Contaminated wastewater is introduced into the activated sludge vents. The efficiency of the biological wastewater treatment process using the activated sludge method in the removal of residual chromium in about 94% (Religa and Żarłok, 2006). However, the high concentration of tannins and other poorly biodegradable compounds as well as metals can inhibit biological treatment (Stasinakis et al., 2002; Farabegoli et al., 2004)

There is a possibility to support biological treatment - biopreparations can be used. These compounds contain unique bacteria and enzymes supported decomposition of organic matter. They also affect the transformation of inorganic substances. Biopreparations are safe for humans, plants, and animals and do not contain genetically modified microorganisms. An example could be isolated *Pseudomonas aeruginosa, Bacillus flexus, Exiguobacterium homiense,* and *Staphylococcus aureus* from soak liquor, marine soil, salt lake saline liquor and seawater. The biodegradation of tannery soak liquor by these halotolerant bacterial consortia allowed to achieve significant removal of COD (80%) at 8% (w/v) salinity for mixed salt tolerant consortia (Senthilkumar et al., 2008).

Biopreparations are often used in case of high chloride concentrations. Activated sludge is severely affected by high levels of chlorides causing the death of microorganisms. Activated sludge biocoenosis is poorer regarding species and species composition. Chloride ions destructively affect the protozoa, which in the results in changes in the shape of the floc and its outflow from the settlers. Influence the viability of microorganisms could be achieved by adding biopreparation into the activated sludge. Thanks to this, high concentrations of chloride ions do not lead to the death of protozoa. They also adversely affect the efficiency of wastewater treatment with activated sludge nitrogen compounds (Mendrycka and Stawarz, 2012). High concentrations of ammonium nitrogen in wastewater are toxic to nitrifying bacteria. This is related to the possibility of occurrence of a nonionized form - ammonia. Ammonia is toxic to *Nitrobacter* bacteria, which makes it difficult to oxidize nitrites to nitrates (Felicjaniak and Przybiński, 2003). According to Mendrycka and Stawarz (2012), it was noted that the addition of biopreparations solved the problem with nitrogen compounds. The amount of total nitrogen decreased by approximately 300mg/l compared to the untreated sample. Biopreparations have also helped in reducing COD (about 90%) and BOD (80%).

It is possible in case of biological treatment to use the wastewater treatment batch method in sequential reactors. It is suitable for small tanneries that do not have large areas. The amount of wastewater cannot be large due to the state of wastewater retention. Moreover, so in the first reactor is the oxygen phase, filling the tank, mixing - duration is 16 hours. In the second reactor, there is a hypoxia and mixing phase - about 5 hours. Subsequent sedimentation 2 hours and a 1-hour downtime. The main problem with this type of treatment is the lack of phosphorus and the high concentration of ammonium nitrogen and its nitrification products. This means that $N\text{-}NH_4$ concentrations should be reduced in earlier processes. This can be done directly in the technological process of skin tanning. Another option would be to use a hydrobotanic treatment plant using plants with high nitrogen requirements. Another solution would be slow filtration after biological sandblast cleaning, or the use of ion exchange (anionics) method (Felicjaniak and Przybiński, 2003).

Another type of biological treatment may be the use of a submerged bed reactor. Due to the fine dispersion suspended solids and hardly removable impurities (especially COD) it should be terminated by sorption on activated carbon with the polyelectrolyte. For the process to proceed correctly it is necessary to protect the microorganisms. It may be done by the use of a multilayer biological membrane on a submerged bed. The biological reactor is a denitrification chamber, nitrification and secondary settling tank. In the settling tank, a flocculation chamber should be isolated to add polyelectrolyte. The use of simultaneous sorption on activated carbon at the end of the process not only proves to be a low-cost solution but also meets the standards for the discharge of wastewater into the surface water. (Kotulska and Ozonek, 2005)

5. Membrane techniques in tannery wastewater treatment

Membrane techniques have been practiced for many years in the treatment of tanning wastewater. However, single technique or systems connecting (e.g., two processes) are mainly used. Interest in membrane techniques in this industry is related to the possibility of recovery of chromium after tanning baths, as well as the reduction of charge pollution after degreasing and dehairing, salt removal and decoloring (Cassano et al., 1997; De Pinho, 2009; Wang et al., 2011, Gierycz et al, 2008). By using membrane techniques as crossflow microfiltration (MF), ultrafiltration (UF), nanofiltration (NF), reverse osmosis (RO) and supported liquid membranes (SLMs) can be recovery of chromium from spent liquors (Fabiani et al., 1996; Ashraf et al., 1997; Cassano et al., 2001; Labanda et al., 2009; Religa and Gierycz, 2010) and reuse wastewater and chemicals of the deliming/bating liquor (Gallego-Molina et al., 2013; Religa and Gierycz, 2010). Post-treatment to remove refractory organic compounds could use reverse osmosis (RO) with a plane membrane (De Gisi et al., 2009).

5.1. Micro- and ultrafiltration

The experience of Rajewski et al. (2009) using a microfiltration membrane installation has shown that the process itself has no advantage in the treatment of tannery wastewater.

Microfiltration is, therefore, used for pre-treatment by removal of suspended solids before ultrafiltration (Gierycz et al., 2008). This solution is safe; protects RO, NF, UF modules against fouling and consequently membranes damage (Religa and Gawroński, 2006).

In the tannery industry, ultrafiltration can be used to treat worn tanning baths. It is possible to regenerate baths after dehairing and degreasing (Gierycz et al., 2008). For dehairing, sulfide-lime baths are mainly used. Sulfide penetrates through the membrane. In this way, the resulting permeate can be reused as a dehairing bath. In contrast, the retentate containing hair and other organic compounds should be directed to biological treatment. The permeate after the ultrafiltration process is used again to give the skin the same properties as in the case of the fresh bath (Religa and Gierycz, 2010). Hair, fat, and protein are stored in the retentate. The consumption of sulfides used for re-bathing is limited to 2.5% (with initial consumption of 10%) (Gierycz et al., 2008).

In case of ultrafiltration only, fouling cannot be avoided. In this case, it is advisable to clean the membrane with compressed air or to use a pre-filter on the screens (Religa and Gierycz, 2010) or the previously mentioned microfiltration. Similarly, regeneration of the degreasing baths is similar. According to Gierycz et al. (2008), the retentate obtained after filtration through the ultrafiltration membrane can be reused at the finishing stage. Water which was used for coloring skins can be treated by ultrafiltration. This is a legitimate and valid solution because the fat emulsion particles are associated with the dye. The quality of the purified water obtained after this process meets the requirements of water used for rinsing colored skins (Gierycz et al., 2008).

UF process with the best efficiency can remove turbidity. Chromium is not sufficiently removed. Chromium tanning bath can be treated using a hybrid system. Such a system is a combination of membrane technology with another traditional process such as chemical precipitation (Bodzek et al., 1997). By using a hybrid system with micro- and ultrafiltration membranes, pre-purified wastewater from suspended solids and fats is obtained. Approx. 98% of fat and 84% of suspended solids are retained. Protein substances are also retained in about 40%. Chromium ions are not retained due to penetration of the pores of the membranes. In this way, the permeate after the ultrafiltration process is used again as a tanning bath. However, it should be supplemented to have the right chromium concentration. The retentate is directed to the biological wastewater treatment chamber (Religa and Gierycz, 2010, Gierycz et al., 2008).

5.2. Nanofiltration

The process that can be used to separate chromium from wastewater is nanofiltration. In this way, it can be obtained a permeate that will contain chloride salts and high chromium (III) retentate. Permeate can be used again as a skin pickling bath. Conversely, the retentate should be concentrated to obtain chromium (III) hydroxide. The precipitate is then mixed with sulfuric acid, and the whole is re-used for tanning. Re-obtaining the tanning baths limits tannin consumption, costs associated with it, and negative environmental impacts (Religa and Gawroński, 2006; Kowalik et al., 2009). This solution is also beneficial because of the removal of toxic chromium from the wastewater. Reusing the pickling bath reduces chloride concentration in wastewater. The regenerated and re-used tanning solution has comparable properties as freshly prepared tanning baths (Religa and Gierycz, 2010). Nanofiltration has also been used to treat wastewater after skin liming. This makes it possible to remove sulfides from the limestone. Studies have shown that turbidity is eliminated completely, salts dissolved by 82.5% and COD by 95.4% (Gierycz et al., 2008).

5.3. Reverse osmosis

The reverse osmosis process could be used for the recovery of chromium from wastewater. Because the RO membrane does not pass on monovalent ions, they are concentrated in front of the membrane, thereby increasing the osmotic pressure. This results in a drop in the permeate stream. The economic course of the process is badly affected by the high temperatures, and the high transmembrane pressure applied. RO process is advantageous when chromium concentration does not exceed 1 g/l and salt concentration 5 g/l (Religa and Gawroński, 2006). Approximate values are obtained after dehydration of chromium (III) hydroxide. The filtrate can be fed into the RO module. In the permeate obtained, the salt concentrations do not exceed the limit value when discharging wastewater into water or soil (Gierycz et al., 2008).

5.4. Liquid membrane

A relatively new technique for the separation of two liquid phases is the liquid membrane system. The chapter consists of the difference in the diffusion rate and the solubility of the substance in solution. However, it is more efficient to introduce into the carrier membrane, thereby forming a complex with one of the components. This technique can be used to recover chromium from tanning wastewater. (Religa and Gawroński, 2006). According to Religa and Gawroński, (2006). The first method is chromium (III) oxidation to chromium (VI). Subsequent ion separation by a liquid membrane, which as carrier contains tri-n-octylamine. During this research, 100% of chromium was removed. Still, this method is not used in industry, because the cost of chromium oxidation is very high, and leads to environmental hazards. Economic considerations are the cost of apparatus as well as the energy consumption of the process. Due to environmental hazards, it should be mentioned that higher concentrations of chromium (VI) ions are toxic and have mutagenic and carcinogenic properties.

The second method is the direct removal of chromium (III) from wastewater. In this case, another type of media was used. For the conditions of the process, both pH and temperature are important. The speed and efficiency of the process are mainly affected by the carrier concentration and the initial chromium concentration. However, additional studies are needed to determine how other ions in solution will behave, and how it will affect the speed of chromium removal. Therefore, as is evident in the desire to use liquid membranes for the removal of chromium from tanning wastewater, the correct choice of carrier and its concentration are important (Religa and Gawroński, 2006).

6. AOPs methods

There has been increasing number of researchers on advanced oxidation processes (AOPs) to treat tannery wastewater. During the AOPs processes, there are used strong oxidizing reagents (H_2O_2, O_3) and/or catalysts (Fe, Mn, TiO_2). Sometimes chemicals are supported in activity by UV light (Schrank et al., 2004). These processes are based on the production and utilization of hydroxyl radicals which are known as powerful oxidants that quickly and unselectively oxidize a wide range of organic compounds. The scientific interest towards AOPs application to high strength wastewater has increased remarkably in the past 20 years.

Fenton based processes are achieved by the reaction of H_2O_2 with ferrous (Fe^{2+}) and ferric (Fe^{3+}) iron in acidic aqueous solutions. The reaction generates highly reactive hydroxyl radicals (Gogate and Pandit, 2003). The maximum catalytic activity of $Fe^{2+}/Fe^{3+}/H_2O_2$ is at pH = 2.8–3.0 (Lofrano et al., 2007a). At pH-index higher than 5, the ferric ion precipitates as ferric hydroxide and at lower pH-index (<2.8), the complexation of Fe^{3+} with H_2O_2 is inhibited (Kuo, 1992). The low pH-index, relatively high temperature (43–45 °C) and the high presence of aromatic compounds, especially in the streams of retaining baths are attractive to use Fenton oxidation processes (Lofrano et al., 2007a, b; 2010).

7. Summary

Wastewater generated by skin tanning is heavily polluted. They have a high concentration of organic and inorganic pollutants, which make them difficult to treat. The composition of the wastewater is varied due to changes in the daily cyclicality of their discharge. Fluctuations in the composition also result in the type of tanned skin. The characteristics of wastewater depend on the stage of production. The other compositions have after tanning - they are much more toxic than salted wastewater after rinsing the raw material. Large quantities of water are needed in the tanning, resulting in significant amounts of wastewater. Proper organization and management, closed circuits allows reducing the demand for water. The introduction of untreated tanning wastewater into the receiver is unacceptable. The mixed industrial wastewater with municipal wastewater badly affects the municipal wastewater treatment plant. Particular attention should be paid to the chromium compounds found in tanning wastes, which are toxic substances. Chromium can accumulate in the bottom sediment of surface water, which negatively affects the life and development of aquatic organisms and aquatic plants.

The paper presents traditional methods of chemical precipitation of chromium or sorption on activated carbon. At present, these processes can be assisted with biopreparations, closed circuit applications, and wet-white systems. Each of these processes helps to re-use the chrome tanned tan. Chromium precipitation is also a pre-treatment of the effluent so that, at further stages of biological purification, the activated sludge is protected from chromium toxicity. Wastewater treatment with activated sludge can be supported by biopreparations. Thanks to them, the sludge's life is longer, and the chloride ions contained in the wastewater do not cause the death of microorganisms. The problem is excessive sediment after wastewater treatment. The search for methods of chromium sludge management, new methods of recovery and recycling are ongoing.

One of the most promising methods for removal of chromium from wastewater is adsorption. Sludge free treat operation is the undoubted advantage of this method, but from the other hand commercial available adsorbents are very expensive. To find efficient low-cost and locally available filter media as an adsorbent (e.g., pumice) tests are carried out (Birhanie et al., 2017).

Nanomaterials which are used for chromium removal are very promising because of excellent adsorption capacity for chromium and allow for a high degree of recovery.

Nowadays membrane techniques are more often used for the elimination of toxic components of tannery wastewater. Using only the microfiltration process for chromium recovery does not affect. It is useful only for pre-treatment of wastes. After ultrafiltration, it is obtained a permeate ready for reuse as a tanning bath. Thanks to the combination of microfiltration and ultrafiltration, it is not observed such intense fouling as in case of ultrafiltration itself. Good use for regeneration of the chromium bath is a hybrid system with aeration of tanning wastewater. The ultrafiltration process is also used to regenerate degreasing and dehairing baths.

Also after the nanofiltration process, good quality chrome bath is obtained, while at the same time, permeate with high salt content. This solution can be re-used for skin picking. The new technique is the use of the liquid membrane for the recovery of chromium from wastewater. However, there are high costs involved, including chromium (III) to chromium (VI) oxidation equipment. It also adversely affects this process of oxidation on the environment, because chromium (VI) compounds are mutagenic and carcinogenic. There are also significant amounts of water needed for the tanning, so closed circuits and wastewater treatment from individual processes are advisable.

The tannery wastewater contains high load of undegradable compounds. One of the best treatment processes for such wastewater are AOPs techniques. The scientific interest towards AOPs application to high strength wastewater (and tannery wastewater as well) has increased in the past 20 years. Although these are promising technologies, they require constant control of parameters such as the pH of treated wastewater. They also generate significant amounts of sludge that must be utilized. Therefore the technology is expensive and it requires a lot of experience.

Various physiochemical techniques used for wastewater treatment can be applied to tannery wastewater, but these processes are expensive. Small tannery, because of the costs, do not modernize their wastewater treatment methods. Therefore, the search for additional traditional wastewater treatment is indicated. Biological wastewater treatment is more favorable and cost-effective as compared to other physiochemical methods for removal of color and organic contents. However, some cases require the use of chemical and physical methods and depend on the process efficiency. The integrated application of a combined physical or chemical process with a biological process to treat tannery wastewater would give satisfactory results compared to individual processes of treatment.

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On additional possibilities of using the blue-green algae substrate and digestate

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Abstract

The factors of natural water surface algal bloom formation have been considered, the dominance of *Microcystisa eruginosa* blue-green algae representative has been noted. The conditions for lipid extraction from cyanobacteria biomass before and after methanogenesis in the natural environment have been experimentally determined, the yield to lipid production depending on the environment temperature and acidity has been compared. The possibility of using Folch method for lipid extraction by improving the existing method has been explored. It has been determined that from blue-green algae biomass it is possible to extract lipids, lipoids and phycobiliproteids as useful substances for the biofuels and biologically active compounds production.

By the substrate and dihestrate physical and chemical characteristics determination, the possibility of their application as biofertilizer has been experimentally proved. The safety of dihetestatus blue-green algae use as biofertilizer, starting from 1:100 dilution, with the use of the biotesting method in the *Daphniamagna Straus* test-object for various dilutions (1:10, 1:50, 1:100, 1:200, 1:500, 1:1000) at ambient temperature has been tested. Possibilities of blue-green algae biomass and activated sludge from urban wastewater treatment facilities mixture methanogenesis have been estimated. It has been shown that in one methanogenesis cycle 2−2.5 times less biogas is released than directly from the blue-green algae substrate. The lipids volumes that can be extracted from cyanobacteria during natural and artificial water reservoirs algal bloom and used for the various biofuels production have been determined.

Keywords

cyanobacteria, lipids, activated sludge, methanogenesis, biofuels

1. Introduction

At the present time there is an urgent problem of technogenic and chemical pollution of natural water surface with specific organic substances. In this regard, considerable attention is paid to researches aimed at organic contamination preventing or limiting, in particular due to the massive development of blue-green algae (BGA) in artificial reservoirs. Flares of "algal bloom" increase the natural waters toxicity level, worsen the reservoirs conditions, suppress the hydrobionts and adjacent biotopes inhabitant vital activity.

The annual seasonal process of "algal bloom" and the hydrobionts ensuing obscurity lead to a thorough analysis of the environmental conditions effects on water production, eutrophication mathematical modeling, and the development of technological options for environmental problem solving (Pasenko et al., 2016). So far about 40 representatives of the toxicogenic cyanobacteria genera, including *Microcystis, Anabaena, Nodularia, Nostoc, Aphanizomenon, Oscillatoria, Cylindrospermopsis*, are known. However, the main battery of organic matter while the Dnipro "algal bloom" is *Microcystisa eruginosa*, a representative of photosynthetic cyanobacteria. That accounts for up to 90% of biomass in the bloom spots − the cyanates largest accumulation places in the reservoir.

In connection with the mentioned above, BGA extraction from the water reservoirs of the Dnipro cascade (Yelizarov and Yelizarov, 2011) and the use of biomass for further processing, it will not only provide an additional nutrients source (Sirenko et al., 1978; Nykyforov et al., 2016), but also will improve the natural water surface and drinking water quality in particular, that is served to riverside areas settlements.

The problem of releasing reservoirs from the BGA surplus can be considered as allowing the targeted use of a biomass natural producer containing valuable food, feed, medical, pharmaceutical, perfumery, agricultural and forestry important components (Nykyforov et al., 2016b) (Fig. 1).

The aim of the work is to determine the possibilities of blue and green algae substrate and digestatus using, to determine the optimal conditions for the lipids extraction from BGA biomass with Folch method using, the determination of the biologically active substances presence in the substrate and the dihestate.

2. Materials and Methods

Lipids are a group of low weight molecular organic compounds (fats and fatty substances) which are extracted from cells by nonpolar solvents. The energy is one of the main functions of fats. The fat content of the cell is 5−15% by weight of dry matter. In adipose tissue cells, the amount of fat increases up to 90%. Accumulating in animal subcutaneous tissue cells, in seeds and plants, fat is a reserve energy source.

Lipids and lipoids also perform the construction function, because they are part of cell membranes. Due to poor thermal conductivity, fat is capable of performing a heat-insulating function (Orel, 2007).

The lipids content in blue-green algae varies in a wide range. In *Anabaenacylindrica* and *Oscillayoriasp* lipids make up 2−12%, *Aphanizomenon flos-aquae* it is 3.75% of the organic part. Up to 21% of all lipids are α-linoleic acid in the gelatolipids form. Sulpholipids are also found. A characteristic feature is that the unsaturated fatty acids synthesis results in the removal of the hydrogen atom with the unsaturated high molecular weight compounds formation. During the blue-green algae studies, digalactosil diglycerides, sulfonohynosil-diglyceride, phosphatidyl-glycerine, etc. have been detected **(**Skorokhod, 1990)**.**

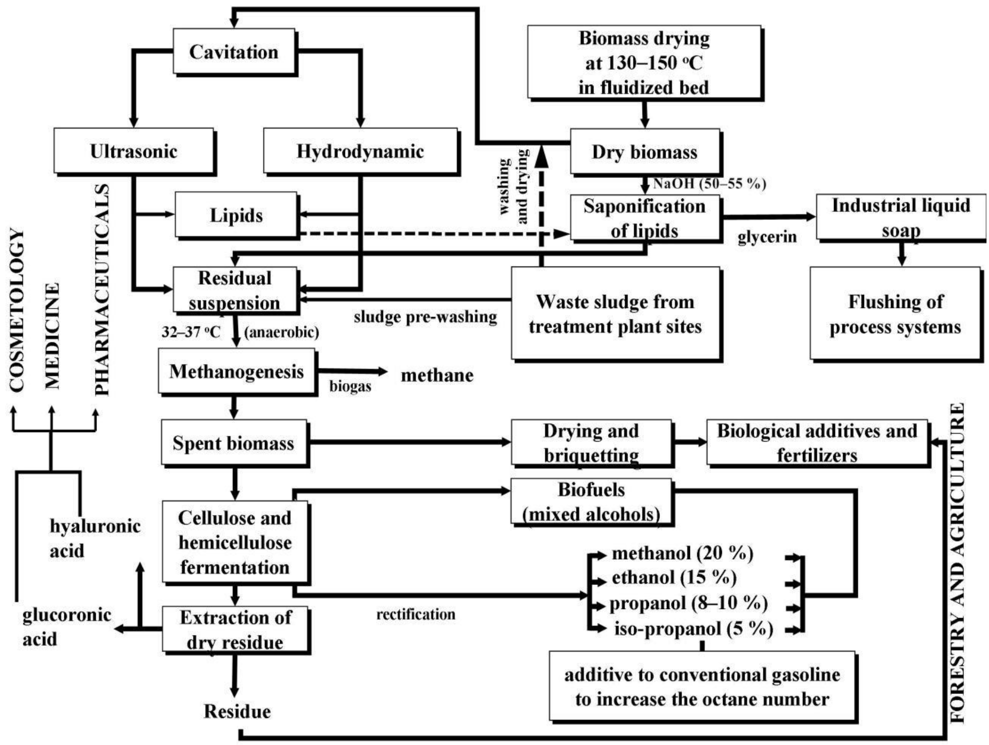

Fig. 1. Biotechnological ways of blue-green algae complex processing and the industries of its product application

However, there are no phosphatidyl-choline, phosphatidyl-inositol, phosphatidyl-ethanolamine in BGA cells. In this case, the intracellular synthesis of each lipid differs according to the kinetic and thermodynamic parameters and characteristics. Nearly in all blue-green algae species, γ-linolenic acid is present in the mono- and digalactosil diglycerides fractions (Skorokhod, 1990).

2.1. Physical and chemical properties of cyanobacteria biomass

Due to facts mentioned above (significant scale of seasonal growth of BGA biomass and a wide range of lipid content in it), the authors have used various methods for lipid extraction, in particular, the method of total lipid fraction quantitative determination or Folche method. It is known that this method is usually used in medicine to determine the overall lipid fraction in biological tissues and liquids of living organisms (Orel, 2007). But the literature does not know the application of this method for the lipid extraction from *Microcystisaeruginos*a biomass (Fig. 2).

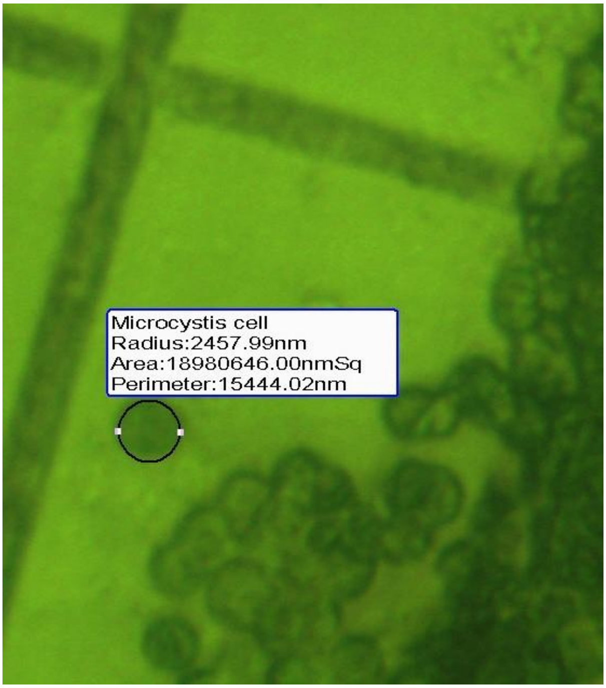

Fig. 2. *Microcystis aeruginosa* on Goryaev camera with smallest square sides definition (0,05 mm)

To compare the completeness of lipids extraction from cyanobacteria (Balanda et al., 2011; Chisti, 2007; Harwood and Jones, 1989; Kyle, 1992) and the application routes, biomass was subjected to anaerobic fermentation, during which a constant gas release was recorded, with chromatographic analyzes that showed that its composition (%) is close in composition to natural gas and does not have components that are harmful while combustion: Methane − 85.26; Nitrogene – 10.36; Oxygen – 2.90; Ethane – 0.95; Propane − 0,33; other gases − 0.20 (Yelizarov and Yelizarov, 2011). The installed calorific value of this biogas was approximately 33 MJ/m^3. environmental clean water from the west cSv complemented by energy-saving. Thus, ecological water purification from the BGA is complemented by energy-saving method. Additional useful product is the biomass that remains after the algae fermentation. Our experiments results show that it can be successfully used as agricultural fertilizer, and they, as it is demonstrated by their chemical analysis (%), do not contain heavy metals (with the exception of a small amount of copper), and therefore, they can be used as fertilizer not only for technical purposes, but also for food crops: C 54,66; O 37,81; Si 0,61;P 1,34; S 0,69; Cl 0,22; K 2,14; Ca 1,92; Cu 0,6 (Yelizarov and Yelizarov, 2011).

It has been noted that in the summer the water in the bottom layers of the middle Dnipro reservoirs gets a reddish color. Our studies have shown that this is a purely optical effect caused by the particles formed during the cyanobacteria decomposition. The water has reddish color only in the reflected light, whereas in the passing light there is pure blue color, which is due to the features of light scattering on suspended particles, the size of which is proportional to the wavelength. The cyanobacteria saturated water eventually loses this reddish color. Tracking of such water reflection and transmission spectra is a source of information on what stage the cyanobacteria decomposition is (Yelizarov and Yelizarov, 2011).

This phenomenon is due to the presence of ficobile proteins in the blue-green algae, the number and composition of which depend on the environmental conditions, and this, in the turn, makes the reason the amount of sequestered lipids depends on. Among the factors that influence the ficobiliproteins formation intensity and their relationship, is the importance of light spectral composition and its intensity: green illumination accumulates ficeroetrin, and in red light it is ficocyanin. Depending on the nutritional conditions, the phycobiliproteins quantitative content varies. With Nitrogen fasting, the ficheretrina content falls sharply, and the addition of Carbohydrates and Nitrogen causes ficoerythrin appearance. Ficobile protein quantitative content in blue-green algae also depends on the temperature and pH of the environment.

3. Results and Discussion

The lipid extraction has been carried out from the BGA substrate, digestate, and dry substratum in two ways: by the classical method (Nykyforov et al., 2016b) and by the modified method, where methanol has been replaced with ethanol, and instead of chloroform, benzene has been used. The experiments have been carried out at a temperature of 14–15°C, pressure of $1{,}013 \cdot 10^5$ Pa, a BGA biomass sample of 350 mg. The results obtained (Tab. 1) indicate the feasibility of replacing solvents, since the ethanol-benzene mixture forms an azeotrope, which contributes to the complete lipid extraction due to a deeper penetration of the ethanol and benzene molecules lipid layer.

Table 1. Results of the experiment on lipid extraction of blue-green algae biomass

№ test	**Mass of extracted lipids, mg (± 0.05)**		
	Substrate	**Digestate**	**Dry substrate**
1	8.0	13.5	6.0
	6.5	13.0	6.0
	5.0	10.0	5.4
2	5.0	9.5	4.0
	6.0	9.5	4.5
	6.5	9.5	3.5
3	5.0	15.0	6.0
	6.5	20.0	5.5
	7.0	22.0	4.5
4	4.5	21.5	2.5
	4.0	20.0	3.0
	3.5	16.0	1.5
5	5.5	11.0	4.7
	7.5	10.0	3.6
	4.0	9.5	5.0
average	5.6	13.8	4.05

Due to the latter, there is a reorientation of the electronic density in the surface layer atoms and molecules; there is a transition from the hydrophilic properties of the lipid molecules substituents to the hydrophobic ones. As a result of the phase marked on the surface, accumulation of lipid molecules in the form of bubbles, pellicles, and individual finely divided particles (Fig. 3) is observed.

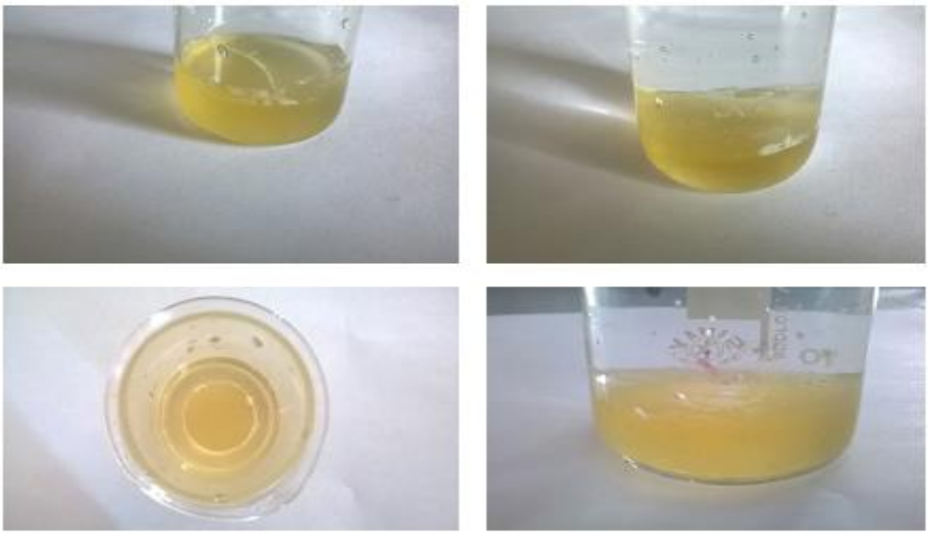

Fig. 3. Lipid formation while extracting of blue-green algae biomass

Analysis of the results obtained shows that the content of lipids extraction from the substrate varies in the range of 1.4−3.5%, which corresponds to the data obtained by the authors (Harwood and Jones, 1989). From the dry substrate it was not possible to isolate the lipids as much as possible as the moisture content influences the completeness of their extraction: at 85−90% moisture content, the mechano-dynamic movement of blue-green algae cell membranes changes, and the mechanical resistance decreases due to the water molecule dipoles pressure on the cell membranes surface.

The lipids content extraction from dihestat is 2.2–3 times higher than from wet and dry substrates. This fact requires more thorough study, since most of the literature resources indicate the opposite: the processed substrate after removing the generated biogas can extract lipids by almost two times less than from the actual substrate before methanogenesis.

3.1. Determination of blue-green algae biomass composition

In order to solve the problem of lipid extraction using Folche method, within the framework of the Ukrainian-Austrian project "Methods for processing cyanobacteria biomass causing reservoirs bloom" at the laboratory of the Institute of Natural Resources and Life Sciences of the University of Vienna, the inorganic composition of BGA biomass has been analyzed before and after methanogenesis (Tab. 2.).

Table 2. Results of cyanobacteria biomass analysis (July-August 2017)

Sample	**Dry matter**		**Substrate**	**Digestat**
1	**2**	**3**	**4**	**5**
Sample processing				
Drying temperature [ºC]	105ºC	105ºC	liquid	liquid
Stabilization of samples	–22ºC	–22ºC	–22ºC	–22ºC
Water content	99.0	99.2	–	
Quick test of dry matter content	0.52	0.39	–	–
pH (in water)	7.1	7.5	–	–
Electrical conductivity, mS/cm	2.67	3.88	–	–
CHC [g/kg]	–	–	10.43	9.43
NH_4 - N [mg/kg]	3.4	6.8	340	550
NO_3 - N [mg/kg]	0	0	1	2
N (Kieldahl method)	9.1	10	900	800
Loss on ignition	84.8	79.0	–	–
C (org.) [%]	40.1	37.1	–	–
C/N	4	4	–	–

Table 2. - continuation

1	2	3	4	5
Lower carboxylic acids [mg/dm^3]				
Acetic acid	–	–	84	<20
Propionic acid	–	–	92	<20
Isobutyric acid	–	–	204	43
Butyric acid	–	–	<20	<20
Isovaleric acid	–	–	<20	<20
Valerian acid	–	–	<20	<20
Total content of volatile fatty acids	–	–	380	43
Other organics				
Gluconic acid racemate	–	–	98	91
Lactose	–	–	21	<20
Galacturonic acid	–	–	25	<20
Xylose	–	–	<20	<20
N-acetyl-D-glucosamine	–	–	30	<20
Glycerin	–	–	36	<20
Humic substances				
Fulvic acid	95	65	745	365
Humic acids	90	110	680	600
Humic substances (amount)	185	175	1425	965
Infrared spectroscopy	160	62	–	–
AT4Sapromat [mg O_2/g]	–	–	1750	530
Results of analysis for additional parameters				
P	0.4%	6.6%	43 mg/dm^3	33 mg/dm^3
K	1.1%	1.5%	115 mg/dm^3	120 mg/dm^3
Ca	0.7%	0.4%	70 mg/dm^3	30 mg/dm^3
Mg	1.3%	1.3%	130 mg/dm^3	105 mg/dm^3

The high content of fulvic and humic acids, as well as nutrients − phosphorus and potassium − allows us to assert that the use of blue-green algae digestat can be a source for useful nutrients and biologically active substances obtaining. In particular, a study on the development of biotechnology for the hyaluronic acid extracrion has begun, the chemical structure of experimental specimens has been confirmed with the use of IR and PMR spectroscopy.

Consequently, blue-green algae are the source of biologically active substances that can be used not only in agriculture for the feeds vitaminization, but also for biotechnological and nanoprocesses. In this case, the biologically active substances of blue-green algae can be delivered to specific tissues cells and organs using magnetic particles − carriers of these biologically active substances − to form the desired therapeutic effect. However, this block requires further focused researches.

3.2. Results of obtained fertilizer biotesting

Previous studies on the blue-green algae complex processing (Zagirnyak et al., 2017) have proved the possibility of obtaining biofuels for agricultural and forestry farms from waste biomass. For further application an integral part of the system of quality assessment and control of aqueous solutions for various purposes, including cyanobacteria substrate exhausted after receiving biogas, is been biotested.

Biotesting has been carried out using a Daphnia magna Straus test-object, which was carried out in test tubes using a substrate in various dilutions (1:10, 1:50, 1:100, 1:200, 1:500, 1:1000) at indoor temperature. As a result, the safety of the use of blue-green algae digestatus as biofertilizers, in dilution starting from 1:100, has been proved.

To determine the seed germination degree, as one of the criteria for evaluating the possibility of using the blue-green algae residues after methanogenesis as a fertilizer, four types of cultivated plants have been used: sowing cucumber ("Competitor" variety) – *Cucumis sativus* L., radish ("Heat") − *Raphanus sativus* L. *var. radicula*, garden cabbage (Langdeycker Dauer variety) – *Brassica oleraceae* L., tomato (San'ka variety) – *Solanum lycopersicum* L. During the research, the most favorable concentration of the exhausted substrate obtained after the blue-green algae biomethonogenesis has been determined for germination (during this period the temperature was +25°C, pH = 6.0). As a result, the effect of the exhausted substrate on the germination quality has been detected; the germination degree of different cultures at different digestat dilutions has been calculated. The similarity has been determined in percentage of seed sprouted from 100 seeds in comparison with control (bidistillate) in two repetitions. The research results are shown in Fig. 4.

The dependence of the germination intensity on the biofertilizer dilution for radish, tomato and cucumber can be given as a mathematical expressions:

- for radish:

$$y = -0.25 \cdot x^3 + 3.035 \cdot x^2 - 12.71 \cdot x + 108.6; \qquad R^2 = 0.963 \qquad (1)$$

- for tomato:

$$y = -1.125 \cdot x^3 + 10.37 \cdot x^2 - 28.0 \cdot x + 100.5; \qquad R^2 = 0.813 \qquad (2)$$

- for cucumber:

$$y = -6.666 \cdot x^3 + 67.46 \cdot x^2 - 209.8 \cdot x + 222.6; \qquad R^2 = 0.990 \qquad (3)$$

For cabbage, the general dependence form has the following form:

$$y = 1.291 \cdot x^3 - 12.73 \cdot x^2 + 36.47 \cdot x + 10.1; \qquad R^2 = 0.990 \qquad (4)$$

where: R^2 – the coefficient of approximation reliability.

The difference in the equation of cabbidge germination from the remaining specimens can be attributed to the seeds quality, the deterioration of which results from violatation the storage condition, expiration, poor-quality seed treatment. Consequently, it can be stated that for cabbage, the optimal biofertilizer dilution is from 1:100 to 1:200.

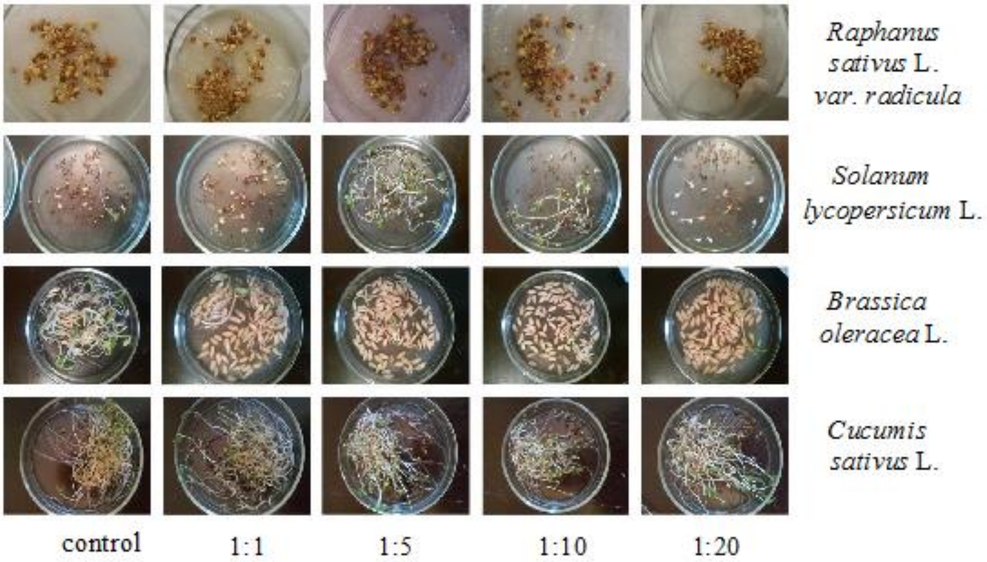

Fig. 4. Germination intensity of different cultures depending on the biofertilizer dilution intensity

Re-testing is required for cucumber, since in the experimental conditions, recommendations regarding the concentration of biofuels can not be given. Results of tomato and radish biotesting are close in their dependence of the sprouts rate on the biofertilizerdilution. At the same time for tomato, according to the analysis results the optimal dilution is 1:50. In our opinion, the result for radishis rather interesting: any dilution gives a positive effect of viability and germination compared with the control.

3.3. Possibilities of lipids extracting from BGA biomass and activated sludge mixtures of urban waste water treatment facilities

In the framework of complex studies of the various organic substances properties and their suitability for use in energy purposes, a comparative analysis of the three types of substrates used in the biogas production technology has been conducted: the active sludge of the treatment facilities of the Communal Enterprise "Gorvodokanal" of Kremenchuk, the concentrated blue-green algae biomass, as well as their mixtures in the ratio 1:1 (Fig. 5).

For this puprose, three weights of 100 g of a sludge and blue-green algae mixture were dried, and an average of 3.36 g of dry mixture was obtained. In the course of laboratory studies, the following parameters of the analyzed organic substrates have been established: a) pH of substrates in the methane tin: blue-green algae biomass – 6.65; a substrates mixture – 6.6; b) the density of active sludge is ≈ 1 g/cm^3; blue-green algae biomass ≈ 0.91 g/cm^3; substrates mixture ≈ 0.98 g/cm^3.

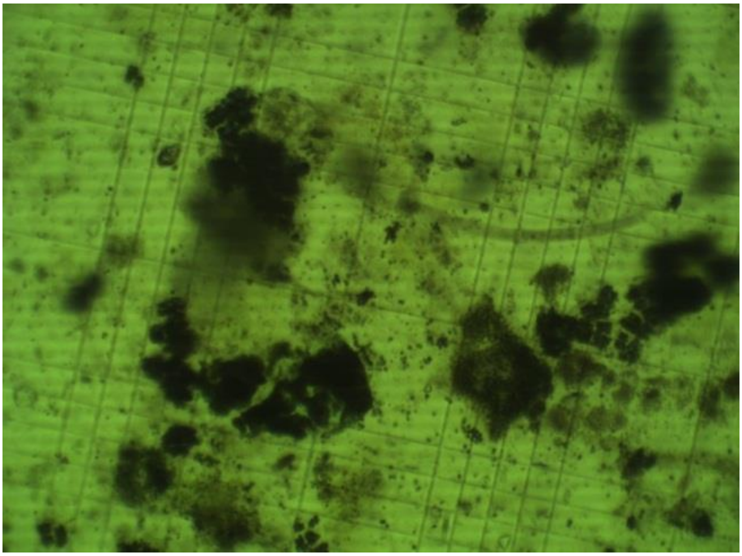

Fig. 5. General view of the active sludge and *Microcystis aeruginosa* (Kützing) Kützing biomass mixture of on Goryaev camera

To obtain a dry residue, the liquid concentrate was being dried for 92 hours at an average daily temperature of +28°C. It has been found out that the mineral components are the following: in the active sludge – 0.82 g; in blue-green algae biomass − 2 g; in a substrates mixture − 3 g (due to the summation effect).

In order to determine the ash elements content in the obtained dry mass, three weighing weights of 10 g were successively burned in a muffle furnace during 40 minutes at 800^{o}C. As a result, it has been proved that a blue-green algae biomass and active sludge mixture contains mineral and organic components in the ratio of approximately 1:1.

In the course of the combined methanogenesis, the biomass collected in the concentration column was 1.16 m^3 (1158 dm^3) in the ratio of 62 dm^3 of aqueous solution of blue-green algae biomass to 1096 dm^3 of active sludge from water treatment facilities. It has been noted that in water BGA biomass and sludge behave differently − blue-green algae concentrate in the upper layers of the solution, and the sludge, on the contrary, in the lower layers of the solution.

Conducting methanogenesis of the active sludge and *Microcystis aeruginosa* (Kützing) Kützing biomass mixture has shown that the process is much slower. Only 70 dm^3 biogas has been created in one cycle. At the same time, factors limiting the process should include the high environmental acidity (Tab. 3), low value of Nitrogen chemical consumption, high content of dry mineral and organic components.

Table 3. Physical and chemical parameters of investigated substrates

Sample	**Dry substance**	**100 g Active sludge from the treatment facilities**	**Substrate**
Date of selection	14.08.2017	16.08.2017	17.08.2017
Stabilization of samples	–22ºC	–22ºC	–22ºC
pH (in water)	–	5.8	5.99
CHC [g/kg]	–	207.3	65.05
N (Kieldahl)	69.12	9.25	5.27
Total amount of dry matter	90.12	33.95	6.06
Dry organic substance	78.80	14.97	5.43

4. *Summary and Conclusions*

It is proved (Sirenko et al., 1978) that water "algal bloom" is a consequence of anthropogenic influence of eutrophication as an appropriate adaptive reaction of the latter and, apparently, as a new stage of its existence in the changed environment − biohydrocenosis succession. Phytoplankton qualitative and quantitative composition of Kremenchug reservoir eutrophied zones has been studied.

It has been stated and experimentally confirmed that the main battery of organic matter during the bloom period of Kremenchug and Dneprodzerzhinsk reservoirs and the Dnipro River is the representative of photosynthetic cyanobacteria – *Microcystis aeruginosa* (Kützing) Kützing. It has been shown that the use of blue-green algae biomass bio-conversion can reduce the degree of environmental risk of harmful products exposure, and, accordingly, reduce the level of biohydrocenosis ecological danger under the influence of environment adverse external factors.

With the use of the classical Folch method and its modifications it has been determined the percentage of lipid fractions in the substrate and digestrate. It has been proved that a chloroform and methanol mixture contributes to the destruction of the blue-green algae cell membrane due to the chemical bonds breakdown in the "sugars-sphingosides", "organic polyspirids-sphingosides" complexes and with the sphingosides release.

The preliminary results of the spectral analysis have shown that the lipids of biological tissues and animal origin fluids in the chemical structure differ from the lipids extraction from the plant material, in particular blue-green algae (60% glycerides and 40% sphingosides). Appropriate biotechnological processing of plant sphygosides reveals a way to obtain biologically active compounds and other target products.

As a result of biotesting, that has been carried out using a *Daphnia magna* Straus test-object, the safety of blue-green algae digestatus using as biofertilizers in dilution starting from 1:100, has been proved.

During the research, the most favorable concentration of the exhausted substrate obtained after the blue-green algae biomethanogenesis has been determined for seeds germination (during this period the temperature was +25°C, pH = 6.0). The optimal biofertilizer dilution for cabbage is from 1:100 up to 1:200. Results of tomato and radish biotesting are close in their dependence of the sprouts rate on the biofertilizer dilution. At the same time according to the analysis results for tomato the optimal dilution is 1:50.

Researches have shown the practical possibility of using for the production of biogas the three types of substrate – the biomass of blue-green algae, activated sludge and their mixture.

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Applying expanded polysterene filters with increasing layer of suspended sediment in reagent water softening technological schemes

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Abstract

Calcium and magnesium ions are among the major impurities in water and in many ways it determines the possibility of its using for various national economic objectives. When we use natural water, during processes of evaporation or decreasing solubility with increasing temperature, it can be occurred hard soluble calcium and magnesium salts excretion on heat transfer surfaces in the solid form, scilicet scale. This phenomenon can lead to reduced equipment performance, the rising production cost, equipment damage and even dangerous accidents.

Existing water softening methods have a number of disadvantages: demand high process passing temperatures, what causes significant energy costs; some reagents are quite expensive; other can cause appearance of additional compounds, which supposed to be removed or neutralized.

Liming and lime-soda ash are the most spread ones. The main device of such technological schemes, sludge-blanket clarifier, has such disadvantages as quite difficult construction and exploitation. Nowadays existing edifices work with much lower productivity combined with designed. So there is a need in finding new and improving already existing water softening methods and technologies.

In this paper we propose to use liming method on expanded polystyrene filters with increasing layer of suspended sediment for water softening. They are already well known for water deferrization in works of professor Valerii Orlov and associate professor Sergii Martynov, but not for water softening.

Here is also described methods and materials for analytical water and suspended sediment layer characteristics what we applied in our work, showed the section of existing clarifier with suspended sediment, laboratory and industrial experimental plants, calculated values of main suspended sediment layer characteristics.

Results of our research showed maximum effect for total hardness decreasing of 83% in laboratory conditions, in industrial - 56%. Using of liming method on expanded polystyrene filters with increasing layer of suspended sediment allows to simplify existing water softening technological scheme, excluding clarifiers with suspended sediment; to simplify exploitation for staff; in building new edifices it requires smaller buildings size, what saves space; it let to make energy savings, by

excluding using pumps for backwash. The proposed method can be used in preparation of water at enterprises, technological schemes of which use steam-power property equipment, that require water of the proper quality.

Keywords

water hardness, water softening, liming method, suspended sediment layer, expanded polystyrene filling, calcium and magnesium ions

1. *Introduction*

Water hardness is due to multivalent metallic ions, primarily calcium and magnesium. Hard water is generally not harmful to human health; however; it can cause scaling problems in domestic and especially in industrial systems (Brown et al., 2012). In water treatment practice a number of methods of water treatment with reagents are used for the purpose of association contained cations – calcium and magnesium – in low soluble compounds, which are removed later from the water by precipitation and filtration.

Liming and lime-soda ash are the most spread ones (Kliachko and Apeltsyn, 1971). For this may be used pellet softener systems such as pellet reactor, what is basically a fluidized bed of grains on which the crystallization of calcium carbonate takes place. Advantages of the pellet reactor are its small size, low installation cost, and rapid treatment. Removing magnesium in these systems is difficult, that's why they should not be considered when the treated water has high magnesium content. Also upflow clarifier, specifically the sludge-blanket clarifier, is often used in water softening operations (AWWA, 2016). The sludge blanket clarifiers are solids contact clarifiers with a distinct solids layer that is maintained as a suspended filter through which flow passes. The sludge blanket unit contains a central mixing zone for partial flocculation and a fluidized sludge blanket in the lower portion of the settling zone (Howe et al., 2012). As the small, coagulated particles enter the sludge blanket, contact with other particles in the blanket causes flocculation to occur. The floc grows in size and becomes part of the blanket (Oregon health authority, 2017). The section of sludge blanket clarifier type VTI-1000I is presented in Fig. 1.

This edifice has such disadvantages as quite difficult construction and exploitation. Nowadays existing edifices work with much lower productivity combined with designed. So there is a need in finding new and improving already existing water softening methods and technologies.

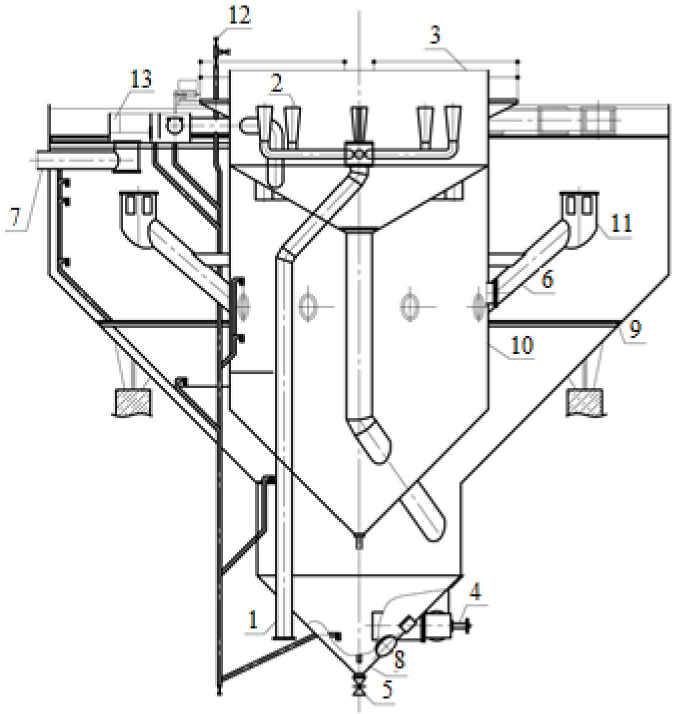


Fig. 1. Clarifier type VTI-1000I:
1 – initial water pipeline, 2 – switchgear, 3 – air separator, 4 – nozzle device, 5 – pipeline for removing coarse particles by purging, 6 – sludge outlet pipeline, 7 – pipeline for removal of mixture: treated main stream water and water which is cleansed of sludge, 8 – sludge box, 9 – horizontal perforated partition, 10 – sludge thickener, 11 – sludge receiving window, 12 – pipeline for cleaning sludge thickener, 13 – receiving switchgear box

2. Materials and Methods

For helding our work we used such scientific methods as theoretical, experimental, analytical and statistical.

In this paper we propose to use liming method on expanded polystyrene filters with increasing layer of suspended sediment for water softening. This type of filter (Fig. 2) was discovered by professor Valerii Orlov and associate professor Sergii Martynov at National University of Water and Environmental Engineering, Ukraine. Their research is engaged with using it for water deferrization, but not for water softening (Orlov et al. 1999).

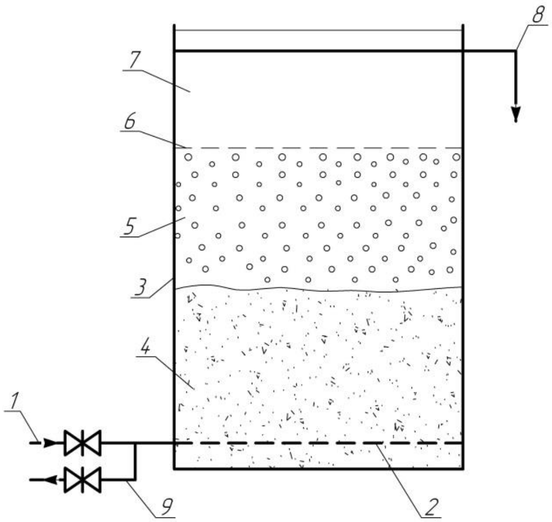


Fig. 2. Scheme of expanded polystyrene filter with increasing layer of suspended sediment: 1 – water supply, 2 – down distributive system, 3 – frame, 4 – increasing layer of suspended sediment, 5 – expanded polystyrene filling, 6 – holding grate, 7 – above filtering area, 8 – purified water outflow, 9 – washing water outflow

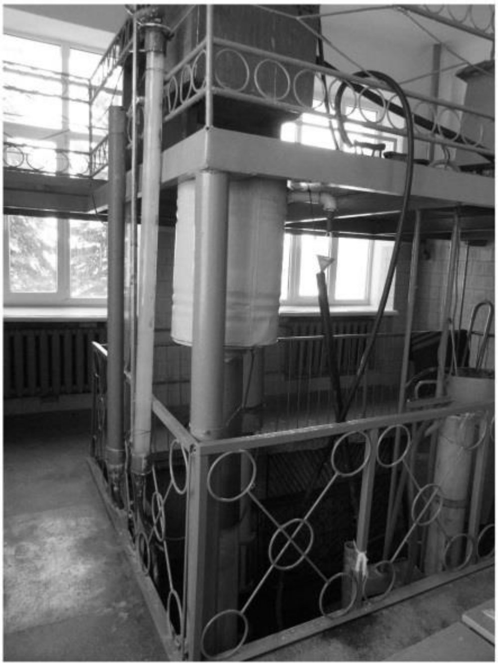
Experimental research process is the following. Water mixes with lime in the contact tube and then passes firstly through a layer of suspended sediment which is performed gradually increasing, and after – floating expanded polystyrene backfill (Orlov et al. 2013).

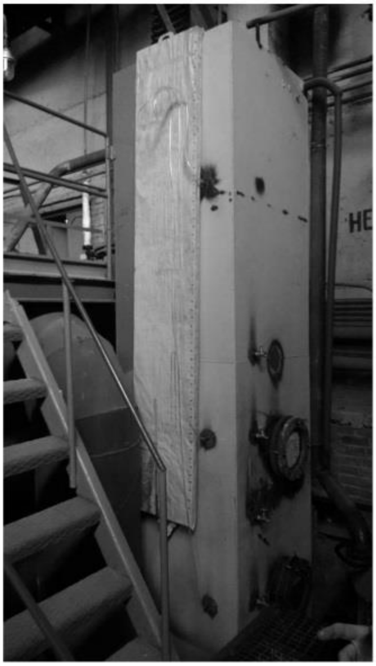
For determination of main analytical characteristics we use following methods. For total hardness determination we put 100 ml of investigated water sample in volumetric flask. Then we add there 5 ml of ammonia buffer and a bit of indicator “Eriohrom black T”. Mix well till dissolving of crystals and appearance of violet colour. Then we titrate the resulting solution by adding trilon B (0.1 n) till appearance of blue colour. The quantity of trilon B (0.1 n) in ml will be the total hardness value.

For alkalinity determination we take 100 ml of investigated water sample and mix it with indicator “Methyl orange” till appearance of yellow colour. Then titrate the mixture by adding hydrochlorid acid till colour changing to bright orange.

3. Results and Discussion

We have conducted research of softening process on experimental plant in laboratory and industrial conditions. Experimental plants for research of softening process are presented in Fig. 3a and Fig. 3b. During filtering cycles we controlled analytical characteristics of water such as: pH, total hardness alkalinity and also filtering rate, pressure losses. We also discovered characteristics of suspended sediment layer.





a) b)

Fig. 3. Experimental plants for research of softening process:
a) in laboratory conditions, b) in industrial conditions

3.1. Results of analytical characteristics determination

The filtering rate range was between 2.5 and 4.5 m^3/hour. In laboratory conditions we served tap water on plant with such characteristics: total hardness – 5.8 mmol/dm^3, alkalinity – 6.10 mmol/dm^3, pH – 7.40. As reagent we used slaked lime OP-1 ISO B V.2.7-90-99 ("Ferezit", Lviv), what we prepared directly in laboratory, and received maximum effect 83% (decreasing of total water hardness from 6.0 to 1.0 mmol/dm^3). In industrial conditions we served river water with such characteristics: total hardness – 7.1 mmol/dm^3, alkalinity – 4.8 mmol/dm^3, pH – 8.10, used lime from lime preparation workshop with concentration of calcium hydrate 4% and received maximum effect 56% (decreasing of total water hardness from 7.0 to 3.0 mmol/dm^3).

To show the difference in laboratory and industrial conditions the results are presented in Fig. 4 and Fig. 5.

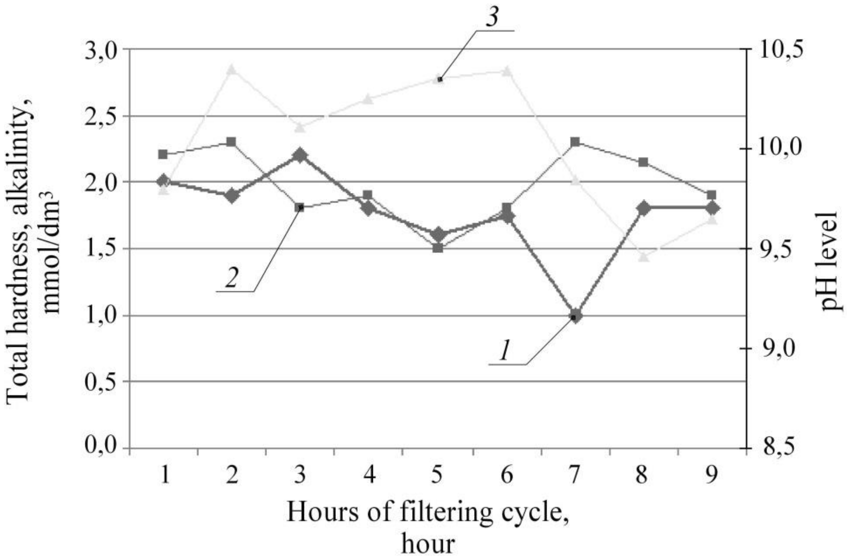


Fig. 4. Total hardness, alkalinity dependence of ph level during filtering cycle with filtering rate 3,5 m/hour in laboratory conditions:
1 – total hardness in filtered water, 2 – alkalinity in filtered water, 3 – pH level in filtered water

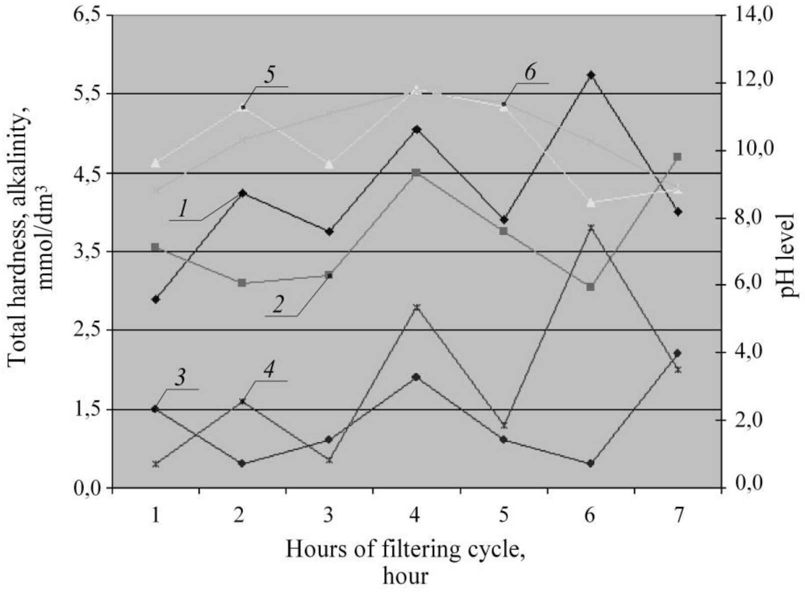


Fig.5. Total hardness, alkalinity dependence of ph level during filtering cycle with filtering rate 3,5 m/hour in industrial conditions:
1 – total hardness after suspended sediment layer, 2 – total hardness in filtered water, 3 – alkalinity after suspended sediment layer, 4 – alkalinity in filtered water, 5 – pH level after suspended sediment layer, 6 – pH level in filtered water

3.2. Suspended sediment layer characteristics determination

We discovered characteristics of suspended sediment layer due to method presented by Kurgaev (1977).

Samples were taken to several glass cylinders, filling them with a layer of water with sediment to different heights of 100 to 400–600 mm. After this, we observed the settling of sediment until the moment of stabilization of the upper boundary position of sediment. After measuring the final height of the sediment we find the solid phase content by weight in these samples. In Fig. 6a, Fig. 6b and Fig. 6c, it is presented the change of height of suspended sediment sample at the beginning and in time of measuring.

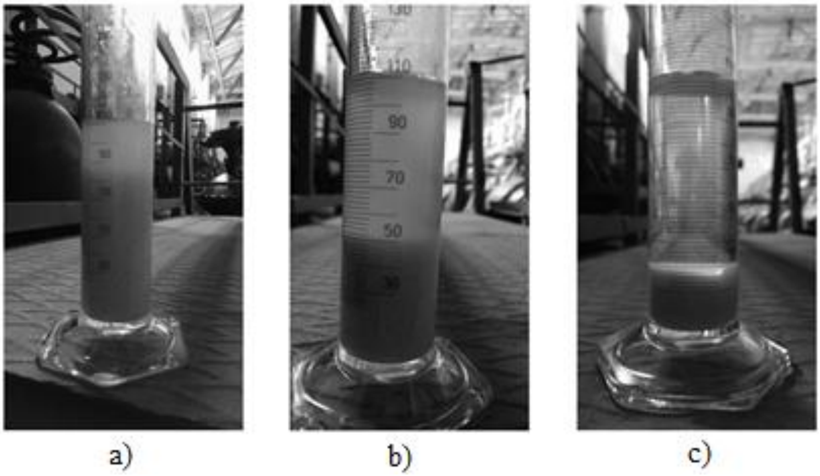


Fig. 6. Change of height of suspended sediment sample:
a) at the beginning, b) and c) in time of measurement

For solid phase mass determination in sediment we filtered sample through filtering paper and defined the value after drying it at temperature of 105 °C.

To determine specific concentration [g/cm³], we used Eq. (1):

$$\gamma_1 = \frac{G}{h \cdot f} \tag{1}$$

where: G – solid phase mass [g], h – the final height of sediment layer [cm] and f – square of cylinder section [cm²].

Solid phase content by weight in volume unit of unsaturated sediment, g/cm³, can be determined by Eq. (2):

$$\gamma_0 = \frac{G}{W_{st}} \tag{2}$$

where: W_{st} – volume of unsaturated sediment [cm³].

Volume concentration, cm, can be determined by Eq. (3):

$$C_0 = \frac{h_n}{h} \quad (3)$$

where: h_n – initial height of sediment [cm].

Density of unscathed flake-like suspension, expresses the mass content in one unit of its volume of solid phase and water, which is contained in the cells of the frame of the flakes, g/cm³, can be determined by Eq. (4):

$$\gamma_{02} = \gamma + \gamma_0 \cdot \left(1 - \frac{\gamma}{\gamma_m}\right) = 1 + \gamma_0 \cdot \left(1 - \frac{\gamma}{\gamma_m}\right) \quad (4)$$

where: γ_m – density of solid matter, which forms frame of sediment flakes [g/cm³].

Value that characterizes the ratio of the volume of water, that is included in the cells of the suspension flakes, to the volume of the solid phase in these flakes can be determined by Eq. (5):

$$\Gamma_0 = \frac{\gamma_m - \gamma_0}{\gamma_0} \quad (5)$$

Ratio of water content and solid phase by mass can be determined by Eq. (6):

$$\Gamma_e = \frac{\gamma}{\gamma_0} - \frac{\gamma}{\gamma_m} \quad (6)$$

Equivalent diameter, cm, can be determined by Eq. (7):

$$d_{eq} = 0.136 \cdot \sqrt{\frac{\nu \cdot v_{sed} \cdot \alpha_1}{\gamma_{02} - 1}} \quad (7)$$

where: ν – kinematic viscosity of water [cm²/s], v_{sed} – sedimentation rate [cm/s] and α_1 – particle shape coefficient, which for a flake-like sediment is equal to 1,65–2.

Suspended sediment is a polydisperse system in which the rate of particles precipitation depends on the volumetric sediment concentration. The chemical composition of sediment consists of calcium carbonate and magnesium hydroxide compounds. Calcium carbonate has characteristic of condensation-crystallization structure. Magnesium hydroxide relates to the coagulation type. Magnesium hydroxide prevents direct connection and growth of calcium carbonate (Kurgaev, 1977).

There is a constant accumulation of sediment, which may negatively affect the contact area, because some part of the sediment "ages", reaching the top of the suspended layer and subsequently descents. Remove of sediment from the contact area is necessary to maintain the height of the suspended layer of solids and preserve optimal physical and chemical characteristics of the contact zone.

The calculated values of suspended sediment layer characteristics are presented in Tab. 1.

Table 1. Characteristics of suspended sediment layer due to calcium carbonate and magnesium hydroxide particles

№	Characteristic, units of measurement	Value
1	Solid phase mass [g]	0,038–0,518
2	Specific concentration [g/cm^3]	0,018–0,12
3	Sedimentation rate [cm/s]	0,02–0,05
4	Solid phase content by weight in volume unit of unsaturated sediment [g/cm^3]	0,00036–0,00264
5	Density of unscathed flake-like suspension [g/cm^3]	1,0002–1,0016/ 1,0002–1,0015
6	Value that characterizes the ratio of the volume of water, that is included in the cells of the suspension flakes, to the volume of the solid phase in these flakes	1021–7514/ 908–6679
7	Ratio of water content and solid phase by mass	378–2783
8	Equivalent diameter [cm]	1,13–1,87/ 1,18–1,95

4. Summary and Conclusion

Proposed method - liming on expanded polystyrene filters with increasing layer of suspended sediment - is new in water softening technologies and shows positive reducing effect on water quality: total and alkalinity. Using of it allows to simplify existing water softening technological scheme, excluding clarifiers with suspended sediment; to simplify exploitation for staff; in building new edifices it requires smaller buildings size, what saves space; it let to make energy savings, by excluding using pumps for backwash. This method is good for industrial water supply use.

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Biotechnology of preparation and using of sewage sediment as a fertilizer-meliorant

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Abstract

In this work the agrotechnical effect of biotechnology on the use of sewage sludge as a fertilizer on medium and slightly acidic soils and environmental safety indicators of the implementation of this agrochemical event is researched. Based on the results of conducted field, single-unit, stationary, short-term, single-factor agrotechnical experiments with independent monitoring programs, fertilizer-meliorant on the basis of sewage sludge (excessive activated sludge of urban wastewater treatment plants and calcium sludge water treatment of the heat power plant) may be recommended as an alternative for use on medium-acid and slightly acidic soils, because it effectively at the level of the traditional chemical ameliorant (lime) improves the properties of the soil and stimulates the biological otsesy in the soil, a positive effect on yield and technological parameters of agricultural crops – sugar beets. The main technological indicator of culture – sugar content improves when applying the proposed fertilizer by an average of 3–5%. It was revealed that the presence of calcium-containing sludge in the granular fertilizer enhances the positive effect of active sludge as an organic component of fertilizer on the fertility of medium and slightly alkaline soils. Changes are due not only to physical and chemical processes in the soil, but also to the biochemical activity of microorganisms of activated sludge and soil. The study of soil samples by the method of biotesting («Growth test») showed that the use of fertilizer-meliorant on the basis of sewage sludge on moderate and slightly acidic soils, taking into account the dosage norms, does not increase their phytotoxic effect on the test culture.

Keywords

biotechnology, fertilizer-meliorant, sewage sludge, agrotechnical effect, phytotoxic action.

1. Introduction

There are many ways to solve the problem of utilization of dewatered sewage sludge (SS). The promising direction is the use of SS in agriculture as organic fertilizers. Introduction of SS in the soil improves the chemical composition, structure of the arable layer of soil and provides increased yields of agricultural crops. Soil is the basis of the existence and productivity of agricultural and natural ecosystems. Land resources of the planet, and soils are one of their components, exhausting, vulnerable and slowly regenerating. In recent decades, more and more attention has been paid to the quality and sustainability of soils when used by humans. Soil quality should be regulated in order to optimize their productivity, therefore considerable attention in the management of agro ecosystems is concentrated on the practice of fertilizer use.

In the structure of agricultural lands in Ukraine there are significant areas of acidic soils. Increased acidity of soils negatively affects their fertility and microbiological activity (low content of nutrients, increased number of toxic compounds). pH – the index is most important in assessing the soil fertility, because it determines the solubility, hence the potential availability or phytotoxicity of nutrients and other elements of the soil, the relative biological activity of plants and soil microorganisms. For most potentially toxic trace elements (Cd, Ni, Pb) the solubility increases with increasing the acidity of the soil solution. Other processes that influence the behavior of elements in soils (cation exchange, sorption, desorption) also depend on pH, increase solubility and toxicity of Al, Mn, Fe. The availability of N is also reduced in conditions of increased acidity, because bacteria capable of nitrification are more active in a neutral or slightly acidic environment. Acid soils (pH $<5,5$) lose fertility (Muha et al., 2003). Neutralization of the soil environment eliminates these harmful phenomena.

Therefore, topical applied issues are the search for non-traditional sources of fertilizers and chemical meliorants that are not related to the high costs of material and energy resources (Vlasova et al., 2013, Peskarev et al., 2011). Such sources include waste from various enterprises. An important positive effect of introducing SS as a fertilizer on the soil is to reduce its acidity, increase the content of humus substances. Precipitated soils are precipitated by lime or contain calcium-containing compounds. From the standpoint of resource conservation in the work, it was proposed to obtain organo-mineral fertilizers on the basis of excess activated sludge of urban wastewater treatment plants and calcium-containing sludge of water purification of a heat and power plant (EPP) as an analogue of natural lime. In previous works, positive results of the use of EPP as a chemical reclamation, organo-mineral fertilizer or a component of composite fertilizers in the cultivation of acidifying soil crops, were obtained (Pasenko, 2008, Pasenko et al., 2016). The sludge was used in a complex with various organic fertilizers: humus, bird droppings, peat, siderates, compost, vinegar. The best results were mixtures: sludge + bird droppings, sludge + coke, sludge + compost. On low acid soils, the yield of crops with a significant and average requirement for liming is increased by 20%, the resistance of plants to the disease of bacterial and viral etiology increases. The aim of the work is to investigate the agrotechnical effect of the technology of SS application as a fertilizer-reclamation medium on medium and slightly acidic soils and indicators of environmental safety of the implementation of this agrochemical event.

2. Materials and Methods

2.1. Recovery of SS-based reclamation fertilizer

For the fertilizer, dewatered sewage sludge was used: excess activated sludge CE «Kremenchukvodokanal» and the waste water of the Kremenchug EPP PJSC Poltavaoblenergo.

The technology of dehydration of excess activated sludge at urban wastewater treatment facilities involves reducing the moisture content of sludge on sludge sites naturally due to gravity forces and their further decontamination.

The sludge waste water treatment, which is a heterogeneous mixture of components of various aggregate states (dispersed and dispersive phases), is designed according to the developed technology to dewater on EPP within the technological cycle. Technical water, which is thus removed, can be reused in the process, reducing the cost of water raw materials and the volume of discharges of the enterprise. In the conditions of minimization of energy efficiency of the process of deep removal of moisture from sludge wastes in the technological scheme of processing of the specified wastes in the work environmentally and economically expedient way of their dehydration with the use of secondary energy resources of the enterprise – the heat of exhaust flue gases. The developed technological scheme involves the use of hydrodynamic methods for dewatering waste: the gravitation of slurry pulp under the action of gravity to 90% moisture and filtration of the slurry suspension under the influence of pressure difference through the filter material on the filter presses FPACM-25. The final dewatering of sludge precipitate to 5–10% humidity is provided in the scheme by the method of thermal drying in the utilization of the heat energy of the exhaust gases and the water vapor formed (Pasenko, 2013).

Dehydrated SS (excess activated sludge and EPP water purification sludge) was mixed in proportion 2:1, respectively, to produce ammonium fertilizer. SS contain the necessary nutrients for plants and, by their agrochemical value, are not inferior to traditional fertilizers. And the presence of calcium-containing compounds in the waste sludge EPP determines the process of neutralization of the acidic reaction of the soil solution in the application of the received fertilizer-melliferous.

2.2. Agrochemistry of fertility restoration of soils at application of fertilizer-meliorant on the basis of SS

Experiments on the restoration of fertility of degraded soils were carried out on slightly acidic and medium acid soils. The amount of fertilizer application in soil was calculated based on the value of hydrolytic acidity of the studied soils, taking into account the background content of heavy metals. The research was conducted in laboratory and field experiments. Soil samples from an arable layer with a depth of 0,25–0,30 cm were selected according to DSTU 4287:2004 «Quality of soil. Sampling». The combined sample of each soil consisted of 5-point samples. Selected soil samples were dried at room temperature to obtain air-dry soil. The average sample of each soil sample was formed by the quarantine method. To investigate the acidity of the soil, a «skeleton» was removed from the fraction «fine earth» in accordance with DSTU ISO 11464-2001 «Soil quality. Preliminary processing of samples for physico-chemical analysis», DSTU ISO 10381-6-2001 «Quality of soil. Sampling Part 6. Guidelines for the selection, processing and storage of soil».

The dose of chemical meliorant for deoxidation of soils was determined by the magnitude of the formula (1):

$$D = \frac{0{,}5 \cdot H_G \cdot S \cdot h \cdot d}{1000} \tag{1}$$

where: D – dose of $CaCO_3$ [t/ha], H_G – hydrolytic acidity of soil, mg-ek/100 g of soil, S – area of the territory [ha], h – depth of the arable layer [m], d – volumetric weight of soil [g/cm^3].

As a calcium-based material for acidifying the soil, an alternative fertilizer-based meliorant based on SS was used, and as a control, lime was used as a traditional chemical meliorant. The specified fertilizer-meliorant contains a certain amount of $CaCO_3$. In calculating the physical norm for the application of calcium-containing materials, corrections were made for the moisture content, the impurities and the inactive $CaCO_3$ (based on the toning of grinding).

2.3. Agrotechnology of SS application as fertilizer-reclamation

Field experiments on the use of meliorant fertilizers on the basis of SS were laid on the territory of the experimental sites of the Kozelshchansky district of the Poltava region. According to different schemes, single, stationary, short-term, single-factor agrotechnical experiments were carried out on independent monitoring programs, in which the effect of introducing fertilizer on yields of agricultural crops under the conditions of the unchanged agrotechnical background of soil for 2 years was studied. To conduct experiments, selected sites were typical for the specified region and homogeneous with genetic characteristics and properties of soils with a previously known history. Selected experimental areas are rectangular, small to reduce the error of the experiment area and the same form, the number of which is the product of the number of variants for repetition. Experiments to reduce the experimental error are laid out in 4-fold repetition, because 4–6-fold territorial repetition makes it possible to more accurately take into account each variation of the experiment qualitative difference of the land. For placement of variants in the field the most commonly used rendering method of placing repetitions on sections of the experimental site was used. For the laying of experiments an agronomically important agricultural crop is used, is prone to liming, is sensitive to changes in soil fertility and is more resistant to adverse weather conditions: sugar beet. For the fertilization, the methods of the pretreatment method were used: under pre-sowing cultivation, under pre-sowing plowing and under silt plowing. Calculation of doses of fertilizer-meliorant based on SS was carried out taking into account the soil's need for liming based on the hydrolytic acidity of soils, taking into account the granulometric and chemical composition of the sludge (in terms of $CaCO_3$). The initial acidity of the soils of the experimental sites was in the range of 5,3–6,0 mg-ek/100 g of soil, therefore, liming with the use of sludge was carried out as reclamation (on soils with $pH_{KCl} < 5{,}5$) and supportive ($pH_{KCl} > 5{,}5$). SS was used as a fertilizer-meliorant. Taking into account the 30% content of organic matter, EPP water treatment sludges were administered in a half dose in a scheme with organic fertilizer – an excessive active sludge. The harvest from the experimental areas was collected manually from the entire experimental area. Determination of sugar content and technological qualities of sugar beet roots was carried out on the automatic line «Venema». The mathematical processing of the experimental data obtained was carried out by the method of dispersion analysis. The agricultural conditions and facilities involved in the experiments are described below.

2.4. Investigation of phytotoxic activity of soils by the «Growth test» method after the introduction of SS as a fertilizer-meliorant

The method of biotesting was used – «Growth test» according to Gorovoy A.I. technique (Gorova et al., 2004). In the growth test, the intensity of growth of the indicator plant, whose seeds are sprouted on the investigated and control soil, control the following indicators:

- energy of germination of plants;
- height of seedlings;
- the length of the roots;
- dry mass of the above-ground and underground part of germs, etc.

Preference is given to test crops that grow quickly and are typical of the region. In this work a test culture with small seeds was used – radish Raphanus sativus var. sativus L. The seeds were purchased at the Veselo-Podolian research breeding station, that is, the seeds were calibrated with pre-sowing processing. To carry out the growth test, the method of propagating test culture in Petri dishes was used. In assessing the toxicity of soil samples in a Petri dish, place a sheet of filter paper, pour 1 g of soil prepared for analysis, which is evenly distributed over the cup. Then add 5–7 ml of water and planted 30 seed of the indicator plant on the soil. Experiment at 25 ^{0}C in a thermostat lasted 5 days. The control substrate in this case was the soil under study without the addition of fertilizer-meliorant. After the experiment, plants are carefully taken out of Petri's cups, they measure the length of the root and stem system of the germs, determine the dry mass of seedlings after drying. The research of all variants was carried out in three replicates.

The obtained experimental data were processed by the method of dispersion analysis according to the following formulas of statistical processing:

- average arithmetic mean value:

$$\bar{x} = \frac{\sum_{i=1}^{N} x_i}{N} \tag{2}$$

where: $\bar{x}$ – arithmetic mean of the indicator, x_i – value of the indicator in the variant, N – number of observations;

- dispersion:

$$\sigma^2 = \frac{\sum_{i=1}^{N} (x_i - \bar{x})^2}{N-1} \tag{3}$$

- error of arithmetic average:

$$m = \sqrt{\frac{\sigma^2}{N}} \tag{4}$$

- statistics to measure the significance of the results:

$$t = \frac{\bar{x}_1 - \bar{x}_2}{\sqrt{m_1^2 + m_2^2}} \tag{5}$$

where: index 1 – control, index 2 – an option.

3. *Results and Discussion*

In this work, agrotechnical indicators of the use of fertilizer-meliorant based on SS were investigated. The trial was laid on sections of 0,90 m x 3,75 m; the plow with fertilization was carried out at a depth of 27–30 cm using the tractor MTZ-82 and the plow PN-3-35. The agrochemical characteristics of the soil (typical black earth of a powerful, low-humus, large-peal medium-grained) according to the

«Observatory fertility» were: pH_{KCl} – 5,8–6,0 mg-ekv/100 g of soil; humus content 3,6–4,2%; nitrate nitrogen content 22–24 mg/kg of soil; moving forms of phosphorus – 26–29 mg/kg; potassium – 114–152 mg/kg. Sowing of sugar beet was carried out in a month by a manual drill in the norm of seeding 19–20 pieces, seeds per 1 running meter; after a week they fixed the appearance of stairs. Accounting for the density of beet seedlings was carried out in the phase of 2–3 pairs of true leaves – on the 23rd day of the experiment after sowing. The results of harvest accounting, determination of sugar content and technological qualities of sugar beet root crops are given in Table 1.

Table 1. Yield, technological parameters of sugar beet

№	**Fertilizer**	**Number of steps [pcs/p.m.]**	**Number of roots [ths. pcs/ha]**	**Mass of hooks [t/ha]**	**Weight of root crops [t/ha]**	**Succidity [%]**	**Potassium content [mg-eq/100 g of pulp]**	**Sodium content [mg-eq/100 g of pulp]**
1	Activated sludge + Sludge	7,4 ±0,20	112 ±2,1	18,1 ±0,5	42,3 ±0,5	14,95 ±0,31	3,97 ±0,06	2,10± 0,03
2	Activated sludge	6,1 ±0,11	108 ±2,0	19,1 ±0,3	37,2 ±0,4	14,60 ±0,28	3,45 ±0,04	1,95± 0,02
3	Lime	5,5 ±0,12	104 ±1,8	23,0 ±0,6	33,5 ±0,4	14,30 ±0,25	3,95 ±0,05	2,08± 0,03
4	Sludge	5,9 ±0,15	106 ±1,9	17,2 ±0,3	32,3 ±0,5	14,35 ±0,30	3,70 ±0,05	2,00± 0,03

According to the data given in Table 1, the use of SS-based meliorant has, on the whole, a positive effect on the crop and technological quality of the agricultural crop (sugar beet) compared with the action of lime, SS as a self-fertilizing fertilizer. The best results are obtained with the application of fertilizer-meliorant «sludge-activated mule» – the yields increase by 20–30%. Thus, the sludge activates and optimizes the effect of organic fertilizers (excessive activated sludge), which has previously been confirmed in laboratory experiments and field studies in small areas. The main technological indicator of culture – sugar content improves by applying all proposed fertilizers by an average of 3–5%.

When silt gets into the soil, the sediments are mineralized, biogenic and other elements are transferred to the plant's nutritionally available compound. Excessive activated sludge of aerotanks is recommended as organic fertilizer enriched with nitrogene and digestible phosphorus compounds. The content of these substances in sediments is determined by the composition of the sewage and the technology of their purification. The ratio of total organic carbon to Nitrogen is 15:1 on average. Accumulation of potassium in the soil does not occur, because sediment is not enough for this element. The mineral part of the sediments contains calcium, silicon, aluminum, and ferrum compounds. Additions to municipal wastewater treatment plants cause the presence of a number of trace elements, such as Bor, Cobalt, Mangan, Copper, Molybdenum, Zinc in sieges. These trace elements when introduced into the soil in the composition of fertilizer increase the rate of flow of many biochemical reactions in plants, contribute to the absorption of organic crops from sediment from agricultural crops, their lack of soil causes a violation of metabolism in plants. The use of SS-based meliorant improves the physical and chemical properties of the soil, changes acid-alkaline properties favorably, soil-forming processes, life of soil biota and plants are activated. The reaction level (pH) improves the ecological state of the soil due to changes in its microbial coenosis, a sharp decrease in mobility and, consequently, the toxicity of Aluminum, Manganese and Ferum. During deoxidation, the amount of phosphates bound to the Ferum decreases and the Calcium phosphate group increases. This leads to better use of Phosphorus of Soil and Phosphorus of fertilizers by plants (fertilizer use factor of 20% versus 5% on acid soils, respectively). The ecological role of acid desulphurisation is to reduce the mobility of toxic compounds of Ferum, Manganese, Aluminum, heavy metals, in particular of radionuclides, which is especially important for contaminated soils. Degradation does not increase the unproductive losses of Phosphorus and Potassium from the soil. In fertilized soils mineralization of organic matter of the soil increases and the intensity of Nitrogen's cycle increases. All these changes are due not only to physical and chemical processes in the soil, but also to the biochemical activity of microorganisms of activated sludge and soil. During deoxidation and fertilization of the soil, qualitative and quantitative changes occur in the soil micrococoenosis: the number of microorganisms using organic and mineral nitrogen of the soil, as well as ammonifiers, is increasing; instead of fungi, bacteria and actinomycetes begin to predominate, which, in general, indirectly improves the effectiveness of fertilizer application. Taking into account the optimum conditions for the soil reaction index for sugar beet – pH_{KCl} 6,8–7,3, higher values of crop yield can be partially explained by the proof in the experiment of values of pH_{KCl} soil to an optimal level.

For the study of phytotoxic activity of soils after application to them of fertilizers-meliorants on the basis of SS was used biotesting method – «Growth test» methodology Gorovoy A.I. (Fig. 1).

Fig. 1. «Growth test» with the use of radish Raphanus sativus var. sativus L

According to the data obtained, the values for all plant growth parameters are set $t < t_{st}$ (N – 1; 0,05) = 1,96. This indicates that the results of the experiment are statistically insignificantly different from the control experiment. The absence of a statistically significant difference between the mean values of bioparameters in the control and investigated variants indicates the lack of phytotoxic activity of soils after their deoxidation. In general, the intensity of growth processes in plants grown on investigated soils after their deoxidation with the use of SS-based or lime-based melioration improves compared to control. A comparative analysis of the application for the restoration of the fertility of acid soils of traditional and alternative materials (respectively, lime and SS) showed the same positive effect when germination of test culture on experimental soils (Fig. 2, Fig. 3, Fig. 4).

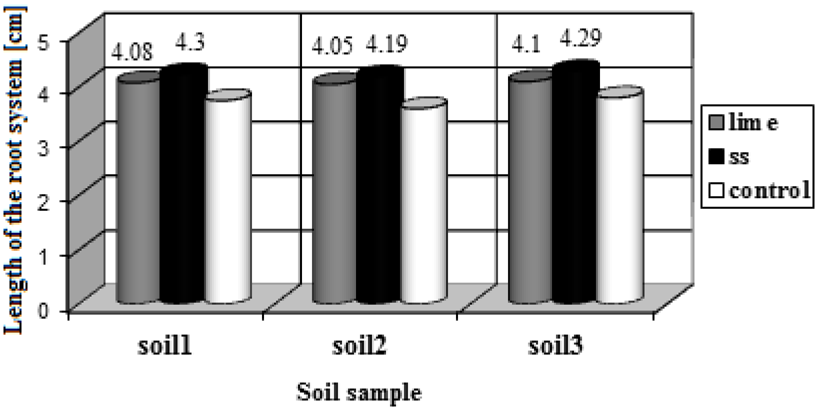

Fig. 2. Indices of growth along the length of the root system of plants

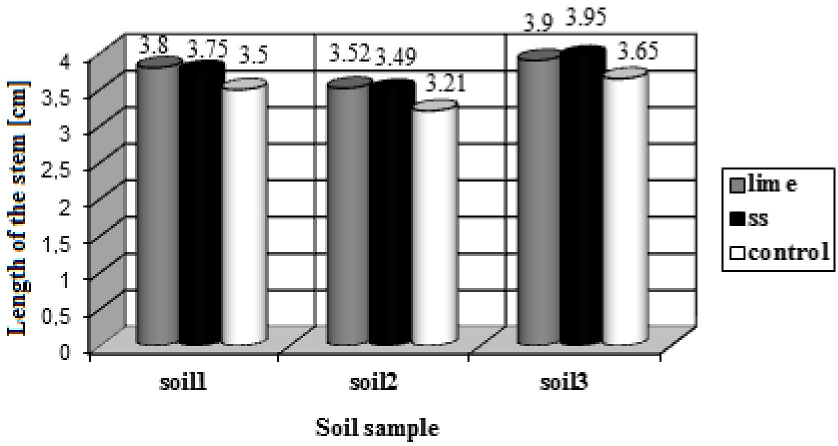

Fig. 3. Indicators of growth along the length of the stem of plants

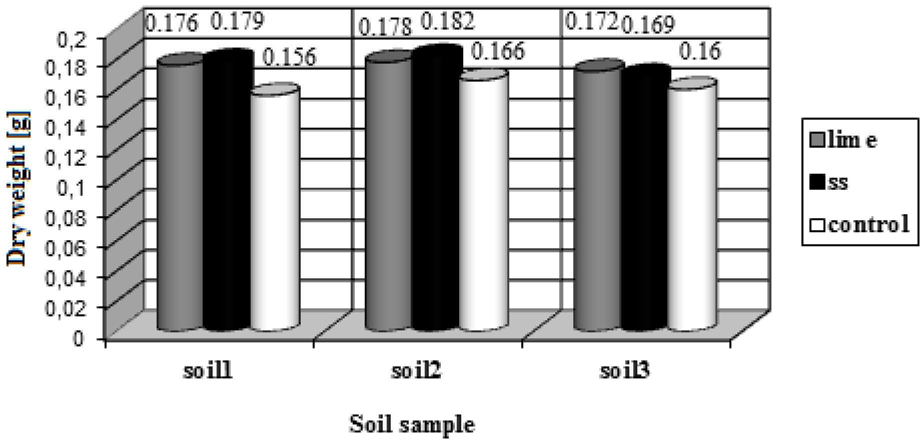

Fig. 4. Indicators of dry weight of plants

4. Summary and Conclusions

A fertilizer-meliorant on the basis of sewage sludge (excessive activated sludge of urban wastewater treatment plants and calcium sludge water treatment of a heat and power plant) can be recommended as an alternative when applied on medium and slightly acidic soils, since it effectively improves soil properties and stimulates on the level of traditional chemical remodeling (lime) biological processes in the soil, has a positive effect on the crop and technological parameters of agricultural crop – sugar beet. The main technological indicator of culture – sugar content improves when applying the proposed fertilizer by an average of 3−5%.

The presence of calcium-containing sludge in the granular fertilizer enhances the positive effect of active sludge as an organic component of fertilizer on the fertility of medium and slightly alkaline soils is revealed. Positive changes are due not only to physical and chemical processes in the soil, but also to the biochemical activity of microorganisms of activated sludge and soil.

The using of fertilizer-meliorant on the basis of sediment of sewage on medium-acid and slightly acidic soils, taking into account the dosage norms, does not increase the phytotoxic effect of soils after their liming on the values of test-culture bioparameters in the study by the method of biotesting («Rostov test») is established.

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System approach to oil production wastewater treatment

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Abstract

Extraction and processing of hydrocarbon-containing raw materials generates a number of environmental problems associated with environmental pollution. Drilling wastewater (DWW) is formed by the drilling of oil wells and affects the environment due to the content of suspended solids, oil products, heavy metals and salt ions in its composition. Mentioned chemicals with high mobility capacity migrate to the soil, underground water, soil water, threat to the natural geosystems, provoke high level of ecological danger for people, and destroy living beings.

Different technologies primarily based on physical, chemical and biochemical methods are applied for the cleaning of DWW. Each of these technologies is aimed at removing a certain range of pollutants and is not capable of completely solving the problem of DWW treatment. The idea of authors is development and implementation of complex DWW management system. The development of similar scheme allows obtaining commodity products such as petroleum products, etc. in pure form, and re-using the purified water, returning it to the circuit.

The purpose of the article is to develop complex DWW management system, determine main stages of technological process according to substance extraction sequence using appropriate methods.

A processing sequence of DWW treatment has been determined as reagent treatment, solid particles purification, organic substances purification, extraction of soluble forms, biochemical post-treatment. The output is pure water, oil products, and biogas and bio sulfur.

The study was conducted by the experimental determination of the sedimentation efficiency in reagent treatment at a different coagulant $Al_2(SO_4)_3$ and flocculant polyacrylamide (PAA) dose. All factors have approximately equal influence. There is a correlation between the $Al_2(SO_4)_3$ and PAA concentration and sedimentation rate of suspended particles

Keywords

produced water, drilling wastewater, spent drilling fluids, modelling, coagulation, complex treatment, bio-filtration.

1. Introduction

Drilling wastes, especially their liquid fraction including toxic organic and inorganic components such as oil and oil products, produced water, spent drilling fluids, drilling sludge and a number of chemical reagents create technogenic pressure on the environment. The sources of drilling wastes largely depend on the cuttings based on the geological condition of the borehole, the depth of the well, and the fluid used in the drilling operation. The technology of drilling production wells for oil production involves usage of drilling fluids, the chemical composition of which is predominantly represented by: polymers, polyacrylamides (PAAs), complex action surfactants, lubricating additives, and acid-soluble weighting agents (Pichtel, 2016).

The accumulation of drill cuttings is estimated in the range of 130 m^3 to 560 m^3 per well. The typical type of drilling wastes and their potential constituents are (Piszcz et al., 2014): water-based muds (WBM) cuttings – heavy metals, inorganic salts, biocides, hydrocarbons; oil-based muds (OBM) cuttings – heavy metals, inorganic salts, hydrocarbons, solid/cuttings; spent OBM – heavy metals, inorganic salts, hydrocarbons, solid/cuttings, surfactants; spent WBM – metals including heavy metals, inorganic salts, hydrocarbons, biocides, hydrocarbons, solid/cuttings; waste lubricants – heavy metals, organic compounds (Onwukwe and Nwakaudu, 2012).

According to Bakke et al. (2013); Isehuwa and Onovaes (2011) drilling wastewater is associated with such environmentally significant chemicals as arsenic, nickel, copper, chromium, zinc, aliphatic hydrocarbons, polyaromatic hydrocarbons (PAH), anthracene, fluoranthene, naphthalene, phenanthrene, pyrene and radioactive materials. Kharaka and Otton (2003) claimed the toxicity of produced water is directly related to high salinity (3000 to > 350,000 mg/L total dissolved solids (TDS).

A recent analysis (Yost et al., 2016) showed that only 8% of the 1,076 chemicals listed as being used in fracking fluids and 62% of the 134 chemicals documented in flowback and produced water had sufficient toxicological data to calculate chronic oral toxicity values.

The results obtained by Siddique et al. (2017) suggest designing a sustainable and viable drilling wastewater management (DWW) model, the first step is to identify the composition and nature of the pollutants in the wastes. Based on this information different waste treatment plan can be placed in operation such as, thermal treatment, thermal and mechanical treatment, biological treatment, encapsulation of pollutants.

Ebrahimi et al. (2009) mentioned disposal of produced water requires imperative environmental regulations and produced water re-injection (PWRI) requires skillful planning and treatment to meet the quality needed for reinjection water to avoid ecological damage.

Nasiri and Jafari (2017) analyzed the possibility of application different treatment methods to achieve certain aim expressed in such outcomes as de-oiling, suspended particles removal, iron removal, softening (Ca and Mg removal), soluble/trace organics removal, desalting. Date obtained as a result of mentioned research and assessment of significant technologies based on physical, chemical and biological methods indicate the following. Nanofiltration (NF) and reverse osmosis (RO) are the most effective treatment methods among other investigated: API separator, deep bed filter, hydrocyclone, aeration and sedimentation, precipitation, ion exchange, biological treatment, thermal desalination, activated carbon, chemical treatment, ultrafltration (UF), microfiltration (MF), electrodialysis.

Total removal percentage of oil content up to 93% with MF as pre-treatment step and up to 99.5% with UF followed by NF as final treatment was shown in study (Ebrahimi et al., 2009). Marie-Pierre et al. (2007) approves the Therma-Flite Holo-Scru® thermal desorption system is an alternative thermal desorption technology which volatilize and recover 99.5% of hydrocarbons from drilling mud.

2. *Materials and Methods*

Implemented analysis of existing approaches to DWW treatment clearly showed none of used technologies can solve the posed problem. It is necessary to develop an integrated technological system that takes into consideration both the wastewater chemical and physical parameters and external factors to achieve full oil production wastewater treatment and minimize the negative impact on the environment.

The most important controlled parameters in evaluating the efficiency of drilling wastewater cleaning are: dry residue, pH, suspended solids, oil products, chlorides and sulfates (Helmy and Kardena, 2015). Drilling wastewater formed during drilling wells in the Bugruvate (DWW-1) and Anastasivka (DWW-2) field of Dnieper-Donets oil and gas regions of Ukraine (Sumy region) was used for the study. Physical and chemical composition of the drilling wastewater samples are shown in Tab. 1.

Table 1. Physical and chemical composition of the studied DWW samples

Parameter / **Sample**	**Dry residue [mg/dm^3]**	**pH [units]**	**Suspended solids [mg/dm^3]**	**Oil products [mg/dm^3]**	**Chlorides [mg/dm^3]**	**Sulfates [mg/dm^3]**
DWW-1	6470	8.7	650	150	750	432
DWW-2	7254	9.1	700	184	827	507

3. *Results and Discussion*

Chemical substances of DWW can be divided into four structural and functional groups such as mechanical impurities, petroleum products, heavy metal cations and anions of acid residues. In this connection it is advisable to carry out their purification in a complex and phased manner using specific technologies and methods (Fig. 1).

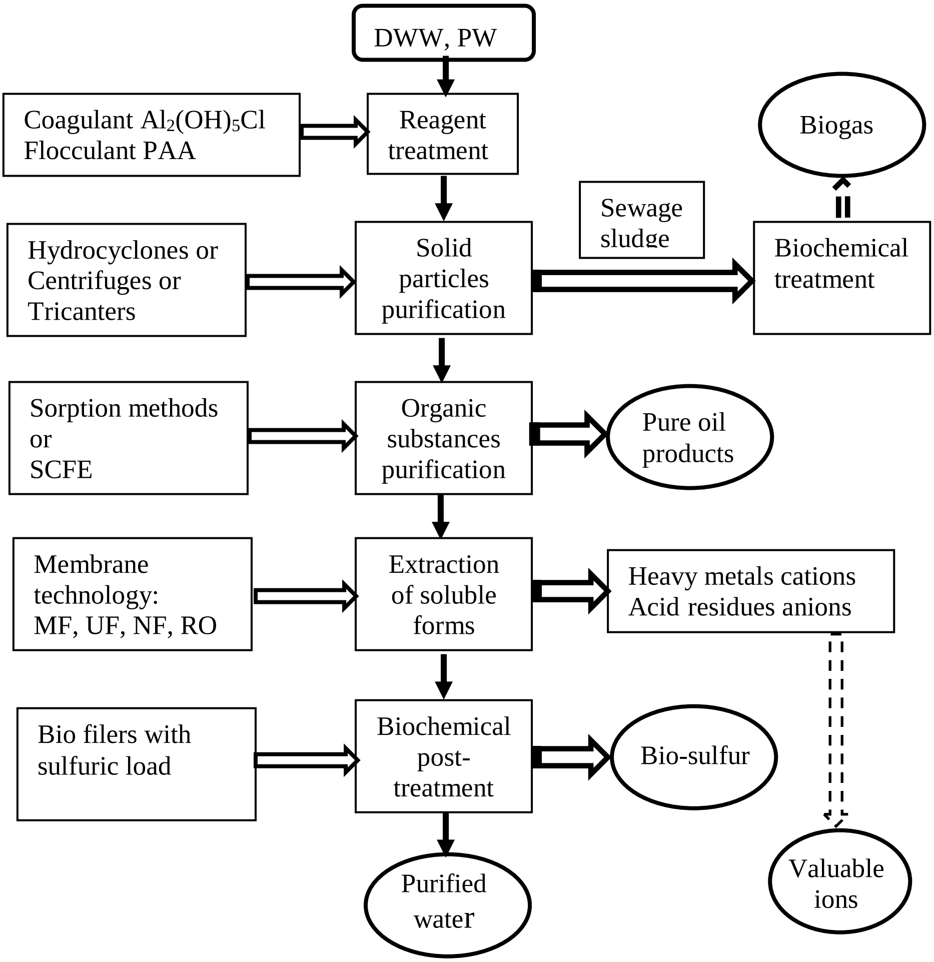

Fig. 1. Principal structure of drilling wastewater treatment:
DDW – drilling wastewater, PW – produced water, PAA – polyacrylamide,
SCFE – supercritical fluid extraction, MF – microfiltration, UF – ultrafiltration,
NF – nanofiltration, RO – reverse osmosis

First, it is necessary to separate the solid phase, then the hydrocarbons and other organic substances, and finally the dissolved substances are extracted from the drilling wastewater. Mechanical, chemical, physical and chemical methods are combined at the initial and basic stages to effectively implement these processes. The biochemical method is used at the last stage for the final purification of drilling wastewater and for the quality of treated waters to be reduced to normative indices. Moreover, the proposed scheme allows not only to reduce the burden on the environment by preventing the discharge of polluted drilling wastewater, but also to use the extracted and received substances as a marketable product.

A four-stage purification system is effectively used for phase separation of drilling waste. It includes vibrating screens, sand separators or sieve hydrocyclones, silos and centrifuges, decanters and tricanters. The process is intensified by electrocoagulation, reagent coagulation and flocculation.

It is advisable to use coagulant aluminum sulfate $Al_2(SO_4)_3$ for accelerating the deposition of suspended particles, and flocculant polyacrylamide (PAA), which increases the size of the flocs during coagulation, to improve the chemical precipitation process.

Experimental research allowed to determine optimal doses of coagulants and flocculants (fig. 2).

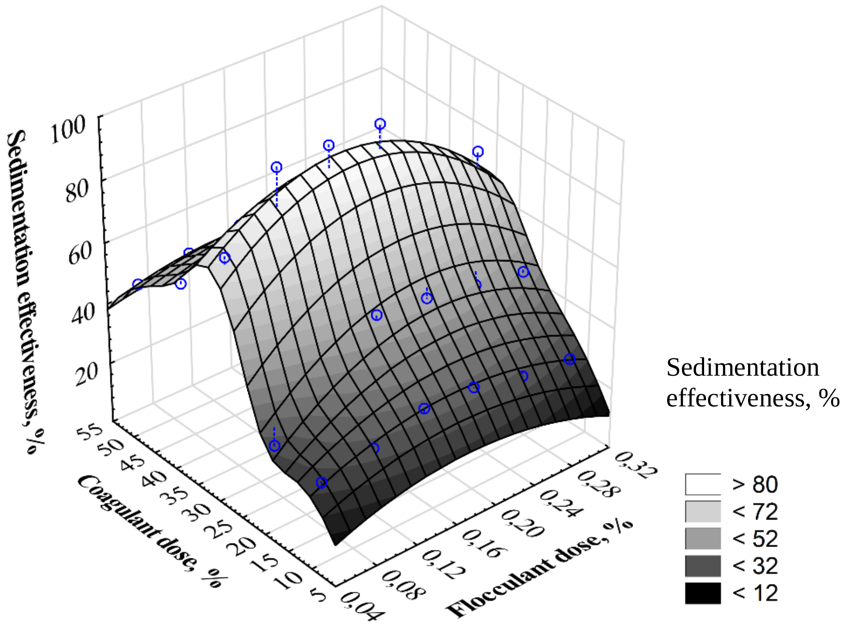

Fig. 2. Dependence of the sedimentation effectiveness from the coagulant dose and flocculant dose

The influence of these factors on the sedimentation effectiveness can be approximated by the regression equation:

$$Y = -49.4767 + 314.9857 \cdot X_1 + 6.0946 \cdot X_2 - 741.428 \cdot X_1^2 - \\ -1.8914 \cdot X_1 \cdot X_2 - 0.087 \cdot X_2^2 \quad (1)$$

where: Y – sedimentation effectiveness [%], X_1 – flocculant dose [%] and X_2 – coagulant dose [%].

For this model, the determination coefficient was 0.9486, the Fisher criterion (f_1 = 18, f_2 = 20) F = 2.0936 ($F_{tab.}$ = 2.18 – model is adequate), the standard estimation error is 0.3031, which satisfies relevance requirements.

An increase in the deposition rate of enlarged particles can reduce the sedimentation time and the slurry separation time, and also reduce the size of the sedimentation tanks. The sedimentation rate of suspended particles under laminar flow conditions is determined by the Stokes equation:

$$\omega_{dep} = \frac{g \cdot (\rho_s - \rho_m)}{18 \cdot \mu} \cdot d^2 \quad (2)$$

where: ω_{dep} – particle deposition rate [m/s], g – acceleration of gravity [m/s^2], ρ_s – density of the particle [kg/m^3], ρ_m – density of the medium [kg/m^3], d – particle diameter [m] and μ – dynamic coefficient viscosity of the medium [kg/m·s^2].

It was found that the use of 10% and 20% coagulant solution is not enough to completely precipitate impurities, a viscous mass, a pasty precipitate was formed. In the case of using a 30% solution, a clear separation of the liquid and solid phases was observed, and the formed precipitate was flakes with a loose structure. A further increase in the concentration of the coagulant to 40% and 50% resulted in the absorption and binding of the water sediment, which made it difficult to separate it. Therefore, the most effective is a 30% solution of the coagulant.

In addition to the coagulant, a PAA flocculant was used, which requires an acidic medium for optimal operation. Therefore, hydrochloric acid was added to the solution. The optimum dose of flocculant and the concentration of acid were determined due to provided experiments. It was found that 0.1–0.2% of the flocculant by the main substance and 9–10% of the acid solution provide maximum effectiveness of the sedimentation process intensification resulted in the increasing of particle deposition rate in view of their aggregation on the agent's surface. It should be noted that the sequence of the addition of reagents is extremely important under investigated conditions. The experiments showed that firstly, hydrochloric acid must be added to the drilling wastewater to create the appropriate medium, followed by PAA and coagulant. In this case, the precipitate contains a minimum amount of water and the separation of the solid phase from the liquid centrifuge become easier.

A high degree of efficiency of cleaning oil-contaminated soils from organic compounds has the technology of supercritical fluid extraction (SCFE). The process of SCFE is based on the use of solvents under critical temperature and pressure conditions, in which the substance has properties intermediate to the liquid and gas, which ensures its anomalously high solubility. As a supercritical fluid (SCF), we propose to use carbon dioxide due to a number of advantages, in particular regenerability, availability, low critical temperature and pressure, before such substances as ammonia, sulfur dioxide, water. Comparative characteristic of physical parameters of some SCF is shown in table 2.

Table 2. Characteristics of the physical parameters of solvents in the supercritical state

Supercritical fluid	**Critical parameters**		
	Temperature [K]	**Pressure [MPa]**	**Density [kg/m^3]**
Methane	191	4.6	162
Carbon dioxide	304	7.38	468
Ammonia	406	11.3	235
Sulfur dioxide	431	7.80	524
Methanol	513	3.04	278
Benzene	562	4.84	293
Toluene	592	4.11	312
Water	647	22.0	322

It is clearly evidence that exclusively biochemical method group have a potential to transform drilling fluids wastewater, particularly dehydrated drilling sludge/mud, into valuable organo-mineral substrate. Biochemical method of processing of drilling wastewater is realized at aeration stations or in bioreactors, where biodegradation of organic compounds of DWW and liquid hydrocarbon-containing wastes is carried out by a specific microflora. Biotechnological scheme of this process should include the use of ground biologics, sorbents, disintegrants, buffer stabilizers, which are introduced in certain quantities and sequences. The most useful form of soil preparations among such as: culture fluid, dry powder, biomass of microflora pure or immobilized (planted) specific carrier is the first one.

Biofilters for the post-treatment of biologically treated sewage are widely used today. Biofilters are structurally the filters with granular loading, on which the oil-oxidizing bacteria are immobilized. It is efficient and expedient to carry out the loading from production waste such as granulated phosphogypsum or granular gas sulfur, which allows to reduce anthropogenic load on the surrounding environment due to industrial wastes utilization (Zinberg and Nenasheva, 2013).

4. Summary and Conclusions

Significant amount of OBM and WBM chemicals are transferred to liquid drilling waste, therefore, the primary preventive measures in protecting the hydrosphere from the negative effects of DWW are the revision of the drilling fluids formulation and the replacement of constituent components with less toxic ones.

Beneficial use of drilling wastewater treatment system must take into consideration its chemical and physical properties, short-term and long-term effects on living beings, possibility of including biogenic, even toxic elements into metabolite biochemical processes. It has been shown that complex system of DWW purification is the most acceptable and effective approach that allows to prevent soil and water pollution by oil products, salts and heavy metals.

Moreover, wastewater treatment according to the proposed schematic diagram ensures the return of purified water to the technological process of drilling wells, which corresponds to the principles of rational water use.

There is a significant need for further field studies, in particular the study of post-treatment of biologically treated DWW. Future DWW management technologies are likely to focus on the development and introduction of biofilters with granular loading, obtained from industrial waste.

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The impact of the contact time on the effectiveness of organic compounds removal from water – pilot scale investigation

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Abstract

Organic compounds which are commonly found in water can react with other pollutants and contribute to the formation of oxidation by-products. In order to obtain the required quality of water intended for human consumption and to have biologically stable water running in the water distribution system, it is necessary to effectively remove these substances from water. In addition to traditional methods of removing organic compounds from water, such as coagulation, adsorption or chemical oxidation, filtration with biological activated carbon filters is more and more often used in water treatment plants. The time of water contact with the filter bed has a significant impact on the biofiltration process. The technological investigation of organic compounds removal from water was carried out on a pilot scale, at the experimental station consisting of two biologically active filters (BAF) differed from each other by the method of the filter bed activation. The filtration rate was 2.5 m/h and 5 m/h and the contact time was 50 and 25 minutes, respectively. The research carried out in 2016 was a continuation of the studies already published (Holc et al., 2016 a; Holc et al., 2016 b; Holc et al., 2017). Their purpose was to determine the impact of water contact time on the effectiveness of organic compounds removal from water in the beds of biologically active carbon filter.

Keywords

biologically active filters (BAF), organic compounds, filtration, pilot studies, contact time

1. Introduction

Organic compounds present in water can react with other pollutants and contribute to the formation of disinfection by-products, deteriorating the quality of water supplied to consumers (Włodyka-Bergier and Bergier, 2011; Gibert et al., 2013; Wolska, 2014; Gibert et al., 2015; Szuster-Janiaczyk, 2016; Włodyka-Bergier et al., 2016). In order to achieve the required drinking water quality and to supply biologically stable water to the distribution network, it is necessary to remove organic compounds from water (Council Directive 98/83/EC, Adamski and Szlachta, 2011; Journal of Laws of 2017, item; Pruss, 2015). In addition to traditional methods of organic compounds removal from water, such as coagulation, adsorption or chemical oxidation, biologically active carbon filters are often used in Water Treatment Plants (Seredyńska-Sobecka et al., 2006; Simpson 2008; Pruss et al., 2009; Zhu et al., 2010; Świderska-Bróż and Wolska, 2011; Olesiak and Stępniak 2014; Gibert et al., 2015; Holc et al., 2016 a; Papciak et al., 2016). The exploitation of biologically active carbon filters is based on two complementary processes: biodegradation and adsorption. In the initial period of BAF operation, sorption is the predominant process. When a sufficient amount of pollutants is adsorbed and biofilm develops, biodegradation becomes the predominant process. The degradation of pollutants as a result of the activity of microorganisms is a very efficient process. Microorganisms are especially effective in the biodegradation of substances which are difficult to remove using conventional methods (Kulągowska and Skonecka, 2006; Feng et al., 2013; Holc et al., 2016 a; Holc et al., 2016 b; Elhadidy et al., 2017). Due to the high effectiveness of dissolved organic matter removal by biologically active carbon filters, the treated water requires much less disinfectants (Kołaski et al., 2017). In such water, the likelihood of harmful by-products formation or bacterial regrowth in the distribution network is much lower. Filtration with biologically active filters also eliminates the need for coagulation, a process frequently used in support of conventional filtration (Simpson, 2008).

UV absorption is a popular and relatively simple indicator used to determine the content of organic pollutants in water. The functional groups of organic compounds which absorb UV and VIS radiation are chromophores. Most chromophore groups are found in humic acid particles. Water containing organic compounds with chromophores demonstrates absorbance within the range of 200 to 350 nm. To track UV absorption changes, the 254 nm wavelength was considered most suitable. Based on absorbance, it is possible to determine the total dissolved organic carbon fraction and organic compounds with a high content of aromatic rings which are precursors of disinfection or oxidation by-products. Its value is often interpreted as an indicator of the degree of aromatic rings activation which serves as the basis for predicting the reactivity of aromatic components during chlorination (Nowacka et al. 2012; Szerzyna et al., 2017).

Effective biofiltration can take place in the beds of both first and second stage rapid filters. There are a number of factors and parameters which have a significant impact on the biofiltration process, in particular including the time of water contact with the filter bed, the filter backwashing parameters, temperature, ozonation or no ozonation before biological filtration and the presence of oxidizing agents in the water supplied to the filters (Urfer et al., 1997).

2. Materials and Methods

The test station consisted of two filtration columns with an inner diameter of 100 mm and the height of 300 cm, filled with WG-12 granular active carbon bed (Fig. 1). Constant temperature was maintained across the entire filter bed height thanks to the use of a water jacket – a pipe with an inner diameter of 140 mm filled with water therefore the water flowing through the pipe had a temperature equal to the temperature of the water subject to filtration. In order to prevent the growth of algae, the filters were covered with black geotextile. The water supplied to the filters was tap water. Across the entire filter bed, stub pipes for sampling the water and the bed were located. The system was launched in April 2015 and is still operated. The filtration columns differed from each other with the filter bed activation method and the identified colonizing microorganisms (Holc et al., 2016 a; Holc et al., 2016 b). In the first filtration column, the microorganisms colonizing the filter bed come from the water which passes through the filter, supplied from the distribution network. In the second filtration column, the filter bed was inoculated with wastewater from the backwashing of carbon filters used in a selected water treatment plant. Water samples for analysis were collected at the inlet of the filtration columns and from the filter bed, at the depth of 45 cm, 85 cm, 165 cm and 205 cm. Water samples for analysis were collected on the same days from both filters. Biologically active filters were backwashed with a one-day shift according to a predetermined schedule. A detailed method of BAC filters backwashing was presented in another publication (Komorowska-Kaufman et al., 2017).

The effectiveness of organic compounds removal from water was evaluated on the basis of the following parameters: COD_{KMnO4}, dissolved oxygen concentration, total organic carbon (TOC), pH, alkalinity and UV_{254} absorbance. The filtration rate was 2.5 m/h and 5.0 m/h, and the contact time was 50 and 25 minutes, respectively. The research was carried out in 2016 as a continuation of studies published earlier (Holc et al., 2016a; Holc et al., 2016b; Holc et al., 2017).

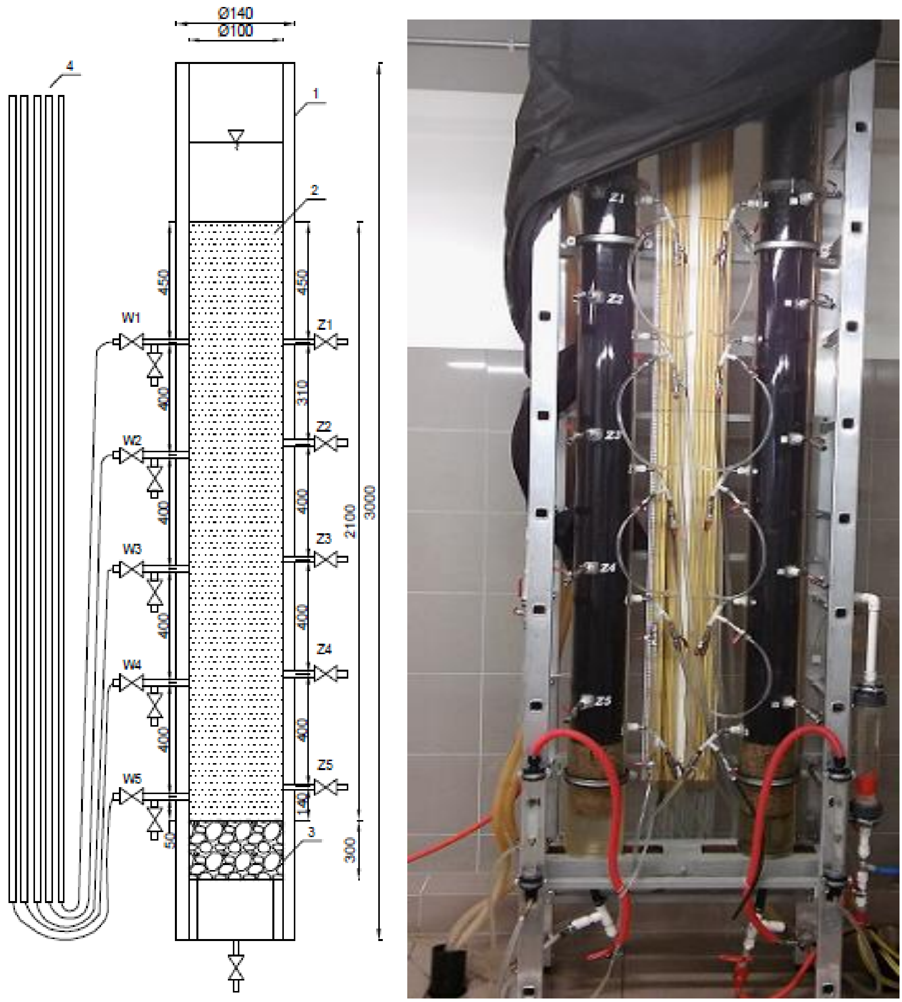

Fig. 1. Pilot plant with two identical filtration columns:
1- filtration column, 2- WG-12 activated carbon filter bed, 3- gravel supporting layer,
4- piezometers, W1-W5 water sampling points, Z1-Z5 filter bed sampling points

3. Results and Discussion

The water supplied to the filtration columns was characterized by a varying content of organic substances. The effectiveness of their removal from water was evaluated through changes in COD_{KMnO4}, the content of total organic carbon (TOC) and UV_{254} absorbance. In the analyzed study period, the COD_{KMnO4} of the water supplied to filter F1 ranged approximately from 3.0 to 4.5 mg O_2/dm^3 (Fig. 2). The values recorded in the case of filter F2 ranged approximately from 3.0 to 4.8 mg O_2/dm^3 (Fig. 3). The COD_{KMnO4} value decreased along with the growing thickness of the filter bed. Similar COD_{KMnO4} levels were found at the outlet from both filters: the average COD_{KMnO4} in the effluent from filter F1 was 3.05 mg O_2/dm^3 and from filter F2 2.96 mg O_2/dm^3. During the study period, incidental surges in COD_{KMnO4} values occurred in samples collected from deeper filter bed layers which can possibly be attributed to the presence of biofilm fragments in the water.

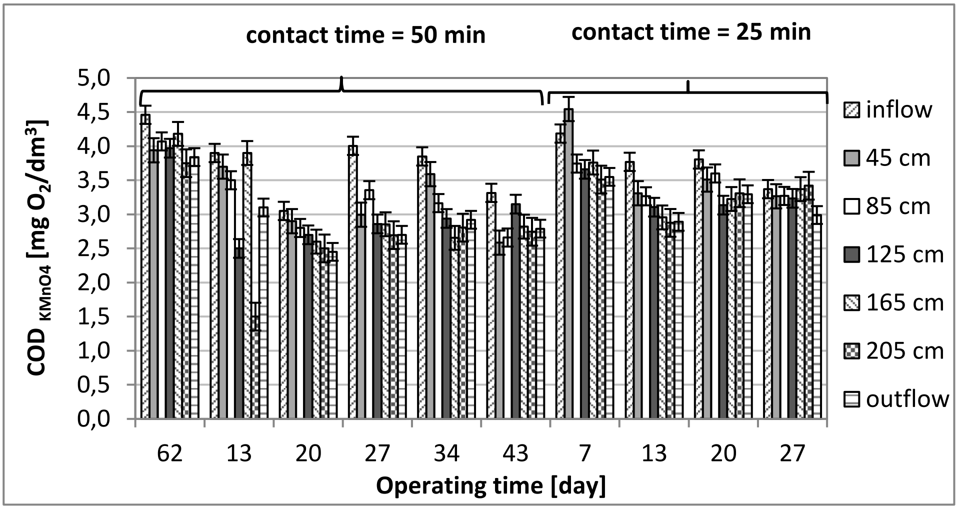

Fig. 2. Impact of the contact time on changes in the COD_{KMnO4} value in filter F1

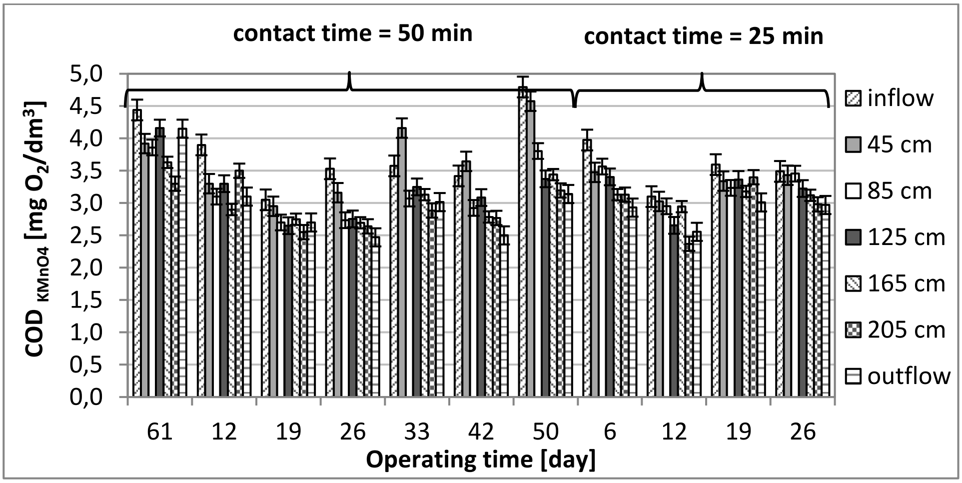

Fig. 3. Impact of the contact time on changes in the COD_{KMnO4} value in filter F2

The highest reduction (by more than 30%) in water COD_{KMnO4} was achieved at the filtration rate of 2.5 m/h, in the case of filter F1 on day 27 of the filtration run, and in the case of filter F2 on days 26 and 50. It was observed that the efficiency of the COD_{KMnO4} reduction in water depended on the filter operating times.

Additionally, a decrease in the efficiency of COD_{KMnO4} reduction was noted when the filtration rate was increased i.e. the contact time became shorter.

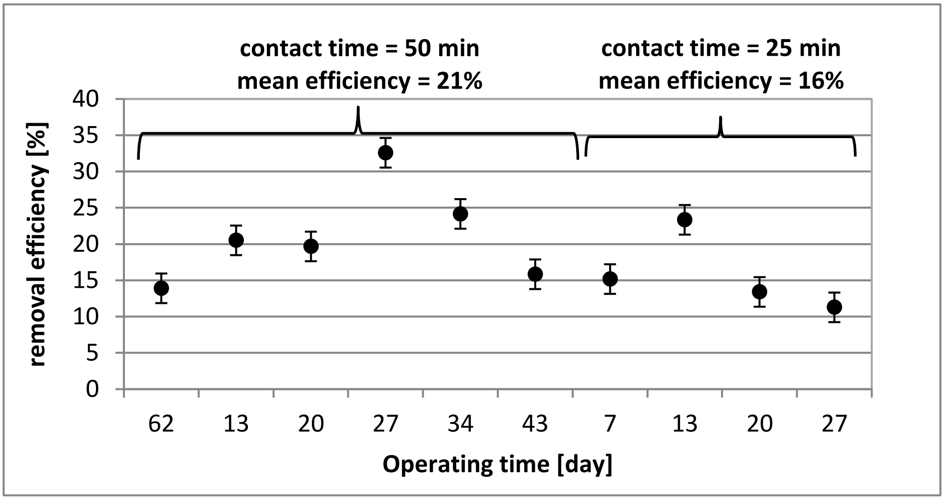

Fig. 4. CODKMnO4 removal efficiency in filter F1

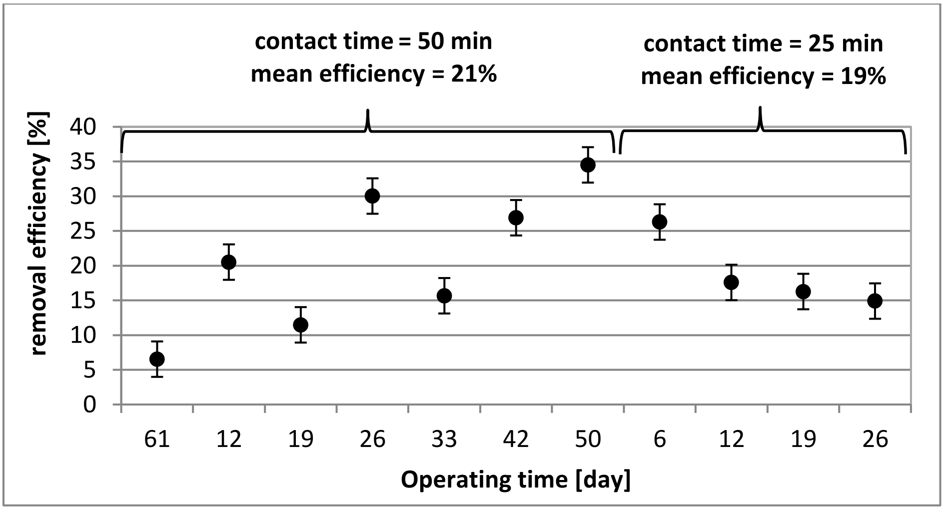

Fig. 5. COD_{KMnO4} removal efficiency in filter F2

Changes in the TOC value during the study period are presented in Figures 6 and 7. The average total organic carbon concentration in water supplied to the model was 3.9 mg C/dm^3. The TOC concentration went decrease as water was passing through the filter bed. The contact time of 25–50 minutes ensured TOC reduction by 10 to 13% in both filters.

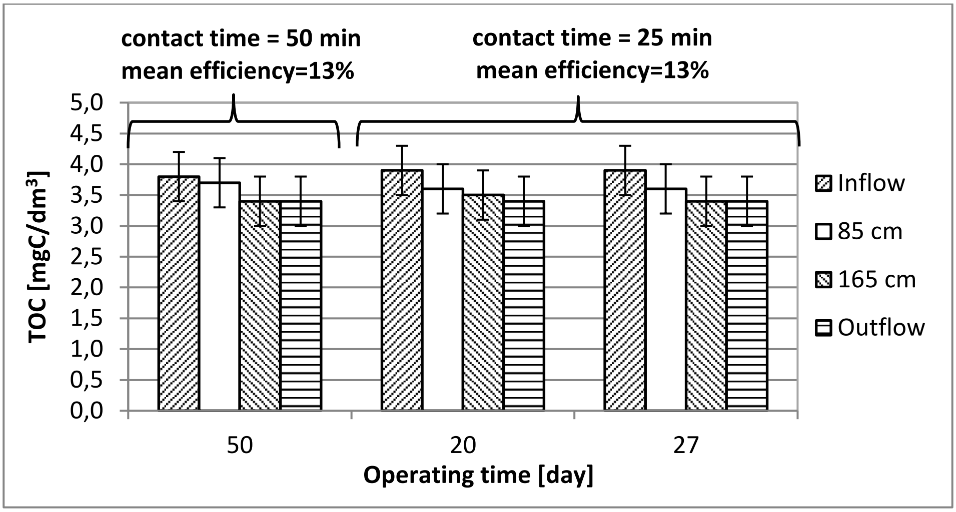

Fig. 6. The impact of contact time on the changes in TOC concentration in filter F1

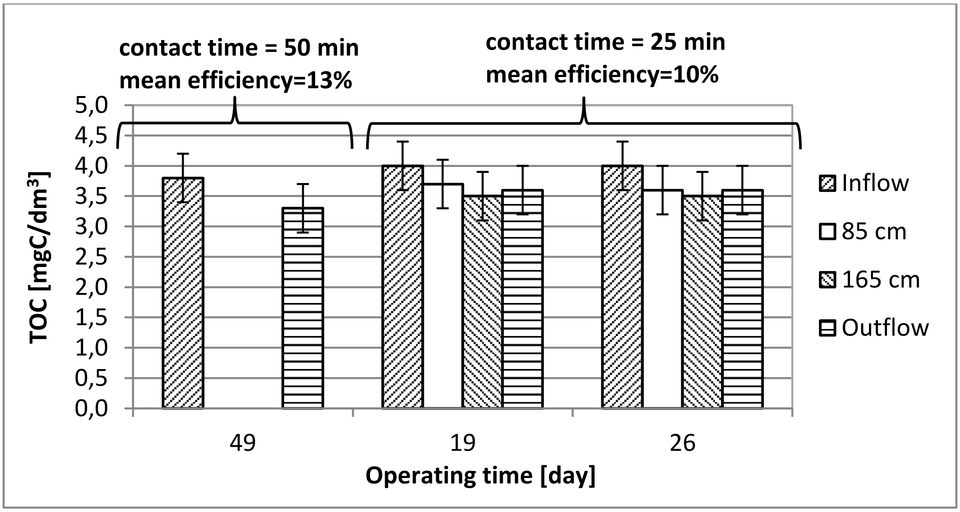

Fig. 7. The impact of contact time on the changes in TOC concentration in filter F2

As a result of the filtration of water through a biologically active filter bed, the absorbance value decreased significantly which evidences high effectiveness of removing the biodegradable fraction of organic pollutants, since the UV_{254} absorbance is an indicator of the concentration of dissolved organic carbon and organic compounds with a high content of aromatic rings – disinfection by-product precursors and oxidation by-products (Nowacka et al., 2012). Different values of UV_{254} absorbance in the filtration beds indicate differences in the efficiency of organic compounds removal which can most likely be attributed to the different methods of filter bed activation and different colonizing microorganisms (Holc et al., 2017).

In order to examine the nature of the organic compounds transformations taking place in the course of the filtration process on a biologically active filter bed, the value of parameter p was calculated:

$$p = \frac{Abs\ UV_{254}}{TOC} [-] \tag{1}$$

where: $Abs\ UV_{254}$ – absorbance UV_{254} [cm^{-1}], TOC – total organic carbon [mg C/l].

The value of the p parameter in the inflow and outflow water of the filters was calculated for selected days on which both indicators were determined and the results are presented in Table 1. The determination of UV_{254} absorbance covers the DOC fraction, mainly complex, cyclic compounds with multiple bonds, including compounds with aromatic rings. The TOC, on the other hand, covers all dissolved organic compounds, also the simple ones which are not determined on the basis of UV_{254} absorbance (Balcerzak and Łuczak, 2006).

Table 1. Relationship between UV_{254} absorbance and TOC.

Filter	**Time**	**T_{cont}**	**Inflow**			**Outflow**			**Δp**
			UV_{254} abs.	**TOC**	***p***	**UV_{254} abs.**	**TOC**	***p***	
	[d]	**[min**	**[cm^{-1}]**	**[mgC/l]**	**-**	**[cm^{-1}]**	**[mgC/l]**	**-**	**-**
F1	50	50	0.0062	3.8	0.0016	0.0000	3.4	0.0000	0.0016
	20	25	0.0064	3.9	0.0016	0.0034	3.4	0.0010	0.0006
	27	25	0.0062	3.9	0.0016	0.0014	4.4	0.0003	0.0013
F2	49	50	0.0132	3.8	0.0035	0.0042	3.3	0.0013	0.0022
	19	25	0.0042	4.0	0.0011	0.0024	3.6	0.0007	0.0004
	26	25	0.0044	4.0	0.0011	0.0000	3.6	0.0000	0.0011

A major decrease in the *p* value was observed in the case of both filters which indicates removal of organic compounds with a complex structure (Balcerzak and Łuczak, 2006). The difference between the inflow and outflow for this parameter amounted to Δp=0.0006–0.0016 for filter F1 and to Δp=0.0004–0.0022 for filter F2. A larger change in the *p* value in both filters was obtained for a longer contact time. Most likely, as the UV_{254} absorbance reflects mainly the biodegradable fraction of organic matter, the changes in the *p* parameter were to a higher extent caused by microorganisms colonizing the filter beds which differed between the two filters (Holc et al., 2016 b), than the time of contact with the filter bed.

4. Summary and Conclusions

The contact time is an important operating parameter of biologically active carbon filters. The conducted research has demonstrated its impact on the effectiveness of organic compounds removal from water determined on the basis of the COD_{KMnO4} value. The biological activity of the carbon filters certainly contributed to it. When the filter was operated at a twice higher filtration rate, the shorter contact time reduced the efficiency of removal of the biodegradable organic matter fraction.

In order to examine the nature of the organic compounds transformations taking place in the course of the filtration process on a biologically active filter bed, the value of parameter p was calculated which depends on the UV_{254} absorbance and the TOC. A significant decrease in the parameter value for both filters was observed which evidences removal of organic compounds from the water.

Acknowledgments

The research was financed under project 01/13/DSMK/0864 "Using of modern microorganism identification methods for the study of biologically active carbon filters".

The authors of the publication would like to thank Bartosz Przybylski and Paweł Białecki, graduates of the Poznan University of Technology, for their contribution to the performance of physicochemical analyses for the purpose of the study.

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Improving environmental safety of food industry enterprises

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Abstract

The development of food industry enterprises in Ukraine in modern conditions is accompanied by an increase of their environmental hazard. They have low levels of treating sewage wastewaters, emissions of pollutants into the air and a significant amount of waste products. That is why it is necessary to create modern environmental management systems for the protection of the environment.

With the traditional methods, it is possible to apply innovative processes of treatment of wastewaters from industrial enterprises using advanced technologies. In order to simplify the technological scheme while preserving qualitative indicators of treated water, the underlying task was to develop a method of treatment and disinfection of wastewaters using physical methods, namely cavitation. The vibration-cavitation treatment of liquid media belongs to these methods. It is based on the results of previous experimental studies, allows to accelerate the treatment of natural and sewage wastewaters by removing both biological and chemical contaminants.

Keywords

cavitation, water treatment, ecological production, treatment facilities of enterprises, vibrocavitator.

1. Introduction

The issue of environmental technologies is of great importance for the food industry. It provides for a system of measures to prevent the negative impact of production processes upon the natural environment.

The sewage waters clearing is an important environmental problem of the national economy of any country, the ignoring of which may lead to the significant negative consequences in the form of the nationwide environmental disaster. The problem of the sewage waters clearing is especially important for Ukraine, where because of the use of out-of-date technologies most sewage waters are characterized by a high level of chemical and biological contamination (Starchevskyy et al., 2017).

Environmental production is achieved through the efficient processing of raw materials and the introduction of non-waste and low-waste technologies. (Kroyer, 1995, Starchevskyy et al., 2017, Kalumuck, 2003). All of this ultimately contributes to the production of high-quality, environmentally safe food products at the minimum cost of natural resources and the preservation of a stable dynamic balance in the natural environment of the biosphere of the Earth (Starchevskyy et al., 2017, Capocelli, 2014).

The use of classical technology of biological clearing involves high-energy costs, a creation of surplus biomass, which causes filter clogging of waste treatment facilities and requires additional expenditures for its recovering (Shevchuk et al., 2012). Sewage waters of food industry enterprises contain a considerable amount of biological and chemical contamination so after they are emptied to the surface water bodies there continues a range of chemical and biological transformations, which has a negative effect on the environment (Shevchuk et al., 2012). In addition, the water obtained after the biological treatment can be used without further treatment for only a few purposes and under the absence of contact of this water with people (Jyoti, 2003, Madhu, 2010). In most cases, biologically treated waste waters are only a raw material for the preparation of technical water (Dyussenov, 2013).

Among the various methods of physical influences upon water treatment processes methods of cavitation treatment of water have been widely used. They are based upon ultrasonic (Capocelli, 2014) and hydrodynamic (Capocelli, 2014) cavitation perturbation in liquids. These methods combine relatively high levels of chemical treatment and biological decontamination by way of initiating and activating oxidative reactions in the cavitation field.

The use of the cavitation technologies may become the alternative to the existing methods of sewage water contamination level decrease. Ecological safety, high level of adaptability to the existing technologies and effectiveness allows using the cavitation technologies both as the main and the additional methods of intensification of the sewage waters clearing and significantly reduce a technogenic pollution of the hydrosphere (Starchevskyy et al., 2017).

However, these methods have certain disadvantages, including the discreteness of treatment, low productivity and high energy consumption for the ultrasonic method, insufficient level of purification and uniformity of treatment, etc. for the hydrodynamic method. Because of that, it is the actual search for modern technologies of water treatment and water treatment, which can provide a high level of water treatment with significant productivity.

The vibration electromagnetic cavitator of resonance action for cavitation perturbation belongs to the equipment that uses cavitation chemical and technological processes. It can be applied for disinfection of drinking water and waste waters of chemical, food and processing enterprises by removing various contaminants, including biological ones. The cavitator functions based on the physicochemical methods of cavitation initiation and the redox reactions activation in liquids by using the energy of the large number of cavitation bubbles splashing (Capocelli, 2014).

Mechanisms of the influence of cavitation field upon microorganisms, when applying the electromagnetic vibrocavitator are similar to those of devices with hydrodynamic action (Madhu, 2010). The energy released by cavitation bubbles leads to a sharp expansion of the cells of microorganisms and the breakdown of their membranes. As it is known, splashing cavitation bubbles activates oxidative reactions in a liquid, which further contributes to the purification of contaminated water, improving its physical and chemical parameters (Capocelli, 2014).

So, the possibility appears, to treat the water additionally, which is important for the processes of water conditioning. Practically, there is the possibility of technologically targeted and effective use of the cavitation action in production processes (Shevchuk et al., 2012, Grigoriev, 2007, Naddeo et al, 2014).

2. *Materials and Methods*

Studies *of the influence* of cavitation upon wastewaters from breweries, east production (private company “Enzym”), yeast solutions with different initial values of chemical oxygen demand (COD) and microbial number (MPN_0) were conducted.

For cavitation treatment of aqueous solutions and solutions with toxic and chemically aggressive substances, special designs of low-frequency vibration cavitators were developed by the Department of General Chemistry of the *Lviv Polytechnic National University*. They can ensure the treatment of liquids under the pressure of up to 3.5–4.0 kg/cm^2 and temperatures up to 220°C. They are made of stainless steel, and the distinctive difference from the traditional vibration cavitators is the lack of elastic corrugations and resonant elastic suspension of oscillating disturbers of cavitation.

A cooling system for electric drives is provided for to prevent the excessive overheating of the drive electromagnets in cavitators of this type during the cavitation treatment of liquids at high temperatures.

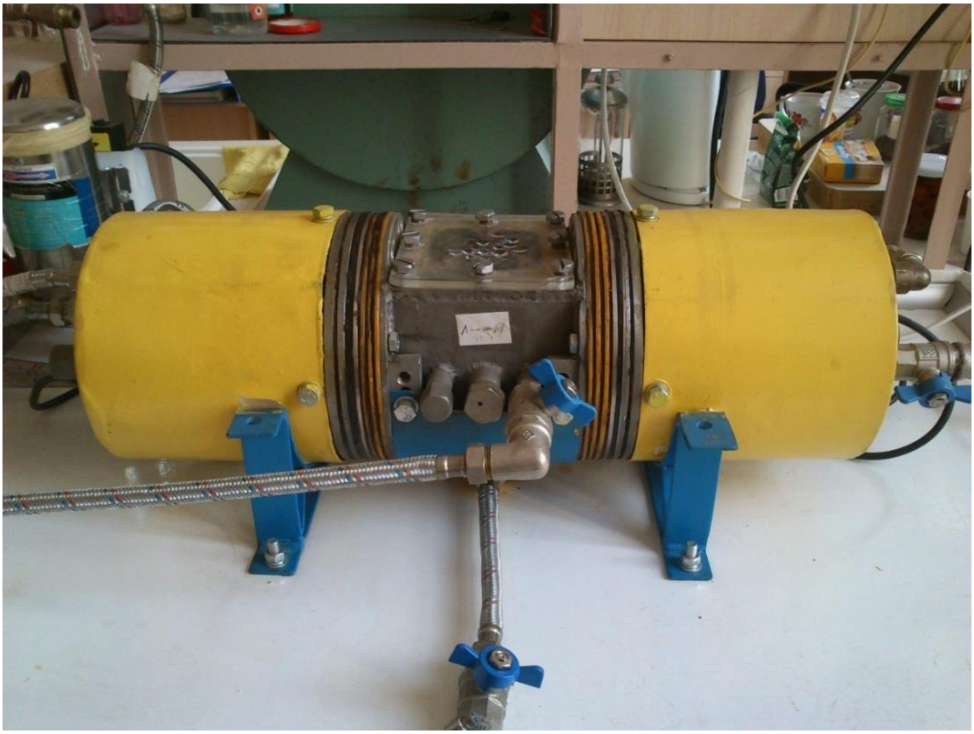

Fig. 1. A picture of a low-frequency vibratory resonance cavitator for cavitation treatment of aqueous solutions with a high content of MPN_0 and COD under pressure and at high temperatures

Considering that there are no elastic suspensions of resonant adjustment in the cavitators of this type, the voltage regulator is included in the electric power supply circuit of their drive electromagnets. It smoothly changes the frequency of drawing an anchor of an electromagnet to its stator, thus changing the frequency of oscillations attached to the anchor rod of the cavitation disturbers. As a result, there is a possibility during the treatment of liquids with this cavitator to smoothly bring the frequency of vibrations of the oscillatory disturbers to frequencies multiple to the oscillation frequencies of cavitation nuclei present in the treated liquids, that is, to maximally approximate the work of the cavitator to resonance modes. After all, the resonance modes feature the minimum energy consumption by the drive and the maximum values of their efficiency factor.

Chemical oxygen demand (COD) and the most probable number (MPN) of living microorganisms were investigated in each sample. COD was studied by the standard method (EPA method 410.4) (Starchevskyy et al., 2017). The MPN was determined by surface planting on the meat-peptone agar medium before and after treatment. Two plates were used for each dilution and incubated at 37°C for 48 h.

The investigations were carried out at the temperature 298 K and the pressure 1·105 Pa, selecting the same experiment conditions for running the processes both in the cavitation field and without it. Oxygen was used as additional gas for the researches, which was bubbled into the wastewater at the rate of ~ 1 cm^3/s. The duration of the process was 1 h. The volume of the investigated dispersion (100 cb.cm) in the glass reactor was cooled by water during the whole process. Water samples were treated during the periods of 15, 30; 45, 60 min.

3. Results and discussion

After 1 hour of treatment at the frequency of 50 Hz, the degree of destruction of live yeast cells reaches 99.8%; the decrease of the index of COD is 45%, which is satisfactory for technological introduction (Fig. 2 and 3).

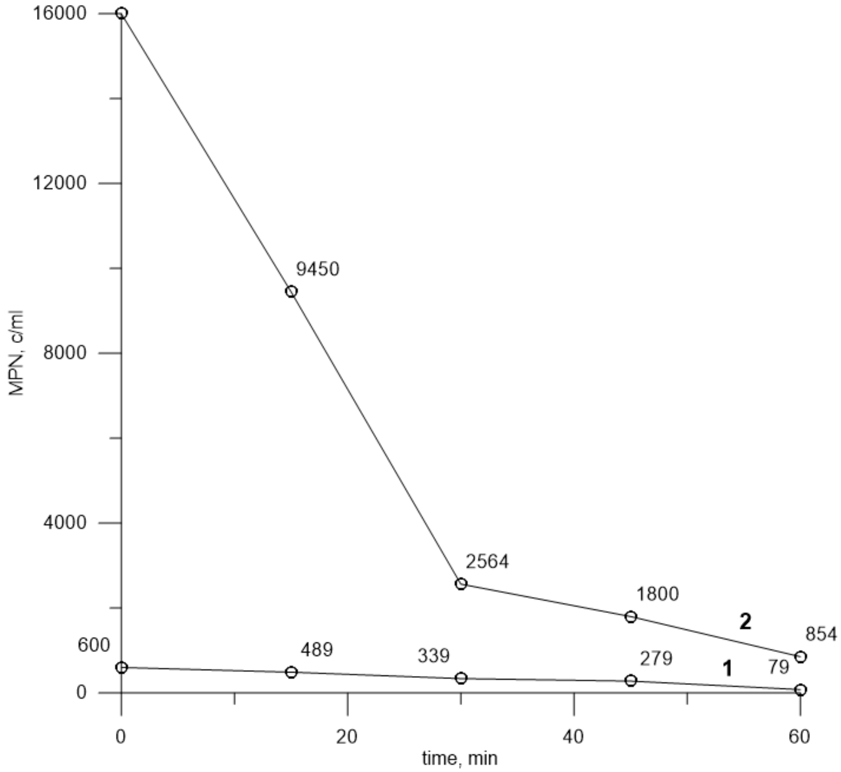

Fig. 2. Kinetic curves of changes in MPN of yeast solutions from waste waters of the yeast factory 1) and the brewery 2) during the application of vibration cavitation, referred to 100 W of electric power. Temperature: 20°C, frequency: 50 Hz

The rate of oxidation reaction describes by equation:

$$-d[RH]/dt = k_2[O_2][RH]^2 + k_p(k_i/k_t[O_2])[RH]^2 \qquad (1)$$

As may be seen from Fig. 4, the decrease of chemical oxygen demand is well described by the second-order kinetic equation ($R^2{}_1$= 0,9816; $R^2{}_2$ = 0,9898). In this particular case, there is also observed the synergetic effect of oxygen activity and cavitation.

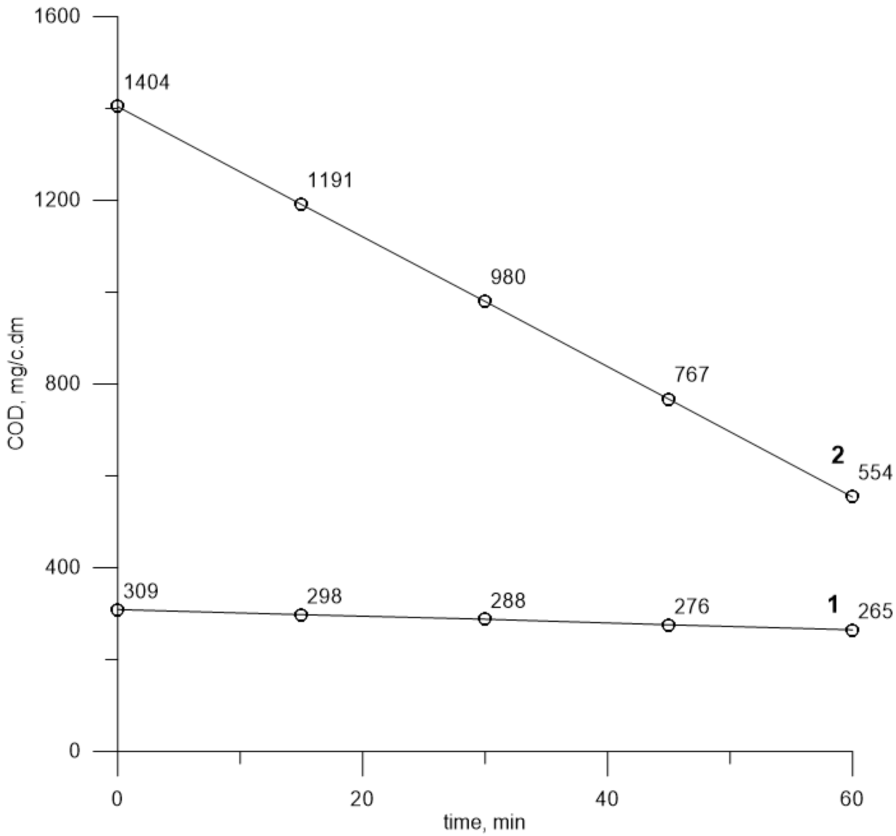

Fig. 3. Changes in time of the value of COD of yeast solutions from waste waters of the brewery 1) and the yeast factory 2) during the application of vibration cavitation referred to 100 W of electric power. Temperature: 20°C, frequency: 50 Hz

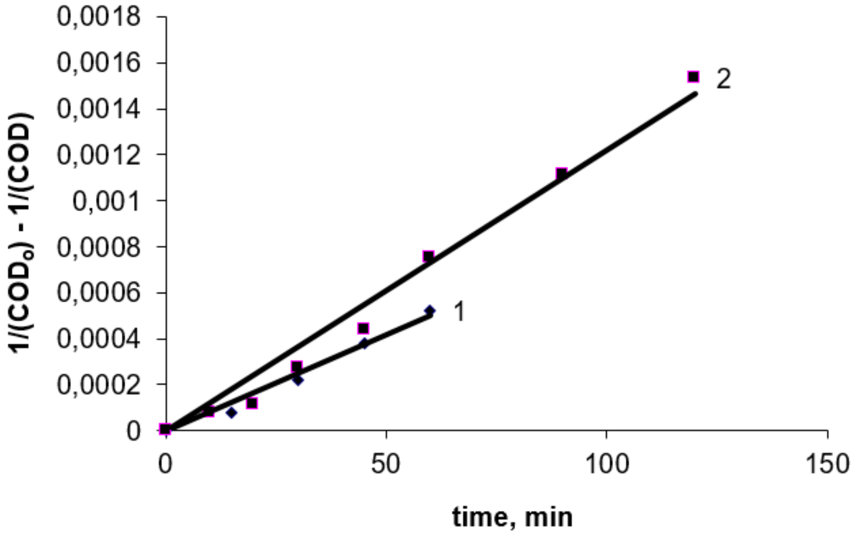

Fig. 4. Dependence of chemical oxygen demand in water after the waste treatment facilities of the brewery 1) and the yeast factory 2) on the duration of cavitation treatment in the coordinates of the second- order equation. Temperature: 20^0C, frequency: 50 Hz

The increase of the efficiency factor in these cavitators is achieved due to the experimental selection of the oscillation frequency of the cavitation disturbers, which are close to the multiple values of the resonance of the oscillation frequencies of the cavitation nuclei. This is done by using a device for investigating frequencies of the oscillations of cavitation nuclei, that are present in liquids, which are multiples to the resonant ones.

Fig. 5 demonstrates the dependence of the dying degree of yeast cells on the frequency of oscillations of cavitation disturbers and it is evident that the greatest effect was achieved at a frequency of 37 Hz.

The decrease of MPN of sewage waters of the brewery and yeast production outlined by the first order reaction levels (Tab. 1).

Table 1. Rate constants of destruction of yeast cells in water solution with the use of vibrocavitation, taken to 100W of electric power and different frequency. The temperature is 20^0C

MPN [kl/ml]	**30 Hz**	**35 Hz**	**37 Hz**	**39 Hz**
600	0,0878	0,0345	0,0342	0,0647
16000	0,0919	0,0563	0,0594	0,0732

The frequency of 37 Hz is resonant, in which the maximum result is achieved at the minimum energy consumption required for the cavitation perturbation, i.e., the oscillation frequency is multiple or equal to the frequency of oscillations of the cavitation nuclei. The vibration cavitator consumes 800 Watts and emits about 800 Watts (the efficiency factor is about 98%).

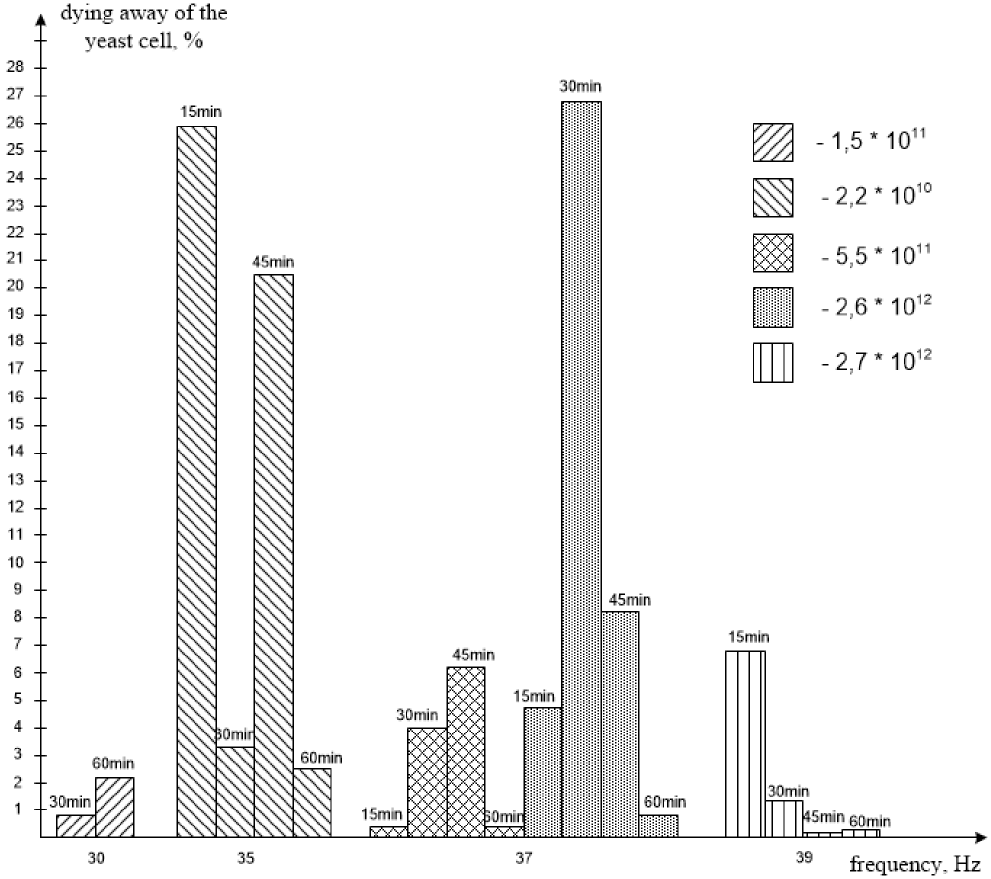

Fig. 5. Dependence of the dying degree of yeast cells on the frequency of oscillations of the cavitation disturbers and the initial concentration of microorganism cells

4. Summary and conclusions

- It has been found out that during the use of the vibration cavitator for the treatment of yeast-containing sewage waste waters, the degree of destruction of living yeast cells amounts to 99.8% and the reduction of the COD indicator is 45%, which is satisfactory for technological implementation.
- The effectiveness of vibration cavitation treatment of sewage waste waters with different concentrations of biological contaminants at different oscillation

frequency of the cavitation has been investigated. The greatest effect is achieved at the frequency of 37 Hz.

- The high efficiency factor of the vibration cavitator and low energy consumption during its use feature the method of vibration cavitation treatment of liquid media as an environmentally safe way to improve the efficiency of treatment of waste waters from food industry enterprises by removing both biological and chemical contaminants.

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Analysis of retention possibilities of selected model green roofs

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Abstract

Bio-retention infrastructure elements are becoming more and more present in urban areas. There are numerous reasons of this increasing popularity: the legal necessity to reconstruct the biological active surface, the attempt to improve the air quality by natural filtration elements, the way to reduce the effects of Urban Heat Island phenomenon, and the possibility of rain water retention. The high percentage of impervious areas in cities, limited capacity of stormwater sewers and rapid character of rain often results in local flooding, which can cause environmental, social and economic losses. The designing of green infrastructure elements can be supported by advanced informatics techniques in order to predict their impact on Integrated Water Management process. Computer simulations enable to analyze the operation of bio-retention facilities and therefore help to forecast their most efficient parameters. One of the most popular software tools is Storm Water Management Model developed by United States Environmental Protection Agency.

In this paper, an exemplary analysis of rainwater retention possibilities of selected model green roofs is presented. Models of green roofs were built basing on design projects for two different public utility buildings. For each building three types of roofs construction were testes: extensive green roof, intensive green roof and classic roof. Three different rain hydrographs were included: moderate, strong and torrential. The examined parameters included: the surface runoff during and after rain and the reduction of stormwater volume in surface runoff. The obtained results confirmed that Low Impact Development elements, such as green roofs, besides their plural environmental values, are crucial elements for a balanced stormwater management. However, the results should be verified by operation of real semi-technical and technical scale green roofs.

Keywords

green roof, SWMM, retention, Low Impact Development

1. Introduction

The proper management of stormwater is one of the main challenging tasks in urban planning. The high percentage of impervious areas in cities, limited capacity of stormwater sewers and rapid character of rain often results in local flooding, which can cause environmental, social and economic losses. The alternative solution for the stormwater management problem is the bio-retention infrastructure, which is becoming more and more present in urban areas. One of the examples of bio-retention elements is a green roof defined as a surface which is partially or completely covered with vegetation. The surface not necessarily has to be located at the rooftop of the building, or be green. The main purpose of green roofs is to reconstruct the natural soil conditions by a building. In literature, there are also many alternative terms for green roofs, such as: *living roof*, *eco-roof* or *roof garden* (Burszta-Adamiak, 2014).

The popularity of green roofs as elements of urban infrastructure is significantly increasing because of numerous reasons: the legal necessity to reconstruct the biological active surface (Szajda-Birnfeld et al., 2012), the attempt to improve the air quality by natural filtration elements (Berardi, 2014) and the way to reduce the effects of Urban Heat Island phenomenon (Mrowiec, 2008). From the point of view of stormwater system operator, the most important advantage of green roof is the possibility of rain water retention. In accordance to literature data, green roofs retention possibilities are up to 50–60% (Rabiński et al., 2013) or even 60–90% (Szajda-Birnfeld et al., 2012). Moreover, bio-retention infrastructure elements such as green roofs can decrease the maximum flow wave in sewer conduits up to 60–90% and delay it by 5 min to 2 hours (Graham and Kim, 2003; Burszta-Adamiak, 2014). Green roofs have also several environmental and social aspects: the quality improvement of stormwater by filtrating it in roof soil layers (Królikowska and Królikowski, 2012), absorption of carbon dioxide and production of oxygen (Rabiński et al., 2013), creation of new habitats for birds and insects (Mrowiec, 2008) or creation of ecological picture of a city (Burszta-Adamiak, 2012).

Despite plural positive aspects of green roofs, there are also many impediments in their wide implementation. Firstly, green roofs structure is the additional load for the building construction, so during its designing the structure calculation correction is necessary (Szajda-Birnfeld et al., 2012). Moreover, green roofs require special operation, including watering (periodic or regular), esthetic care or fertilization. The additional barrier in green roofs popularity in Poland is also the lack of precise law requirements (Burszta-Adamiak and Sylwester, 2011).

In this paper, an exemplary analysis of rainwater retention possibilities of selected model green roofs is presented. The aim of the analysis was to evaluate the possibilities of exemplary designed green roofs and comparison of retention potential with literature values.

2. *Materials and Methods*

The analysis of retention possibilities of green roofs was performed for 2 different public utility buildings (presented in the fig. 1). Object A – the Innovation and Advanced Technologies Centre of Lublin University of Technology was built in 2014, with the roof area equal to 4811.76 m^2 and roof slope: 2%. Object B – the Shopping Centre in one of the cities in Lublin Voivodships, was built in 2014, with the roof area equal to 2685.2 m^2 and roof slope: 2%. For each public utility building two different green roofs were designed – intensive and extensive green roof (Zawrotniak, 2016; Łukasik, 2016; Anaszewska, 2017, Kasprzak, 2017).

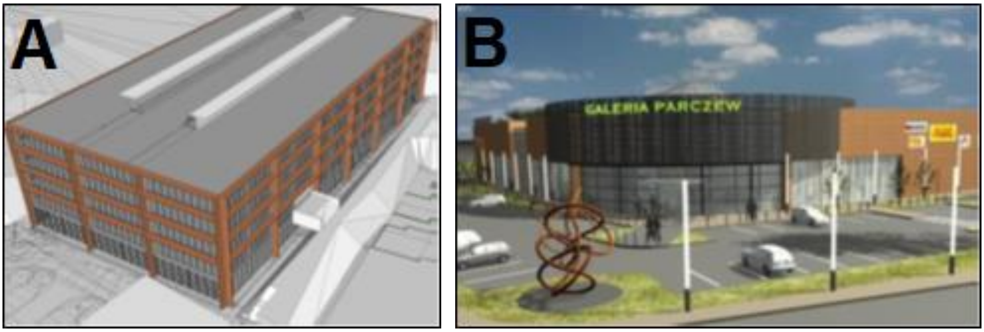

Fig. 1. Visualizations of public utility buildings (objects A and B)

To analyze the retention possibilities of designed green roofs the software Storm Water Management Model developed by United States Environmental Protection Agency (EPA SWMM 5.1) was used. The software enables to simulate the operation of many Low Impact Development facilities, including green roofs and therefore help to forecast their most efficient parameters. The representation of model green roofs in SWMM was done in three steps: 1) define a 100% impervious subcatchment with properties representative of classic, non-vegetated rooftop, 2) use of the LID editor tool to describe physical properties of the modelled green roofs (intensive and extensive for object A and B), and 3) use the LID implementation tool to include a green roof surface for a prescribed area of the subcatchment. Characteristic parameters of green roofs, such as *Surface slope [%]*, *Soil thickness [mm]* and *Drainage mat thickness [mm]* were established in accordance to design projects. Other parameters were based on DAFA (2015) and FLL (2002) guidelines (2015), and defaults values from SWMM 5.1 Users Manual (Rossman, 2015). The precise parameters of LID control for object A (A-i – the intensive green roof for object A, A-e – the extensive green roof for A) and B (B-i and B-e as previous) are presented in table 1. During the analysis, three different exemplary rain hydrographs were included: moderate (total precipitation: 13.7 mm/day), strong (20.8 mm/d) and torrential (23.6 mm/d). Rainfall hydrographs from SWMM 5.1 software are presented in figure 2. The examined parameters included: the surface runoff during and after rain and the reduction of stormwater volume in surface runoff.

Table 1. Parameters of LID controls for intensive and extensive green roof (object A and B)

ROOF TYPE	**SURFACE**				**SOIL**							**DRAINAGE MAT**		
	Berm height [mm]	**Vegetation volume fraction**	**Surface Roughness**	**Surface Slope [%]**	**Thickness [mm]**	**Porosity**	**Field Capacity**	**Wilting Point**	**Conductivity [mm/h]**	**Conductivity slope**	**Suction Head [mm]**	**Thickness [mm]**	**Void Fraction**	**Roughness**
A-i	0.0	0.1	0.41	2	500	0.489	0.4	0.1	0.00461	65	3.5	86.6	0.5	0.25
A-e	0.0	0.0	0.24	2	100	0.464	0.376	0.1	0.00461	65	3.5	40	0.5	0.25
B-i	0.0	0.1	0.41	2	300	0.489	0.4	0.1	0.00461	65	3.5	70	0.5	0.25
B-e	0.0	0.0	0.24	2	190	0.464	0.376	0.1	0.00461	65	3.5	42.7	0.5	0.25

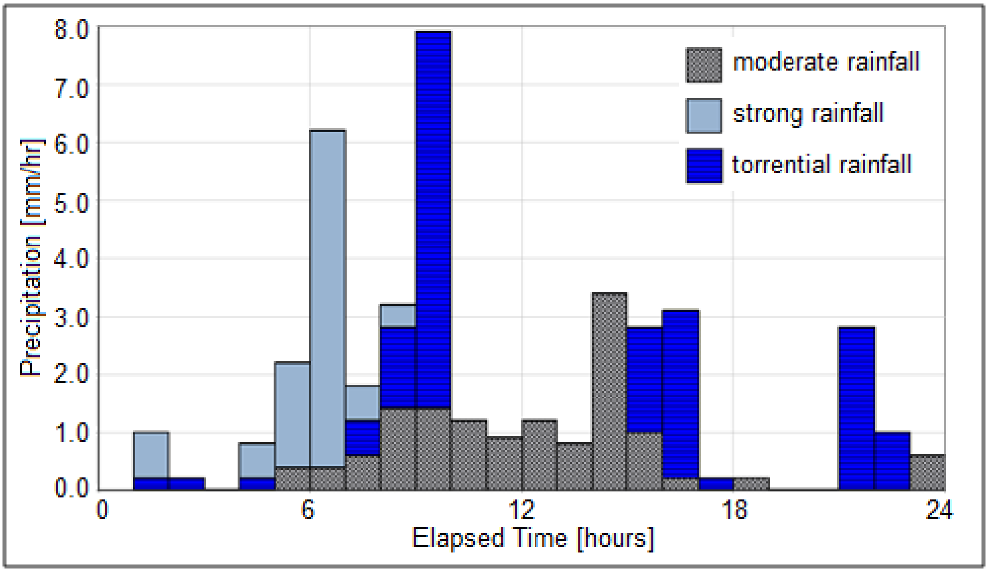

Fig. 2. Rainfall hydrographs from SWMM 5.1 software

3. Results and Discussion

In accordance to the presented analysis method, the percentage of stormwater volume reduction in runoff from green roofs in comparison with classic totally impervious rooftop is presented in table 2. The intensive green roof revealed the biggest retention potential (25.62% of the stormwater reduced during strong rainfall). In all analyzed rainfall cases the intensive green roofs reduced more volume than extensive green roofs, because of higher soil thickness. In both cases (object A and B) stormwater was reduced slightly more during the strong rainfall than during torrential rainfall. Apparently, the precipitation during torrential rainfall was too big and exceeded the retention possibilities of the analyzed green roofs.

Table 2. Percentage of stormwater volume reduction from analysed green roofs

Object	**Precipitation**	**Rooftop**		
		classic	extensive	intensive
		Stormwater volume reduction [%]		
A	moderate	-	8.53	13.50
	strong	-	17.21	25.62
	torrential	-	16.58	23.77
B	moderate	-	5.18	8.43
	strong	-	11.34	16.65
	torrential	-	11.14	16.49

Additionally, the exemplary results of calculation are presented in the form of dependence time series graphs. In figure 3 the surface runoff from object A and precipitation during strong rainfall is presented, while in figure 4 the surface runoff from object B and precipitation during torrential rainfall is shown. Analyzing the surface runoff values during strong and torrential rainfalls from object A and B, the following main specifics about the functioning of modelled green roofs can be noticed: the maximum stormwater runoff wave is significantly reduced comparing classic rooftop with green roofs surfaces. In case of object A the maximum peak of runoff (approx. 8.25 LPS) is reduced to approx. 3.5 LPS (extensive green roof) and to approx. 2.5 LPS (intensive green roof). Due to the high hourly values of precipitation during torrential rainfall the maximum peak wave reduction is relatively smaller in the case of object B. Moreover, the smaller retention possibilities of object B are also caused by differences in layer structure of green roofs (the soil layer thickness is 200 mm higher in intensive green roof for object A). Moreover, due to the fact that some part of the stormwater is being accumulated in the soil layer, the runoff from green roofs is smooth and delayed in reference to the precipitation moment while form classic roof construction the runoff is short and rapid.

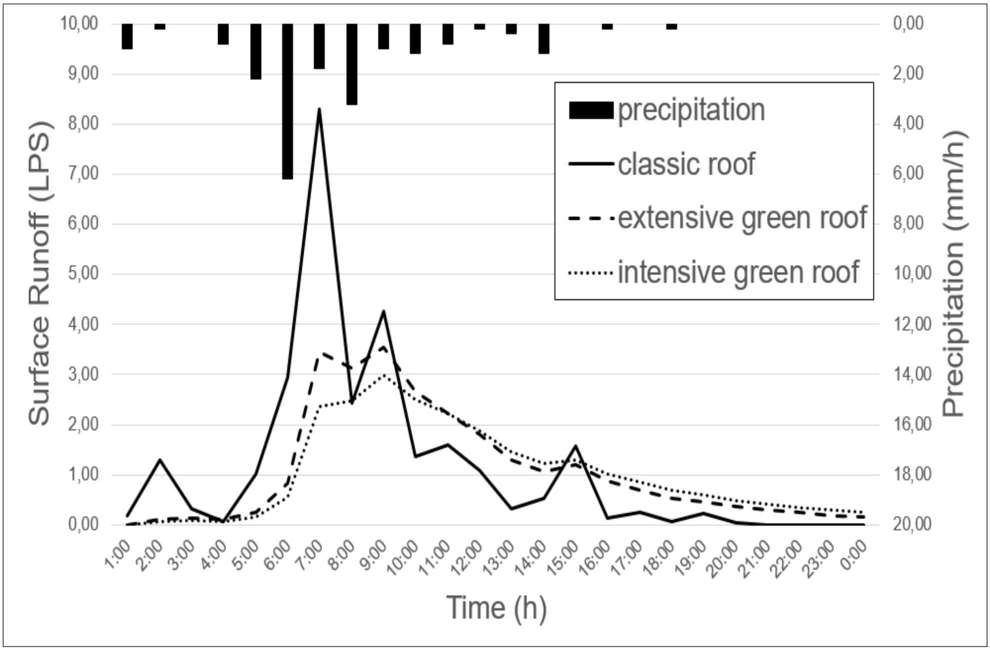

Fig. 3. Surface runoff and precipitation during strong rainfall (object A)

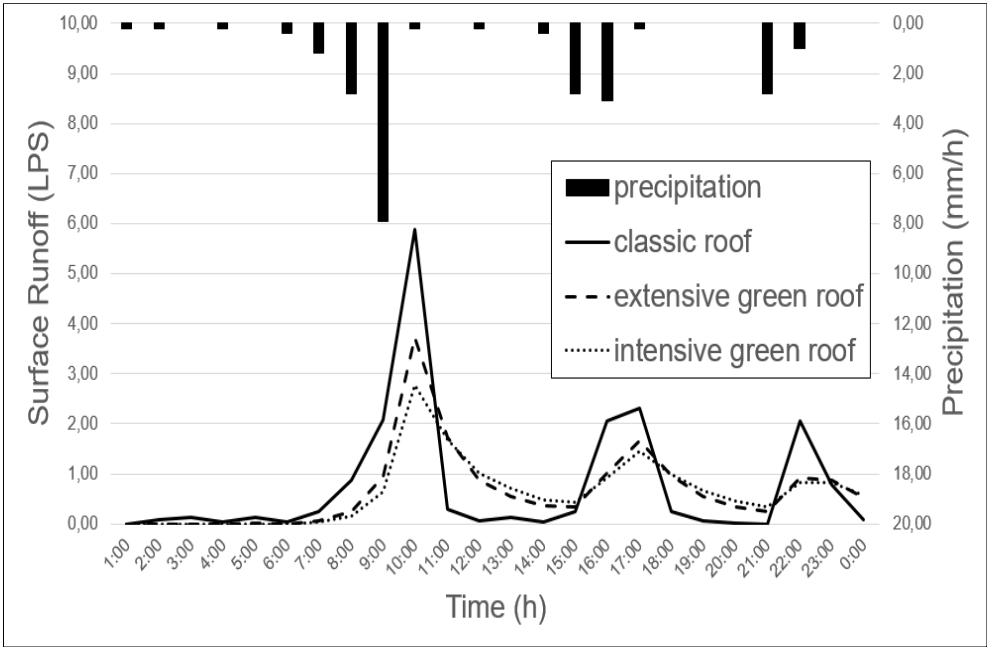

Fig. 4. Surface runoff and precipitation during torrential rainfall (object B)

4. Summary and Conclusions

The obtained results confirmed that Low Impact Development elements, such as green roofs, besides their plural environmental values, can be used as crucial elements for a balanced stormwater management. The analyzed green roofs, due to their layer structure and parameters, revealed significant retention possibilities. However, the retention potential of analyzed green roofs were smaller in comparison do literature data (Burszta-Adamiak, 2014). The reason of this smaller retention abilities is probably theoretical estimation of the green roofs parameters. Instead of literature data, particular properties of the layers should be based on real exemplary green roofs, designed especially for local weather and climate conditions.

Considering bioretention elements as supportive tools in proper stormwater management, it is recommended to build green roofs of an intensive type rather than extensive. However, the calculation results of retention possibilities should be verified by operation of real semi-technical and technical scale green roofs, as the character if its retention possibilities is individual and depends on rainfall frequency and amount. Moreover, the investment in bio-infrastructure elements, such us green roofs, reflects not only in retention benefits, but also is a major improvement in public image of a city or company.

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Anaerobic co-digestion of food industrial waste and municipal sewage sludge

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Abstract

Food industrial waste is becoming a serious environmental issue in many European countries. Every year, about 38 million tons of this type of waste is generated in EU territory. With the increasing cost of waste disposal and the growing interest in renewable energy, anaerobic co-digestion gains more attention. This process is considered as one of the most successful technologies for sustainable alternative energy recovery as well as management of organic wastes. Anaerobic co-digestion can be defined as the combined anaerobic treatment of two or more substrates with complementary characteristics. In comparison to mono-digestion, this technology offers several advantages, such as increasing the biogas production, process stabilization, improvement of the balance of nutrients and C/N ratio, dilution of potential toxic compounds and synergetic effects of microorganisms. Furthermore, it allows to decrease the emission of greenhouse gases to the atmosphere, improve the economic viability as well as reduce the potential pollution of wastes. Various substrates have been co-digested anaerobically in a successful way, such as municipal waste, industrial waste and agricultural waste. Agro-food industries generate large amounts of waste biomass, which can be easily used in this technology. However, inappropriate selection of co-substrates may cause the process instability and significant reduction of methane production. The chemical compositions and biodegradability are the factors that should be taken into account to choose the best co-substrate. Another key factor is the availability on the local market. The aim of this study is to present a review of the achievements and perspectives of anaerobic co-digestion of food industrial waste and municipal sewage sludge. In this work, the main co-substrates have been presented, such as cheese whey, brewery spent grain, slaughterhouse waste, and sugar beet pulp.

Keywords

co-digestion, sewage sludge, biogas production, organic waste, food industry

1. Introduction

Nowadays, resource depletion, energy insecurity as well as environmental pollutions are the greatest concerns of our time. For this reason, a change in resource and environmental policy is highly required (Zhan et. al, 2016; Hagos et al., 2017). One of the most successful technologies for sustainable energy recovery as well as management of organic wastes is anaerobic digestion (AD) (Naran et al., 2016). It could be defined as a biological process performed in the absence of oxygen to stabilize organic matter. The main products of AD are biogas which mainly contains methane, carbon dioxide, ammonia, and trace amounts of other gases as well as biosolids which are susceptible of valorization by land application. Anaerobic digestion is a highly sensitive and complicated process involving many groups of microorganisms with ultimate operational environmental conditions (Hagos et al., 2017). It includes the following steps: hydrolysis, acidogenesis, acetogenesis and methanogenesis (Mata-Alvarez et al., 2014).

Many kinds of organic waste such as industrial waste, manure, fruit and vegetable waste, slaughterhouse waste and agricultural biomass have been digested successfully (Murto et al., 2004). However, sewage sludge processing has become the most widespread application of AD . However, digestion of single substrates which is called mono-digestion presents some disadvantages (Mata-Alvarez et al., 2014). Many problems refer to the substrate composition. Sewage sludge is generally characterized by low volumetric productions of biogas because of the inherent difficulties in digesting secondary sludge that presents a lower methane yield than that of primary sludge (Fernández et al., 2014). Animal manure contains high nitrogen concentrations that could inhibit methanogenic activity and low organic load. In the case of slaughterhouse wastes, there is also the possibility of methanogens inhibition caused by high concentration of N or the presence of long chain fatty acids. On the other hand, the fruit and vegetable wastes are not available all the time. The organic fraction of municipal solid waste has a relatively high concentration of heavy metals. Many different improvements in mono-digestion process have been presented such as mechanical, chemical and thermal, ultrasound, microwave pre-treatments (Mata-Alvarez et al., 2014; Zawieja et al., 2015). However, the most promising way to increase biogas production and improve AD process is the addition of appropriate co-substrate with high biogas/methane potential. This process is known as anaerobic co-digestion (AcoD). In generally, it involves simultaneous anaerobic digestion of two or more substrates with complementary characteristic (Grosser et al., 2017). Compared to mono-digestion, co-digestion presents many benefits. As was mentioned above, the main advantage is the improvement of biogas production and methane yield. Enhanced process stabilization and nutrient balance in the feedstock constitute other benefits. Moreover, this approach allows to dilute some of the toxic compounds. Besides, co-digestion could improve the content of macro- and micronutrients and the biodegradable fraction in the digester feed. The AcoD gives an opportunity to manage many kinds of wastes, which could cause inhibition of the process or can hamper digestion stability when treated separately (Bień et al., 2010). This technology could provide many economic as well as environmental benefits, such as reducing the emission of greenhouse gases to the atmosphere, and improve the profitability of many wastewater treatment plants (Grosser, 2017). Currently, the operation of WWTPs is relatively energy consuming, about 3% of global electricity demand corresponds to WWTPs functioning. In practice, a typical WWTP can offset only 20–30% of the total energy consumption. Thus, this technology allows many WWTPs to become energy self-sufficient and reduce the cost of sewage sludge management (Grosser, 2017; Nghiem et al., 2017). Another economic advantage is the fact of sharing the apparatus and cost of digestion of co-substrates. Additionally, the application of this approach is not problematic, does not require many changes in construction of reactors. Moreover, many digesters at existing WWTPs are over dimensioned (Bień et al., 2010).

However, certain disadvantages of AcoD are also known, mainly due to transport or/and storage costs, harmonization of the WWTPs operation and co-substrate generators as well as selection of an appropriate proportion of substrates. In some cases additional pretreatment of co-substrate is also needed (Mata-Alvarez et al., 2014). A continuously increasing interest in anaerobic co-digestion applications has been observed over the last 10 years (Xie et al., 2016). Currently, there are a number of full-scale digesters in operation (Murto et al., 2004).

2. Co-substrates in anaerobic co-digestion process

Successful substrate selection is crucial to achieving synergistic effects during the anaerobic co-digestion. Preferable properties include the high content of readily biodegradable organic fraction that increase the kinetics of biogas production as well as provide sufficient nutrients content (Astals et al., 2014; Xie et al., 2016). Another required features include high buffering capacity to avoid pH shock and a balanced C:N ratio to maintain the methanogenic activity (Wang et al., 2012). A low concentration of nitrogenous matter in co-substrate to reduce free ammonia inhibition is also highly demanded (Yenigün and Demirel, 2013). Moreover, relatively low sulfur content to suppress the activity of sulfur reducing bacteria is also highly desirable co-substrate property (Chen et al., 2008; Xie et al., 2016). Furthermore, the high biogas potential of co-substrates is also a crucial parameter that should be taken into account.

Thus, another thing to consider is the transportation cost from the generation point to a wastewater treatment plant. For this reason, appropriate identification of the local market is needed. The most preferred wastes are the ones for which there is no other possibility for management. Another crucial aspect in co-digestion is finding the blend ratio as well as operating conditions in order to promote the positive interaction, dilute inhibitory or toxic compounds, optimize the methane production as well as preserve the digester stability (Mata-Alvarez et al., 2014; Fonoll et al., 2015).

2.1. Sewage sludge as main substrate

Sewage sludge (SS) is a complex by-product generated during the wastewater treatment process. With the significant improvement of biological treatment methods, a serious problem associated with increase production of the sewage sludge at wastewater treatment plants has been observed (Rulkens, 2007). In Poland, estimated 951.5 thousand tons of dry solid of sewage sludge were produced in 2016. Mostly, it was treated in anaerobic digesters. Typical sewage sludge comprises the primary sludge separated from wastewater during pre-settling and biological excess sludge from the activated sludge system.

Quantitative and qualitative composition of the sewage sludge is very complex and depends on many factors e.g. technology used at WWTPs, water consumption and the presence of local industry. Usually, it is rich in organic matter, nitrogen, phosphorus, calcium, magnesium, sulphur and other microelements that can be reused as fertiliser or soil improver. On the other hand, sludge can contain toxic compounds (heavy metals, pesticides) and pathogenic organisms (bacteria, eggs of parasites) (Rulkens, 2007; Kosobucki et al., 2000). Moreover, it is also characterized by high moisture content varying from a few percent to more than 95% and its typical methane production potential is approximately 300–400 dm^3/kg VS added (Einola et al., 2001; Davidsson et al., 2008). The fundamental problem of sewage sludge is that all of these compounds are present in one mixture (Rulkens, 2007). A typical chemical composition and properties of untreated and digested sludge is presented in table 1.

Due to a relatively low C/N ratio and high buffer capacity, sewage sludge could be co-digested with co-substrates with high amounts of easily biodegradable organic matter and with low alkalinity values. Moreover, co-digestion of sewage sludge with some of the substrates allows the dilution of certain unwanted compounds in SS such as heavy metals, pharmaceuticals and/or pathogens (Iacovidou et al., 2012; Dai et al., 2013).

Table 1. Typical chemical composition of primary and activated sludge (Techobanoglous et al., 2003)

Parameter	Primary sludge	Activated sludge
Total dry solids (TS) [%]	2.0–8.0	0.83–1.16
Volatile solids [% of TS]	60–80	59–88
Protein [% of TS]	20–30	32–41
Nitrogen [% of TS]	1.5–4	2.4–5.0
Phosphorous [% of TS]	0.8–2.8	28–11.0
Potash [% of TS]	0–1	0.5–0.7
Cellulose [% of TS]	8.0–15.0	-
Iron [% of TS]	2.0–4.0	-
pH	5.0–8.0	6.5–8.0
Alkalinity [$mgCaCO_3/dm^3$]	500–1500	580–1100
Organic acids [mg/dm^3]	200–2000	1100–1700

From this point of view, the organic fraction of municipal solid waste as well as oil and grease waste generated by the grit chamber at the WWTP, are convenient co-substrate for SS. Additionally, many types of agro-industrial wastes produced near the WWTP could be also used (Mata-Alvarez et al., 2014).

2.2. Food industry wastes as co-substrates

Food industry by-products have shown to be a potential biomass to produce clean energy using anaerobic co-digestion process (Anwar et al., 2014; Mata-Alvarez et al., 2014; Ward et al., 2008; Aboudi et al., 2016). Besides, food industry represents one of the leading sectors of the economy in Poland. Lubelskie voivodeship is one of the largest and most important agricultural regions of the country. Agricultural land accounts for 70% of the voivodeship area. There are numerous dairies, meat processing companies, breweries, spirits as well as herbalists companies. This developed production leads to the generation of large amounts of waste (Pellera and Gidarakos, 2017). Those by-products require appropriate management, in order to avoid the potential environmental problems related to their disposal. Thus, the application of these materials for energy production in anaerobic co-digestion would seem as a suitable solution (Aboudi et al., 2015, 2016; Anjum et al., 2016).

2.2.1. Cheese whey

One of the examples of food processing industry that generated large amounts of various residuals that can be used as substrate for biogas generation is dairy industry (Hagen et al., 2014). During cheese manufacturing, two main effluents can be distinguished: washing and pasteurization waters, mixed with detergents and milk as well as cheese whey (CW). The first one, characterized by low organic load, can be treated on-site in suitable aerobic treatment units, while the second stream has a high organic load and requires more advanced treatment (Chatzipaschali and Stamatis et al., 2012).

Generally, cheese whey is the liquid byproduct remaining after the precipitation and removal of milk casein during cheese-making (Teixeira et al., 2010). The EU countries are worldwide leaders in cheese production, approximately 9.8 mln tons of cheese were produced in EU in 2016 (USDA, 2017). About nine kilograms of cheese whey is generated from every kilogram manufactured cheese. Additionally, the whey production rate is increasing with similar rates of milk production. Annually, the global growth rate of whey generation is around 2% (Smithers, 2008). Although this byproduct finds application in the food industry (sports, medical nutrition production), pharmaceutical industry or serves as animal feeding, in some localized areas, it may become a serious environmental concern (Fernandez et al., 2015). Still, about 50% of total CW production is being disused (Marone et al., 2015). It should be also noted that the transportation cost of this waste is significant, because of the high water content, about 92–95%. Therefore, transportation over long distances is unprofitable (Rico et al., 2015). For that reason, the place of processing, disposal or use should be located close to the cheese manufacture.

The detailed composition of this side-stream may be different, depends mainly on the source and the composition of milk. Moreover, the cheese production technology and operational conditions such as the type of acid as well as the period and temperature of coagulation also have influence on its characteristic. It can be obtained from any kind of milk, including cow, goat, sheep and camel milk (Kavacik et al., 2010; Chatzipaschali and Stamatis et al., 2012). Mostly, cheese whey contains a large amount of carbohydrates (4–5%), mainly lactose (45–50 g/dm^3), proteins (6–8 g/ dm^3), lipids (4–5 g/dm^3) and mineral salts (8–10% of dried extract). The mineral salts include NaCl and KCl (>50%), calcium salts and others. Additionally, CW includes significant quantities of lactic (0.5 g/ dm^3) and citric acids and B-group vitamins (Prazeres et al., 2012; Dareioti et al., 2015). According to the production process and the coagulation of casein two main type of cheese whey can be distinguished: acidic whey, which has a pH of less than 5 ($pH < 5$), and sweet whey with a pH value between 6 and 7 ($6 < pH < 7$) (Bylund, 1995).

Table 2. Comparison of composition of sweet and acid whey (g/dm^3) (Chatzipaschali and Stamatis, 2012)

Components	Sweet whey	Acid whey
Total solids	63–70	63–70
Lactose	46–52	44–46
Proteins	6–10	6–8
Calcium	0.4–0.6	1.2–1.6
Phosphate	1–3	2–4.5
Lactate	2	6.4
Chloride	1.1	1.1

The acid whey contains less proteins and more salt content than the sweet one. Besides, it has an acid flavor. Those aspects determine its limited application (Mawson, 1994; Siso, 1996; Venetsaneas et al., 2009). Comparison of composition of sweet and acid whey is presented in table 2. Furthermore, this by-product is characterized by high organic matter content (up to 70 000 mg/dm^3 chemical oxygen demand COD), high biodegradability (approximately 99%), and relatively high alkalinity (approx. 2500 mg/dm^3 $CaCO_3$) (Mawson, 1994; Ergurder et al., 2001; Comino et al., 2012). Besides, cheese whey contains some heavy metals such as Cd, Cr, Cu, Hg, Pb and Zn in trace amounts (Cimino et al., 1990). A detailed composition of cheese whey is listed in table 3.

Table 3. Typical chemical composition of cheese whey (Carvalho et al., 2013)

Parameter	Reference				
	Ghaly and Singh (1989)	Malaspina et al. (1996)	Farizoglu et al. (2004, 2007)	Saddoud et al. (2007)	Ebrahimi et al. (2010)
pH [-]	5	-	4.5–5.0	4.9±0.3	6.0–6.5
BOD_5 [g/dm^3]	40–60	-	-	37.7±2.8	27–36
COD [g/m^3]	75.8	68.8±11.5	73–86	68.6±3.3	50–70
BOD_5/COD [g/dm^3]	0.53–0.79	-	-	0.55	51–0.54
TS [g/kg, g/dm^3*]	56.8*	3.19	-	5.93±0.38	55–65*
TSS [g/kg, g/dm^3*]	21.82*	1.3±1.14	20–22*	1.35±0.06	10–15*
TN [g/dm^3]	-	-	0.9–1.2	-	-
TNK [g/dm^3]	1.5	1.46±0.26	-	1.12±0.01	0.01–0.02
N-NH_4^+ [g/dm^3]	0.27	0.064	0.06–0.15	-	-
N-NO_3^- [mg/dm^3]	-	9.1	7–10	-	-
TP [g/dm^3]	-	0.38±0.05	0.42–0.54	0.05±1.8^{x}10^{-3}	-
P-PO_4^{3-} [g/dm^3]	-	-	0.34-0.43	-	-

Due to its high organic load and high biodegradability, biological processes are required to manage this side-stream. However, the application of aerobic treatment such as activated sludge process is significantly uneconomical. The other alternative is anaerobic digestion, because of the lower energy requirements, the generation of methane and lower amount of sludge are produced (Fernandez et al., 2015; Rico et al., 2015).

However, the anaerobic digestion of cheese whey presents some disadvantages. The main difficulties during digestion this substrate are due to the low alkalinity content and the rapid acidification of cheese whey that can exhaust the buffering capacity, leading to a drop in pH, volatile fatty acids (VFA) accumulation and subsequent reactor failure (Kalyuzhnyi et al.,1997; Ergüder et al., 2001; Rico et al., 2015).

Another problem with AD of this substrate is the difficulty in obtaining granulation and the tendency to produce an excess of viscous exopolymeric materials that severely reduces the sludge settleability and can cause biomass wash out in high-load anaerobic reactors (Malaspina et al., 1996). A mono-digestion of CW is also characterized by low biogas production and methane yields that are associated with the low pH of this substrate (Lo et al., 1989; Yan et al., 1990; Ghaly, 1996; Comino et al., 2009). An alternative solution could be the application of a co-digestion process. For this purpose, the complementary co-substrate with adequate characteristic should be selected. Many successful examples of co-digestion of cheese whey and dairy manure were presented (Rico et al., 2015; Kavacik and Topaloglu, 2010; Ghaly, 1996). However, the agricultural biogas plants are not available everywhere. Recent studies also indicated that cheese whey can be a suitable substrate for co-digestion with sewage sludge. As was mentioned above, sewage sludge should be co-digested with substrates that are characterized by high amounts of easily biodegradable organic matter and with low alkalinity values such as CW presents. In the co-digestion process, sewage sludge can provide the necessary buffer capacity to ensure the process stability. The higher C/N ratio (C/N = 11–24) of acid cheese whey could also ensure more optimal ratio in the feedstock, that could improve the biogas production. However, some problems caused by impropriate dose of CW and incorrect operational conditions might be found (Zielewicz et al., 2012). The inhibitory effect might be caused by the accumulation of volatile fatty acids. Fernandez et al., (2014) indicated that higher temperature corresponding to thermophilic conditions could help to avoid this inhibitory effect. However, in this study lower methane yields were obtained under thermophilic temperatures than mesophilic. A successful example of co-digestion of these two substrates was achieved in the study conducted by El-Mashad and Zhang (2010), the COD loading ratio of primary sewage sludge to cheese whey were 70:30 v/v and HRT = 20 and 10 days, respectively. In the research conducted by Maragkaki et al., (2017) a thermal dried cheese whey was also applied as another co-substrate to food waste, olive mill wastewater to co-digest with sewage sludge. The laboratory experiments were performed at concentrations of 3%, 5% and 7% and hydraulic retention time (HRT) of 24 days.

2.2.2. Brewer's spent grain

Another waste which has a potential to be a source of renewable energy is brewer's spent grain (BSG) (Gonçalves et al., 2015). It is the main by-product generated in breweries, corresponding to around 85% of total wastes produced by this industry (Mussato et al., 2006; Xiros et al., 2008). BSG is the insoluble residue that is separated from the mash before fermentation (Fig. 1.) (Mussato et al., 2006; Gonçalves et al., 2015). The EU is the second largest beer producer in the world, after China. There are over 6,500 active breweries, which produced around 383 million hectolitres of beer in 2014. On the basis of the available technologies, approximately 15–20 kg of BSG is generated per every hectolitre of beer, resulting in the annual production of 7 million tonnes of BSG in EU (Niemi et al., 2012).

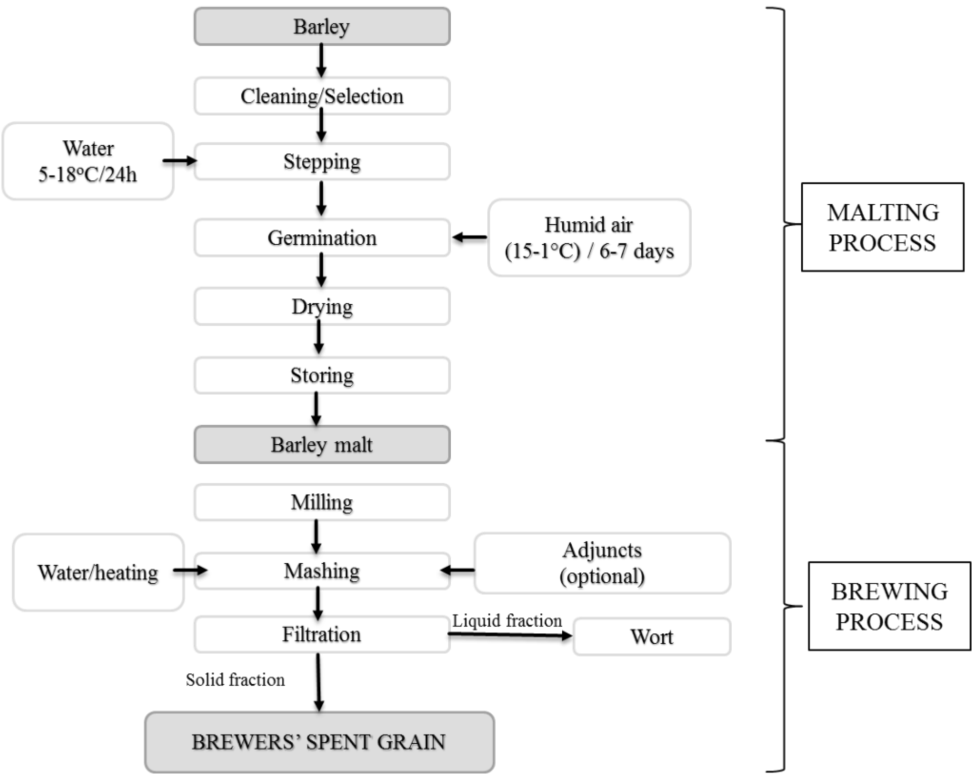

Fig. 1. Brewer's spent grain generation from natural barley (Mussatto et al., 2006)

The chemical composition of this substrate depends on many factors such as barley variety, harvest time, malting and mashing conditions as well as the quality and type of adjuncts added in the brewing process (Huige, 1994; Santos et al., 2003; Xiros et al., 2008). BSG is mainly composed of (on dry weight basis) cellulose (16.8–25.4%), hemicellulose, particularly arabinoxylans (21.8–28.4%), lignin (11.9–27.8%) and protein (15.2–24.0%) (Perez et al., 2002; Gonçalves et al., 2015). Besides, minerals, vitamins and amino acids are found in BSG. The mineral elements such ascalcium, cobalt, copper, iron, magnesium, manganese, phosphorus, potassium, selenium, sodium and sulphur are presented in concentrations lower than 0.5% (Huige, 1994; Mussato et al., 2006). Generally, lignocellulosic wastes are characterized as carbon rich, poor in buffering capacity and deficient in nutrients. The C/N ratio of BSG is less than 25 and methane yield exceeds 300 dm^3 CH_4/kg TS (Mata-Alvarez et al., 2014).

Commonly, it is mainly used as animal feed, compost or disposed of on a landfill. However, this valuable material should be utilized in a sustainable way that makes use of its potential. (Xiros et al., 2008; Niemi et al., 2012). Moreover, because of its high humidity more than 70% v/v and fermentable sugar content, brewer's spent grain is a very unstable material and quickly loses its properties (Valverde, 1994, Mussato et al., 2006). For this reason, the application and marketability of BSG as feedstuff is significantly limited. In many cases, the pretreatment of BSG is required. Physical, chemical (alkali and acid), thermal as well as enzymatic methods have been used (Macheiner et al., 2003). However, the most common method is dried. It allows to reduce the volume and maintain the main properties of the product. Thermal methods are rather expensive, but they significantly decrease the transport and storage costs (Santos et al., 2003).

Mono-digestion of this lignocellulosic material often results in a long retention time and low methane yield (Sawatdeenarunat et al., 2015). Additionally, one of the major barriers to anaerobic digestion of BSG is the presence of lignin, which blocks the enzymatic hydrolysis on the internal cellulose and hemicellulose (Wu Yi-R and He, 2013). What is worth noting is that recent studies indicated the mesophilic anaerobic bacteria are capable of producing extra cellular enzymes named cellulosomes that degrade cellulose and hemicellulose. However, the process is still poorly understood (Wu Yi-R and He, 2013; Gonçalves et al., 2015).

Thus the biogas production is not efficient enough without additional substrate treatment before or during the biogas production process (Cater et al., 2015). Several methods have been described such as mechanical, biological, chemical, and thermal treatments, or a combination of these. However, physicochemical pretreatments (adding 0.2 M NaOH at 70^{o}C, crushing by wet rotor grinding, or ball milling) demand high energy inputs and application of aggressive chemicals that are environmentally unfriendly and economically unsuitable. Many examples of biological methods where presented e.g. application of anaerobic bacteria, wood degrading fungi, actinomycetes or enzymes that selectively degrade lignin, cellulose and hemicellulose (Mussato et al., 2006; Cater et al., 2015). Those methods significantly reduce the retention time and consequently improve the economics of the anaerobic process (Panjičko et al., 2017). However, recent studies indicated that mono-digestion of pretreated BSG (mechanical, chemical and thermo-chemical methods) and untreated-raw BSG could be inhibited by phenolic degradation products, mainly p-cresol (Sežun et al. 2011). Due to a high protein content (C/N ratio of 3–5 and total nitrogen (TN) of 11–13 g/kg of wet weight), which may lead to ammonia inhibition when BSG is used as a mono-substrate (Sung and Liu, 2003). Lately, a two-stage system was also used for anaerobic digestion of BSG. In the first solid-state anaerobic digestion reactor, microbiological hydrolysis and acidogenesis occurred and in a granular biomass reactor mostly methanogenesis was performed. The overall process exhibited total solids degradation efficiency between 75.9 and 83.0% was achieved. Average specific biogas production was 414 ± 32 dm^3/kg, whereas the biomethane production was 224 ± 34 l dm^3/kg of added total solids. P-cresol was also presented in the concentrations up to 45 mg/dm^3, but during the process it was successfully degraded by granular biomass (Panjičko et al., 2017).

The limitations and problems occurred during the mono-digestion of BSG could be overcome by using different co-substrates. A significant advantage of this waste is that BSG is a low-cost material which is widely available through the year. However, dilution or additional pretreatment of this substrate is necessary to perform co-digestion (Panjičko et al., 2017). There are many successful examples of co-digestion with this material. BSG was co-digested with Jerusalem artichoke (5:1 total solids (TS) ratio of BSG vs. Jerusalem artichoke) under thermophilic (55°C) and mesophilic (30°C) conditions. Under the thermophilic conditions, the maximum total methane production reached 64%, and it comprised around 6–8 and 9–11 of l CH_4 per 100 g of fermented BSG without and with co-digested JA, respectively (Malakhova et al., 2015). Another successful example of co-digestion was co-digestion brewery wastes (brewery spent grain and trub) with glycerol. The co-digestion with 10% (v/v) of glycerol increased the methane production from 199±2 dm^3 CH_4/kg COD to 328±5 dm^3 CH_4/kg COD (Costa et al., 2013). BSG was also co-digested with sewage sludge and digested maize in batch experiments. During the experiment, specific biogas production of 150 dm^3/kg VS and 769 dm^3/kg VS of added mixture were achieved (Dido et al. 2014). Moreover, Tewelde et al. (2012) applied cattle dung as a co-substrate with BSG (70:30 v/v) achieving a specific biogas production of 410 dm^3/kg VS of added mixture in a continuous experiment with a hydraulic retention time of 40 days.

2.2.3. Slaughterhouse waste

According to Food and Agriculture Organization of the United Nations, the meat production in the world has increased threefold since 1960s (Ortner et al., 2014). Due to the growing demand of this product, the amount of organic solid wastes generating during meat processing is also significantly increased. Additionally, as much as 50% of the total weight of the animal is not suitable for human consumption. For this reason, a large amount of meat by-products are generated every year (Pagés-Díaz et al., 2017). Currently, the meat industry has the largest share in the food industry in Poland. In 2015, about 6157.4 thousand tons (in live weight) of animals for slaughter were produced in this country (GUS, 2016).

Slaughterhouse waste (SHW) usually consists of blood, manure, offal and paunch contents, so that its disposal can lead to serious environmental problems (Ahmad et al., 2014; Escudero et al., 2014). This kind of waste contains large amounts of biodegradable organic matter, mainly composed of proteins and lipids as well as pathogens (Gwyther et al., 2012, Marcos et al., 2012). It is characterized by high BOD and COD and nitrogen concentrations and low C/N ratio. Generally, the average levels of COD for this waste range from 18 000 mg/dm^3 to 43 000 mg/dm^3 (Wang et al., 2002). This value can be higher and reach even 100,000 mg COD/dm^3, if the blood content is significant (Ahmad et al., 2014).

Because of their composition (especially high content of fat and protein), slaughterhouse wastes are attractive substrates for biogas production (Pitk et al., 2012; Pagés-Díaz et al., 2014). The theoretical biogas yield from fat can reach up to 1250 dm^3/kg TS with around 67–68% methane content, whereas the corresponding value for carbohydrate is between 790 and 800 dm^3/kg TS and 50% CH_4 content (Weiland, 2010; Borowski and Kubacki, 2015). Moreover, the production in slaughterhouses is high energy demanding (conveyor systems, heat and vapor productions, slicing and cleaning machines, etc.). Thus, anaerobic digestion of their wastes to produce energy which can be reused internally is recommended (Rodríguez-Méndez et al., 2017). However, one of the major problems associated with the anaerobic treatment of this waste is the high fat and lipid content that may contribute to inadequate mixing, blocking the pipes, causing the breakdown of stirrers as well as foam formation in digesters. Furthermore, mono-digestion of SHW might lead to the accumulation of ammonia, volatile fatty acids (VFA) and long chain fatty acids (LCFA). That could inhibit methanogenesis, which results in reduced biogas production (Cuetos et al., 2008; Hejnfelt and Angelidaki, 2009; Bayr et al., 2012; Bjorn et al. 2012; Moestedt et al., 2016,). A solution of this problem may be the co-digestion of slaughterhouse wastes with other waste types containing less N-rich substrates, e.g. sewage sludge, food waste or energy crops (Pitk et al., 2013). Additionally, in the literature this by-product have been successfully co-digested with many different wastes such as pig manure (Edstorm et al., 2003; Alvarez and Liden, 2008; Hejnfelt and Angelidaki, 2009), fruit and vegetable waste (Alvarez and Liden, 2008), organic fraction of municipal solid waste (Cuetos et al., 2008; Zhang and Banks, 2012) and pharmaceutical waste (Braun et al., 2003).

Sewage sludge was also used as a co-substrate. In the study conducted by Borowski and Kubacki (2015), co-digestion of sewage sludge with large amount of slaughterhouse wastes (50% SHW weight basis) was undertaken. Batch and semi-continuous experiments were performed at 35°C. In this study, the achieved methane yield was approx. 600 dm^3/kg VS. However, when the loading rate exceeded 4 kg VS/m^3 d, a slight inhibition of methanogenesis was observed. Luste and Luostarinen (2010) evaluated the effect of hygienization and organic loading rate in anaerobic co-digestion of a mixture of animal by-products from meat-processing industry and of sewage sludge. The three reactors were fed with meat by-products and sewage sludge in a ratio of 1:7 (v/v), in the same ratio but with hygienization (70°C, 60 min) and in a ratio of 1:3 (v/v). Hydraulic retention time (HRT) was decreased from 25 to 20 days and finally to 14 days, while organic loading rates (OLR) ranged from 1.8 to 4.0 kg VS/m^3 day. In this research, co-digestion of by-products from meat-processing industry and sewage sludge (respective feed ratios 1:7 and 1:3), gave the highest methane yield and the steadiest digestate quality at 20-days-HRT. Moreover, hygienization pre-treatment of the feed ratio 1:7 was found efficient at improving the degradation and thus increasing the methane production. Another example of co-digestion of these products was presented by Pitk et al. (2013). In this study, the obtained results showed that sterilized solid slaughterhouse waste addition in the feed mixture up to 5% (v/v), corresponding to 68.1% of the organic loading, increased the methane production by 5.7 times, without any indication of process inhibition. Further increase of this co-substrate addition at 7.5% (v/v) caused a decrease of methane production and volatile solids removal reduction that was mainly related to the remarkably increased free ammonia concentration in the digester of 596.5 ± 68.6 gNH_3/dm^3. Sterilized mass addition of 10% (v/v) caused intensive foaming, LCFA accumulation of 9172 ± 701.2 mg/g and termination of the experiment.

2.2.4. Sugar beet pulp

Another example of agro-industrial waste that has a great biotechnological potential is sugar beet pulp (SBP). It is the main by-product of the sugar beet industry, which is produced annually in large amounts (Hutnan et al., 2000). Currently, the European Union countries are the world's biggest producer of beet sugar. The most significant share in this market corresponds to France, Germany, the United Kingdom as well as Poland. In 2016/2017 about 2 084 thousand tons of sugar were generated in Poland. About 170–330 kg of wet sugar beet pulp is generated from a single ton of the processed sugar beet (Fang et al., 2011; Stoyanova et al., 2014; Borowski et al., 2016).

Commonly, this by-product is used as animal feedstock or soil fertilizer (Medina et al., 2007). However, because of the high water content (70–80%) and the presence of monosaccharides, this material could easily become spoilage (Zheng et al., 2011; Ziemiński and Kowalska-Wentel, 2015). For this reason, this material should be dried and pelletized before application. However, these operations consume even 40% of the overall energy costs of sugar factories (Hutnan et al., 2000; Brooks et al., 2008; Borowski et al., 2016). Additionally, its seasonal production is another disadvantage of application this by-product. During the season, fresh sugar beet pulp is accumulated in much higher quantities than those, which can be immediately utilized (Ziemiński and Kowalska-Wentel, 2015). Anaerobic digestion could be an attractive option for the SBP application. However, mono-digestion of this waste is quite problematic due to the low pH and unfavorable C/N ratio as well as relatively low methane yields. Moreover, due to the high content of sucrose in sugar beet and high amount of pectin in its cell wall, problems with the stable foam formation in the digesters could occur, especially when they are highly loaded and operated at mesophilic temperatures (Moeller et al., 2015). In order to improve the nutrient balance and achieve stable process conditions as well as satisfactory methane yields, sugar beet pulp might be co-digested with other organic wastes including cow (Aboudi et al., 2016) and poultry manure (Borowski et al., 2016), pig slurry (Boldrin et al., 2016) and potato processing by-products (Kryvoruchko et al., 2009). Additionally, there are numerous studies in the literature regarding the anaerobic digestion of sugar beet pulp with sewage sludge. In the study presented by Montañés et al. (2015), the biochemical methane potential (BMP) tests were conducted to investigate the effect of pH control on the co-digestion of sewage sludge (SS) and sugar beet pulp lixiviation (SBPL) at mesophilic range (35°C). In this research, the inhibition of methane generation occurred, as measured by the BMP test in the absence of pH adjustment. In another study performed by Montañés et al. (2015) the effect of temperature on the anaerobic co-digestion of sewage sludge (SS) and sugar beet pulp lixiviation (SBPL) was evaluated. Mesophilic and thermophilic batch assays of five different SS/SBPL ratios were done. In this test, biodegradation was limited under the thermophilic conditions, because of the increased VFA concentration. In both cases, the authors indicated that the anaerobic co-digestion of sewage sludge and lixiviation of sugar beet pulp is a promising application that improves the biogas productivity and the organic matter removal.

3. Summary and Conclusions

- Anaerobic co-digestion process is a sustainable alternative for energy recovery as well as management of organic wastes. In comparison to mono-digestion, this process presents several advantages such as increasing biogas production, process stabilization, improvement of the nutrient balance and C/N ratio and dilution of potential toxic compounds.
- One of the most important factors that should be considered to achieve effective co-digestion is adequate substrate selection. The preferable co-substrate properties include high content of readily biodegradable organic fraction, high buffering capacity balanced C:N ratio, low nitrogen concentration, high biogas potential of co-substrates as well as accessibility on the local market.
- Food industry generates a large amount of waste, which could be a potential source of removable energy using anaerobic co-digestion process. Due to the composition and availability of cheese whey, brewer's spent grain, slaughterhouse waste as well as sugar beet pulp, they constitute potential co-substrates in anaerobic co-digestion process.

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Study of vermicompost properties of sewage sludge and biodegradable waste using zeolites

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Abstract

The European Commission published on 2 December 2015 an Action Plan for Circular Economy together with revised legislative proposals on waste. This Circular Economy package is an effort to rethink economic model, with measures covering the full lifecycle of products. The transition from the linear approach: "production – use – waste utilization in the next production cycle" – aims to reduce the consumption of raw materials and reduce the amount of deposited waste. They become valuable resources, and reuse and recycling, including organic recycling, are becoming a priority.

This strategy perfectly fits the vermicomposting process, which is a natural process that take place in nature. Conducting it under controlled conditions allows to determine the vermicomposting as a technological process. The element of the technological process is the selection of the appropriate material for processing by the vermiculture. The research shows that in this process it is possible to process the mass of biodegradable waste (also sewage sludge) and use it as an organic fertilizer. The main goal of the vermicomposting process is the ability to produce organic fertilizers or plant growth aids for biodegradable waste.

The development of a rational composition of the biodegradable waste composting substrates in order to process them and produce material for use as an organic fertilizer, perfectly realize the main aims of circular economy. For the highest quality organic fertilizer production, were used substrate mixtures with zeolites. Zeolites were used in the substrate mixtures to stabilize the process and eliminate some harmful or dangerous substances.

The aim of this article is to present the changes in the characteristics of recycled mixtures in the vermicomposting process and the zeolites influence for vermicomposting process. As well as to present the possibility of biodegradable waste organic recycling in order to fulfill the requirements of the circulation system as much as possible.

Keywords

biodegradable waste, sewage sludge, vermicomposting process, organic recycling, vermicompost, zeolites, organic fertilizer

1. Introduction

Biodegradable wastes are characteristic group of waste. They have a varied composition, so there are problems with choosing the optimum processing methods. The main methods of recovery and disposal this group of waste (apart from storage) are biological methods: composting, fermentation and biological – mechanical processing.

The issue concerning on the processing of this group of waste is particularly relevant due to the recent tightening of legislation in this area. Approximately 0.5% increase in the share of biodegradable fractions (including paper and cardboard, clothing and textiles from natural materials (50%), green waste, kitchen and garden waste, wood (50%), multi-material waste (40%), fine fraction <10 mm (30%) in the general stream of municipal waste. It is certainly one of the factors that submitting to take optimization actions in the processing of this waste group. Another group of biodegradable waste, which has been subject to drastic legal changes in recent years, is development of sewage sludge. So far, the main method of developing municipal sewage sludge was their disposal by landfill at municipal landfills. This is the simplest method, and seems to be economically justified, since the storage of sewage sludge was the cheapest available method. The changing legal regulations force changes in the current management of sewage sludge, because of the prohibition on depositing them in landfills.

Whereas in larger cities, municipal biodegradable waste and sewage sludge can be subjected to a number of processing methods, such as recovery in composting or biogas plants, thermally transforming in incineration plants or co-incineration plants (eg. in cement plants), these processes are often not economically feasible in small towns.

The vermicomposting process is a perfect complement to the possibility of introducing effective organic recycling for such municipalities. The use of vermicompost technology, will help reduce the amount of waste stored and their use, reduce emissions to the environment, and lead to sustainable organic waste management. Previous studies have shown that it is possible to implement a rational vermicompost process in sewage treatment plants or small municipalities and such a waste treatment model takes into account the condition of maximum sludge management and the use of other biodegradable waste as a complement to the process. With a high level of reduction of the biodegradable waste stream in the vermicompost process (about 65% by weight), the proposed process may be a support to the tasks of local government units to achieve the required levels of reduction of biodegradable waste. The added benefit of the rationalization process is the production of high quality compost, certified organic fertilizer and relatively high social acceptance of the proposed solutions.

2. Legal regulations

According to the Act of 14 December 2012 *On waste* (Dz. U. 2013 pos. 21) communal sewage sludge – sludge from fermentation chambers and other installations for the treatment of municipal waste water and other effluents of a composition similar to that of municipal waste water, classified in group 19 as waste with code 19 08 05 – stabilized municipal settlements sewage (Dz. U. 2014 pos. 1923).

So far, the main method of developing municipal sewage sludge was their disposal by landfill at municipal landfills. This is the simplest method, and seems to be economically justified, since the storage of sewage sludge has been the cheapest method to date, among the available methods.

Depending on the form in which they are transferred for disposal, municipal sewage sludge can be thermally transformed into incineration plants or co-incineration plants (e.g. in cement plants), recovered in composting or biogas plants or directly used on soil for soil improvement, after their stabilization. In addition, changing legal regulations force changes in the current management of sewage sludge.

KPGO 2014 indicates that the reduction of sludge storage should be sought by increasing the amount of municipal sewage sludge processed prior to introduction into the environment and of thermally transformed sludge and maximizing the utilization rate of biogenic substances contained in sediments while meeting all sanitary and chemical safety requirements. The natural use of sewage sludge is regulated by the Waste Act of 14 December 2012 (Dz. U. 2013 pos. 21) (Chapter VII – Specific rules for the management of certain types of waste, Chapter 4 – Municipal sewage sludge). According to the law, it recommends the use of sewage sludge through their use in agriculture, plant cultivation, or land reclamation. The legislator imposes the form of sludge transfer and indicates procedures related to the use of sewage sludge, both on the producer side and on the host side. Subsequently, the ordinance of the Minister of the Environment of 6 February 2015 on municipal sewage sludge indicates the exact conditions to be met for the application of soil sediments (eg permissible content of heavy metals, *Salmonella*, number of live parasite eggs intestinal *Ascaris sp.*, *Trichuris sp.*, *Toxocara sp.*). This limitation also involves the ban on depositing sewage sludge in landfill sites. The main problem with the implementation of alternative waste disposal methods is the economic aspect. Hence cheaper methods are sought, both at the stage of the installation and at the stage of operation. The vermicomposting process seems to be the best for this because of the low cost of building compost piles and the low cost of operating an active vermicompost.

3. *Vermicomposting as a biodegradable waste processin technology*

In Poland, the vermicomposting process was popularized in the mid-1990s. The first sewage treatment plant that undertook operations in this direction was a wastewater treatment plant in Pyrzyce. The vermicompost that was produced from sewage sludge was used for agricultural purposes (Bożym, 2012). Quite extensive research was also conducted at the Kluczbork Sewage Treatment Plant. These studies concerned the biological processing of biodegradable fractions of biodegradable waste and sewage sludge (Bożym, 2012; Bożym, 2004). These studies, however, were completed in 2014, with the discontinuation of the vermicompost process. Research on the vermicomposting process was carried out, but according to available literature, the research was mainly focused on the possibility of utilizing the vermicomposting process to manage organic waste in households (Kostecka et al., 2014). Research on the vermicomposting process in Poland, on a fairly large scale, was also carried out on the vermiculture of the Municipal Sewage Treatment Plant in Zambrow (Boruszko, 2011; Boruszko, 2010a; Boruszko, 2010b; Boruszko et al., 2005).

Despite the fact that the composting using earthworms is a process commonly known as coprolites (dung worms) is a product benefit of improving the properties of the soil, due to large water, and the contents of ingredients that improve plant growth, its application is reduced mainly for home composting and Pilot installation. This is related, inter alia, to the need to create appropriate living conditions for these organisms, to maintain adequate stability of the composting parameters and to protect the culture by external factors. Maintaining process stability is primarily a regular supply of adequate amounts of food and protection against sudden and significant temperature and humidity changes. Protecting against temperature and humidity changes is achieved through covers such as straw or nonwoven fabric, but even the thickness of the protective layer plays an enormous role. The protective layer protects the vermiculture against overheating and drying by evaporation on hot days, while on cool days, it protects by sudden drop in temperature but must also allow access to fresh air. They also play an important role in protecting against the interference of external organisms, such as moles, for which earthworms feed. Properly guided process vermicomposting on a technical scale, it requires constant control and verification of the parameters by the operator but on the basis of long-term observations indicate the optimal ranges of living earthworms and vermicomposting: temperature: 22 to 28ºC, the relative humidity of the substrate: 70 to 80%, pH of the medium (pH 6.4 to 8, C: N ratio: 25: 1, appropriate crumbling, oxygenation and salinity of the medium (0.5%), as well as appropriate protein content (Kalisz et al., 2000). Earthworms are also susceptible to ammonium nitrate, which is a hazard if present in quantities greater than 0.5 mg/g substrate. Unfavorable conditions, and therefore the parameter values below or above these ranges decrease the physiological activity of earthworms, consequently, they lead to a weakening of the intensity of the process waste, and in extreme cases can cause death vermicultures (Kalisz et al., 2000). Based on previously conducted studies, apart from the characterized abiotic extrapulmonary factors, extrapulmonary and intraoptotic biotensive factors play an important role (Kalembasa, 1995).

The study referred to by the literature indicates that the vermicomposting process has been carried out by different *Lumbricidae* species such as *Lumbricus rubellus*, *Eudrilus eugeniae*, *Perionyx excavatus*, *Dendrobaena veneta* and *Eisenia fetida* (Kostecka, 1994; Edwards, 1995; Kalembasa, 2000). Research conducted to rationalize the vermicomposting process on prisms in Ociece was conducted by species of *Dendrobaena veneta* and *Eisenia fetida*.

4. Technical possibilities of using zeolites in the vermicomposting process

According to the definition (Mining Lexicon, 1989), zeolites are hydrated aluminosilicates of alkaline elements (Na, K, Li), alkaline earths (Ca, Mg, rarely Ba and Sr) or other mono- or polyvalent metals (Anielak, 2010; Szostak, 1998; Ciciszwili et al. 1990; Auerbach, 2003; Sarbak, 2009; Weitkamp, 2000). They belong to a group of silicates - groups of hydrated aluminosilicate skeletons. According to the classical definition of Smith and Breck zeolites have interchangeable cations, as well as zeolitic water in the crystalline lattice. According to the accepted definition of structure not meeting the above requirements cannot be classified as zeolites (Szostak, 1998; Ciciszwili et al., 1990; Lee, 1997). The basic classification that can be made for zeolites is their origin. Zeolites can occur as natural, unevenly occurring minerals in nature, as well as synthetic materials, produced on an industrial scale through synthesis. Among the most common zeolites in nature are clinoptilolite, mordenite, analgesic and chabasite, but they are often found in the form of single clusters (Colella et al., 2005; Yusupov et al., 2000). Zeolite used in the vermicomposting process is a natural zeolitic clinoptilolite characterized by specific chemical properties: ion exchange, adsorption and desorption of water, gas adsorption, sorption of petroleum impurities and catalytic properties (Petrus et al., 2000). Due to its porous structure, it is possible to adsorb through its pores, and it is characterized by ion exchange (high affinity for ammonium ions). In addition, it does not disturb biology of the vermicompost process. It is also indifferent to participating microorganisms in vermicomposting and later, on organisms living in the soil. Little by little, microbiological life can develop on its surface, and thanks to its structure, the zeolite retains water-insoluble nutrients that are gradually released and absorbed by plants. Zeolite also has a high sorption capacity due to its high ionic capacity (mainly ammonium ions), which prevents nitrogen deficiency. Zeolite increases the sorption capacity of the soil and contributes to reducing the amount of heavy metals that can be absorbed by the plants. The introduction of zeolite into the vermicomposting process results in increased water retention, nitrogen retention and other microelements, enrichment in microelements and minerals, increased ion exchange of the soil and its infiltration and aeration. Ion exchange additionally has a neutralizing effect on the aroma produced during the process (Colella et al., 2005; Yusupov et al., 2000; Anielak, 2006; Wójcik W. et al., 2014).

5. *Application of zeolites in the vermicomposting process*

5.1. Material and Methods

The scope of laboratory tests was planned in such a way as to be able to observe the changes taking place during the vermicomposting process as well as to determine the fertilizer properties after the process. The use of zeolites in the vermicomposting process required the determination of nitrogen, phosphorus, potassium, dry matter, organic dry matter. During the course of the process, temperature, pH and humidity and weight loss in beds where the vermicomposting process was also controlled. During the process representative samples were taken from the entire working volume of vermiculture, according to the bed and sampling scheme and in accordance with standard PN-R-04006: 2000 - Organic fertilizers - Collection and preparation of manure and compost samples (Wójcik et al, 2015).

5.2. Characteristic of the research object

Research on the vermicomposting process was carried out on wastes of code 20 01 08 - biodegradable kitchen waste and 02 01 03 - waste vegetable mass (straw) in vermicomposted beds (length x width x height) 2.0 m x 2.0 m x 1.0 m. The vermicomposting process was carried out using two species of Lumbricidae soil dendrites: *Dendrobaena veneta* and *Eisenia foetida*. During the vermicompost process, natural zeolite, clinoptilolite was used. The applied zeolite was characterized by a homogeneous grain size of 0.3–0.5 cm. Used zeolite was supplied by BIODRAIN, type BIOZEO.R.01.3,0–5,0 mm.

5.3. Results and Discussion

Below are the results of the research carried out in the technological cycle. The following parameters were tested: moisture content, pH, dry mass [g/g d.m.], phosphorus content [mass % as P_2O_5], nitrogen content [mass %], potassium content [% K_2O].

Tables 1–6 in green indicate cells in which are values, consistent with the fertilizer and fertilization law (Dz. U. 2007 No. 147, item. 1 033), allowing to determine the product obtained as an organic fertilizer and the literature of the subject (Kalisz et al., 2000).

Table 1. Results of changes in nitrogen content [mass %] during the vermicomposting process (own study)

	18.08.2016	**19.08.2016**	**22.08.2016**	**24.08.2016**	**26.08.2016**
test "0"	0.5046	0.5755	0.4845	0.5644	0.5744
10% zeolite	0.5124	0.5564	0.4561	0.4964	0.4323
20% zeolite	0.4997	0.5694	0.4496	0.5123	0.4694
30% zeolite	0.5014	0.5546	0.4934	0.5139	0.4964

Table 2. Results of changes in potassium content [mass % calculated as K_2O] during the vermicomposting process (source: own study)

	18.08.2016	19.08.2016	22.08.2016	24.08.2016	26.08.2016
test "0"	0.2527	0.1940	0.1999	0.2471	0.2587
10% zeolite	0.2496	0.1824	0.1799	0.2103	0.2236
20% zeolite	0.2549	0.1796	0.1654	0.1985	0.1987
30% zeolite	0.2513	0.1845	0.1843	0.1934	0.1846

Table 3. Results of changes in phosphorus content [mass % calculated as P_2O_5] during the vermicomposting process (source: own study)

	18.08.2016	**19.08.2016**	**22.08.2016**	**24.08.2016**	**26.08.2016**
test "0"	0.210	0.116	0.129	0.137	0.220
10% zeolite	0.199	0.126	0.136	0.154	0.204
20% zeolite	0.210	0.097	0.115	0.129	0.175
30% zeolite	0.208	0.106	0.154	0.144	0.165

Table 4. Results of humidity changes [mass %] during the vermicomposting process (source: own study)

	18.08.2016	**19.08.2016**	**22.08.2016**	**24.08.2016**	**26.08.2016**
test "0"	77.45	78.15	79.31	80.14	78.15
10% zeolite	78.15	72.23	74.03	74.62	73.94
20% zeolite	79.15	68.48	69.87	70.15	71.56
30% zeolite	78.04	65.21	67.17	67.54	68.54

Table 5. Results of pH change tests during the vermicomposting process (source: own study)

	18.08.2016	**19.08.2016**	**22.08.2016**	**24.08.2016**	**26.08.2016**
test "0"	7.13	7.25	7.19	7.21	7.17
10% zeolite	7.35	7.45	7.48	7.59	7.61
20% zeolite	7.28	7.59	7.89	7.78	7.81
30% zeolite	7.34	7.49	7.78	7.94	7.94

Table 6. Results of organic matter change tests $[g/g_{d.m.}]$ during the vermicomposting process (source: own study)

	18.08.2016	**19.08.2016**	**22.08.2016**	**24.08.2016**	**26.08.2016**
test "0"	0.7383	0.8062	0.4542	0.3448	0.5369
10% zeolite	0.6998	0.7124	0.4189	0.3985	0.3423
20% zeolite	0.6536	0.6945	0.2974	0.2654	0.2646
30% zeolite	0.6124	0.6014	0.2749	0.2514	0.2278

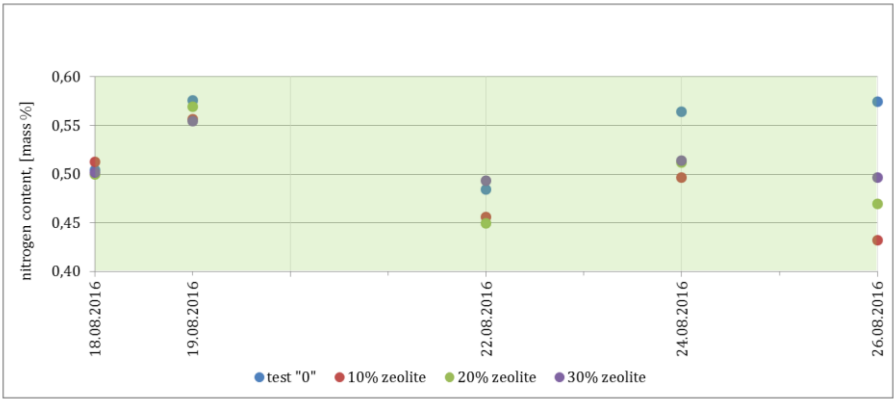

Fig. 1. Change of nitrogen content [mass %] during the vermicomposting process, depending on the amount of zeolite in the test sample (source: own study)

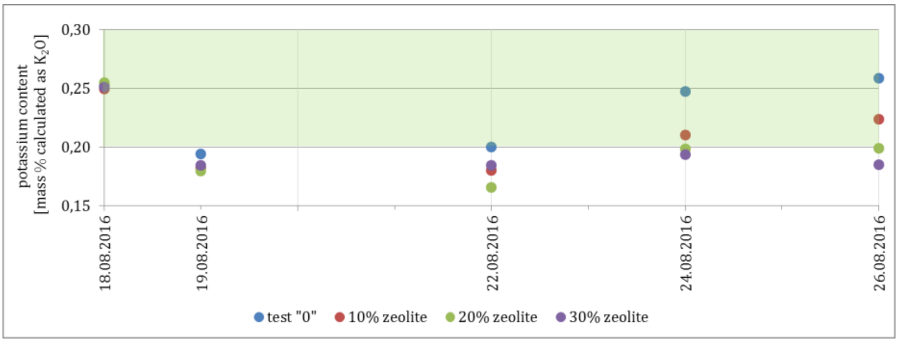

Fig. 2. Change of potassium content [mass % calculated as K_2O] in the vermicomposting process, depending on the amount of zeolite in the sample (source: own study)

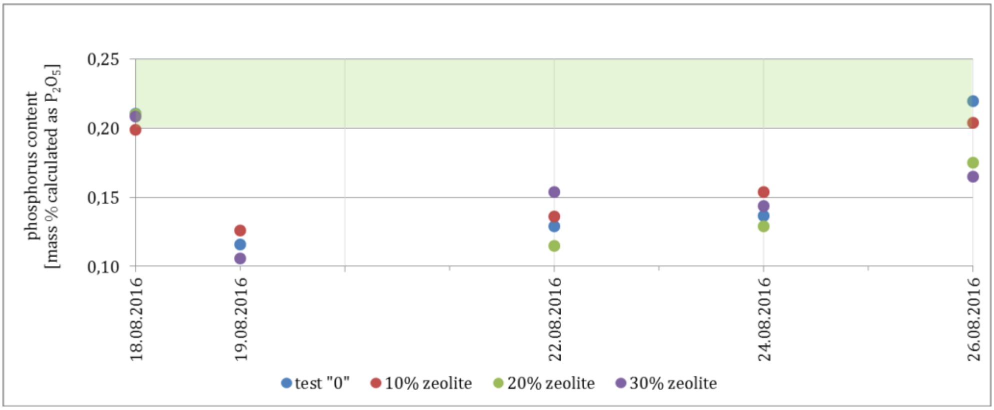

Fig. 3. Change of phosphorus content [mass % calculated as P_2O_5] in the vermicomposting process, depending on the amount of zeolite in the sample (source: own study)

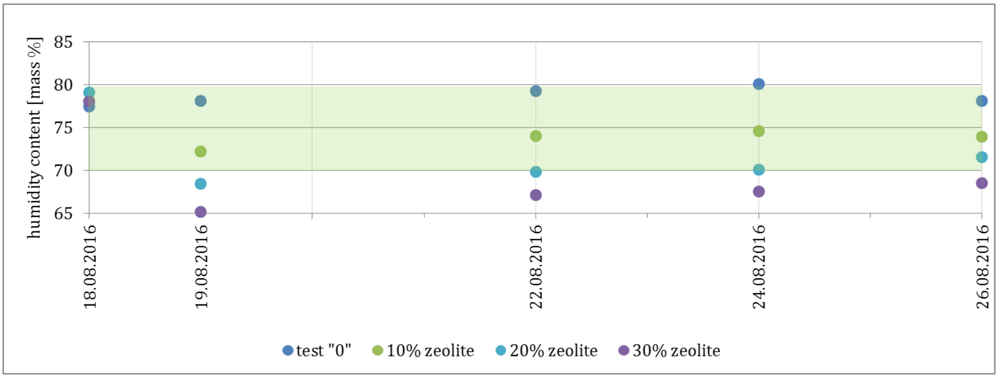

Fig. 4. Change of humidity content [mass %] in the vermicomposting process, depending on the amount of zeolite in the sample (source: own study)

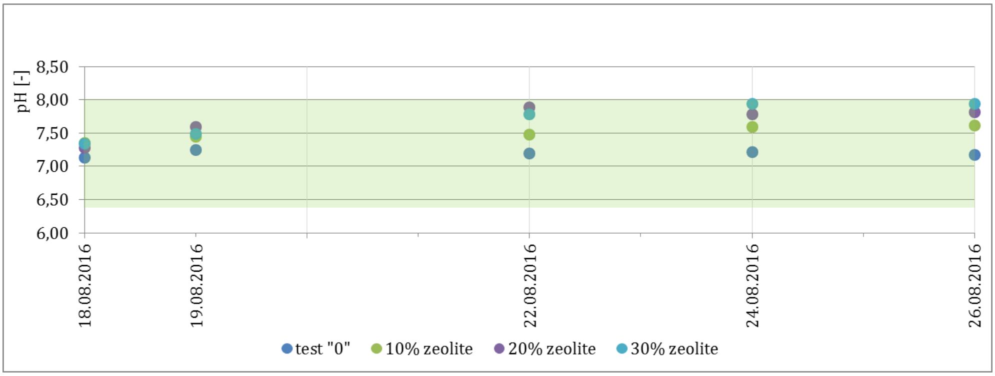

Fig. 5. Change of pH change tests in the vermicomposting process, depending on the amount of zeolite in the sample (source: own study)

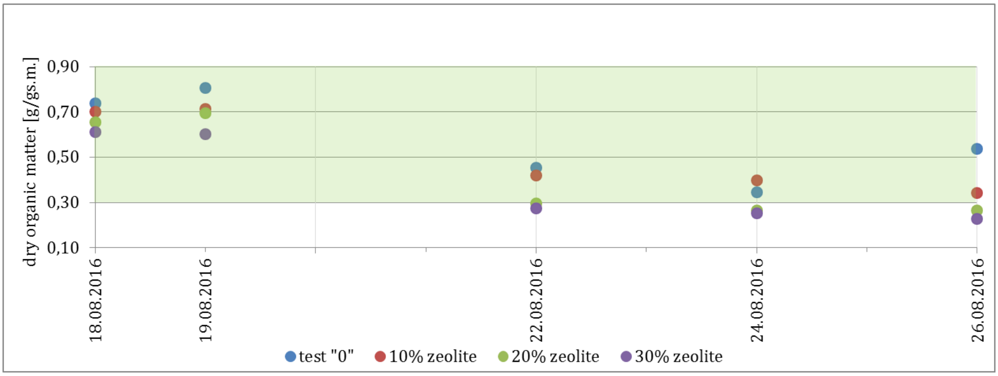

Fig. 6. Change of dry organic matter [g/$g_{d.m.}$] in the vermicomposting process, depending on the amount of zeolite in the sample (source: own study)

5. *Summary and Conclusions*

The technological cycle carried out allowed to evaluate the influence of zeolite (in different amounts) on the vermicomposting of sewage sludge and the influence of zeolite on the obtained vermicompost. On this basis it is possible to assess the effectiveness of zeolites in the vermicompost process. The optimum zeolite dose was chosen based on the analysis in Table 7. The green color table shows which of the parameters were met according to the test sample and the zeolite dose. As can be seen from the table, the most favorable test results and process stability are obtained using 10% of zeolite.

Table 7. Evaluation of compost parameters for its usefulness as an organic fertilizer, depending on the zeolite dose (source: own study)

	nitrogen	**phosphorus**	**potassium**	**dry organic matter**	**pH**	**humidity**
test "0"	YES	YES	YES	YES	YES	YES
10% zeolite	YES	YES	YES	YES	YES	YES
20% zeolite	YES	NO	NO	NO	YES	YES
30% zeolite	YES	NO	NO	NO	YES	NO

Studies have shown that it is possible to use zeolites for vermicomposting to produce a rational batch of feedstock, as well as to process the maximum possible mass of biodegradable waste and to obtain material that can be used as an organic fertilizer. Apart from the presented results, it has been observed that one of the positive effects of the process was the reduction of the stream of biodegradable waste. The results obtained and the experience gained show that it is possible to use zeolite vermicompost technology as organic recycling in small settling units to reduce waste streams. The use of zeolites can in turn allow the removal of hazardous substances, including heavy metals, from waste (mainly sewage sludge).

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Improving the energy balance in wastewater treatment plants by optimization of aeration control and application of new technologies

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Abstract

The methods to improve the energy balance of a wastewater treatment plant (WWTP) by optimization of aeration process control and application of innovative nitrogen removal technologies were overviewed in the study. The control of aeration based on the ABAC (Ammonia-Based Aeration Control) system allows not only for significant savings in electricity consumption, but it can also increase the efficiency of the denitrification process. In addition to obtaining a shortened nitrification path, the use of the AVN (Ammonia vs. Nitrate/Nitrite) system can provide even more efficient nitrogen removal when controlling the process of conventional nitrification-denitrification. It is expected that the future of the control systems in WWTPs belongs to smart process control systems cooperating with simulation models. The innovative technologies for nitrogen removal are based on the shortcut nitrogen removal pathways. Compared to the conventional nitrification-denitrification, the advantage of the deammonification (partial nitritation/anammox) process is the reduction of both electrical energy demand for aeration and the production of the sludge, as well as no need for the organic carbon source. The significance of pH, temperature, dissolved oxygen concentration, as well as the aeration patterns were emphasized and investigated in details by numerous researchers. The implementation of the innovative nitrogen removal technologies allows to separate organic and nitrogen waste streams and thus maximize energy capture. A study on the implementation of both the chemically enhanced primary treatment and the deammonification process in the sidestream treatment line revealed the potential for an increased biogas production in the anaerobic digester and reduction in the electric energy demand for aeration, while still maintaining the required total nitrogen effluent standard. The proposed upgrades can lead a WWTP from the energy deficit to the energy neutrality. A few WWTPs have already achieved almost 100% (or higher) electricity self-sufficiency using combined approaches and proved the usability of the innovative nitrogen removal technologies.

Keywords

aeration control; deammonification; energy neutrality; nitrogen removal

1. Introduction

Energy consumption in municipal wastewater treatment plants (WWTPs) constitutes a significant share in a municipality total energy balance. The unit electricity consumption for treating a cubic meter of wastewater is 0.1–0.7 kWh in facilities employing different treatment technologies and process control (GWRC, 2009). Electricity intensity in biological nitrogen removal processes is approximately 0.5 kWh/m^3 in typical practice and approximately 0.3 kWh/m^3 when best practice treatment processes are applied (Wang et al., 2016). A number of factors, including implementation of energy-intensive processes to meet increasingly stringent regulations, rising energy costs, policies aimed at reducing energy consumption and carbon footprint, enforces measures to improve the energy balance in WWTPs. Energy neutral or even energy positive wastewater treatment is a new paradigm widely spread in the world.

The actions undertaken towards energy neutrality should be integrated and directed not only to an even greater reduction of energy consumption, but also to energy recovery from the internal resources. It is estimated that the total amount of energy contained in municipal wastewater is about 23 W/inhabitant in organic carbon as well as 6.0 and 0.8 W/inhabitant in ammonium nitrogen and phosphates, respectively (Gao et al., 2014). Jenicek et al. (2012) found that the self-sufficiency depends on the optimization of the total energy consumption of the plant and an increase in the specific biogas production from sewage sludge. The separation of organic matter from wastewater influent can enhance biogas production, while the separated nitrogen stream can be treated with the energy efficient deammonification processes.

In the activated sludge (AS) systems oxygen is consumed in the processes of nitrogen removal (nitrification) and carbon removal (aerobic oxidation). From the point of view of the operation of WWTPs, aeration is the most energy-intensive process that can account for up to 50% of the total energy consumption (Gori et al., 2011; Zhou et al., 2013; Gao et al., 2014). Thus, a majority of the energy-saving measures aims at upgrading the aeration system with very effective control strategies or implementation innovative treatment processes. The new promising methods for nitrogen removal can replace or support the traditional biological nutrient removal AS system. Ammonia-rich anaerobic digester liquors can be treated with the very economic autotrophic nitritation/anammox process requiring half of the aeration energy and no organic carbon source compared to the conventional nitrification-denitrification (Siegrist et al., 2008).

The aim of the study is to present an overview of the options to improve the energy balance of a WWTP by both optimization of the aeration process control and application of innovative nitrogen removal technologies. Several results of research work have been selected to show a contribution to the development in the field of the advanced processes shifting WWTPs towards energy neutrality.

2. *Optimization of aeration control in a bioreactor*

The vast majority of currently used aeration systems is controlled based on the measurement of dissolved oxygen (DO) concentration in a nitrification zone. Modulating the air flow to maintain a constant value of DO concentration is usually an effective measure, ensuring the expected nitrogen removal efficiency. On the other hand, this solution is not very energy efficient, especially in the case of high variability of pollutant load in wastewater influent. A new approach aims at more advanced aeration control strategies based on on-line measurements of nitrogen compounds (Amand et al., 2014). Such opportunities have emerged with the introduction on the market more reliable ammonium (NH_4^+) and nitrate/nitrite (NO_x) sensors. In the ABAC (Ammonia-Based Aeration Control) system, the control of DO concentration changes is based on measurements of NH_4^+ concentration, while the AVN system (Ammonia vs. Nitrate/Nitrite) is based on measurements of both NH_4^+ and NO_x concentrations. The location of NH_4^+ and NO_x sensors in the ABAC and AVN systems is shown in Fig. 1.

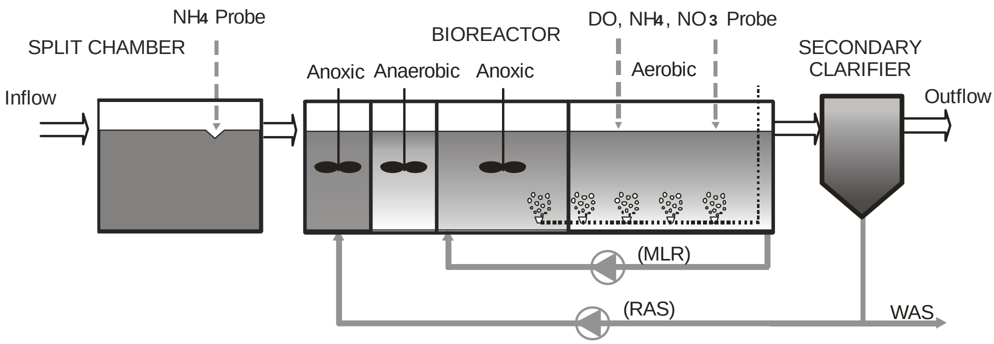

Fig. 1. Location of NH_4^+ and NO_x sensors in the advanced aeration control systems (Amand et al., 2014; Regmi et al., 2014; Rieger et al., 2014); MLR – mixed liquor recirculation, RAS – returned activated sludge, WAS – waste activated sludge

In the ABAC system, there are two types of control strategies depending on the location of NH_4^+ sensors in the biological reactor:

- feedback control for the measurements taken in the outflow from the nitrification compartment,
- feedforward control for the measurements taken in the inflow to the nitrification compartment.

The control of aeration based on the ABAC system allows not only for significant savings in electricity consumption (about 10–20%), but it can also increase the efficiency of the denitrification process due to reduction of alkalinity and organic carbon consumption (Rieger et al., 2014). The feedforward control is more complex, but ensures the required wastewater quality at lower energy consumption.

The AVN system was originally developed for systems with a shortened nitrification path in order to remove from the system nitrifying bacteria of the second stage (oxidizing nitrites to nitrates), the so-called NOB bacteria (Regmi et al., 2014). In addition to obtaining a shortened nitrification path, the use of the AVN system can provide even more efficient nitrogen removal when controlling the process of conventional nitrification-denitrification. The adjustment of the nitrate load to be denitrified is performed by setting the concentrations or proportions of NH_4^+ and NO_x in the outflow.

The future of the control systems in WWTPs belongs to smart process control systems. A core of the systems will be the simulation model implemented in the computer program together with the appropriate algorithm for biochemical processes control depending on the current on-line measurements at the designated locations in the bioreactor. Currently, computer models of wastewater treatment processes are used as an auxiliary tool to forecast various technological variants and enable searching for optimal solutions, i.a. in the field of aeration costs.

3. *Traditional vs. innovative technologies of nitrogen removal*

The traditional methods of nitrogen removal from wastewater, using the conventional processes of nitrification and denitrification, are cost-intensive because of high oxygen demand for oxidation of ammonium nitrogen as well as because of the need for supplying organic carbon sources for the reduction of nitrate nitrogen. An additional issue, emerging in the case of dosing an external carbon source (such as methanol, ethanol, acetate or fusel oil) is the increase in the excessive sludge production. Moreover, a kind of the carbon source is essential in achieving the complete degradation of both nitrogen and phosphorus (Drewnowski and Makinia, 2011). AS mathematical models have been proven to be useful tools for evaluating and optimizing the effect of external carbon sources for nitrate/nitrite removal. However, more reliable model predictions can be found when a newly defined readily biodegradable substrate is considered under anoxic conditions (Swinarski et al., 2012) .

In recent years, innovative cost-effective methods of nitrogen removal from wastewater have emerged. The methods are based on partial nitritation and anammox (anaerobic ammonium oxidation) process. The anammox process, although being common in the natural environment (e.g. in oceans), has not been discovered until 1990s. The main idea of the nitrite shunt is the termination of the nitrification on the NO_2-N stage via the inhibition of the growth of bacteria that oxidize nitrite to nitrate

– the nitrite oxidizing bacteria (NOB). It results in the reduction of cost for aeration in the process of nitrification and the addition of organic carbon in the process of denitrification. The oxygen demand and organic carbon demand can be reduced by ~25% and ~40%, respectively. A comparison between the conventional and shortcut nitrogen removal is shown in Fig. 2.

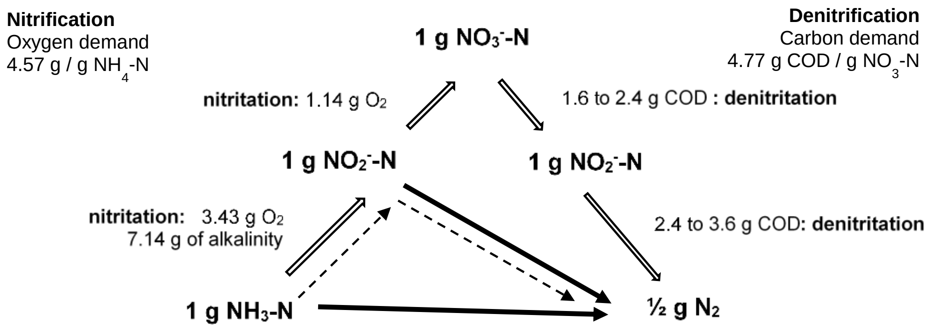

Fig. 2. Comparison between the conventional and shortcut nitrogen removal

In the anammox process, the removal of ammonium nitrogen is carried out by autotrophic microorganisms *Planctomycetales*. The anammox bacteria convert the ammonium nitrogen and the NO_2-N (in the ratio of 1:1.3) to molecular nitrogen (~90%) and nitrates (~10%) without the need of organic carbon addition. The process is beneficial for nitrogen removal from wastewater with low biological oxygen demand to nitrogen ratio, which commonly applies to reject water from digested sludge (the sidestream treatment line).

The combination of the processes of partial nitrification and anammox is the deammonification, which can be implemented as one-step process in a sequencing batch reactor (SBR) (e.g. DEMON process) or in hybrid systems (e.g. AnitAmox system). Compared to the processes using conventional nitrification-denitrification, deammonification has significant advantages including: the reduction in the electrical energy demand for aeration (~60%), the reduction in the production of the sludge (~90%), the lack of the organic carbon demand, as well as the reduction in CO_2 emission to the atmosphere (more than 90%) (WERF, 2014).

The highest number of the facilities using the deammonification process for sidestream treatment is located in Netherlands, Germany, Switzerland and Belgium. To date, the implementation of the anammox process in the mainstream bioreactor has been a subject on an intensive research work (Wett et al., 2013; Al-Omari et al., 2013; Regmi et al., 2013). The results identified the following issues: removing the NOB bacteria from the system, growing and keeping the anammox bacteria in the system, the competition between the anammox and the heterotrophic bacteria in the presence of organic carbon, as well as the lower process temperature in comparison with the sidestream treatment (Mozo et al., 2016). The aeration pattern is a key factor in the first step of the deammonification process, i.e. partial nitritation, therefore the subject have been investigated by several authors (e.g. Lackner et al., 2014). Application of an appropriately selected aeration strategy can reduce energy consumption. Moreover, a good aeration strategy could be successful in inhibiting the activity of NOB organisms compared with ammonia oxidizing bacteria (AOB) to avoid accumulation of NO_3-N in the effluent (Jardin and Hennerkes, 2012).

Sobotka et al. (2016) carried out a series of batch tests, as well as long-term experiments in continuously operated SBRs in order to recognize pH and temperature influence on the anammox or/either deammonification systems performance. Technological measurements were supported by mathematical modeling and microbial analysis of total bacterial communities. In terms of pH, the inhibitory effect on nitrogen removal rates was associated with nitrous acid and free ammonia metabolic mechanisms. The estimated optimal pH were 7.4 in case of AOB and 7.6 for the anammox bacteria. The experimental data as well the model predictions allowed to estimate specific anammox activities (SAAs) in the range of 0.83 to 0.90 g N/(g VSS·d) (Lu et al., 2017). The study on the anammox system performance revealed that the temperature range is between 10 and 55°C. Application of both extreme temperatures practically inhibits the nitrogen removal process and may lead to irreversible loss of biomass integrity (Fig. 3).

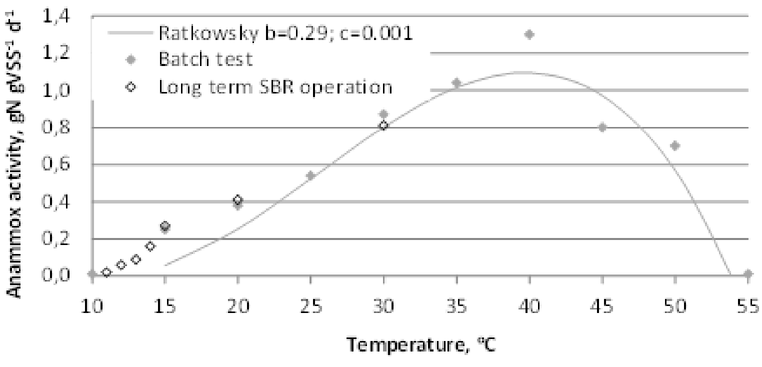

Fig. 3. Effects of temperature on the anammox activity during batch tests and long-term SBR operation (Sobotka et al., 2016)

While the temperature of 40°C ensured the maximal nitrogen removal rate (1.3 g N/(g VSS·d), the effective performance of the system was obtained at 15°C, however some adaptation period had to be applied (Sobotka et al., 2016).

The experiments carried out by Sobotka et al. (2015) on the DO set point as well the aeration mode showed a significance of this parameters in the deammonification process. The continuous aeration at low DO set point (0.4 g O_2/m^3) ensured the highest values of the ammonia uptake rate (AUR), however the overall processes suffered from the high rates of NO_3-N production. It means that such conditions do not suppress the NOB activity effectively. Intermittent aeration seems to be more perspective strategy for the successful maintaining of the deammonification systems. To ensure the best performance and decrease the risk of the NO_3-N accumulation in such systems, the time duration of the aeration/mixing phases should be reduced. Microbial studies over bacterial community composition during long time biomass adaptation to the anammox/deammonification processes revealed substantial reorganization of its structure. Basically, abundance reduction of typical heterotrophs with parallel autotrophic bacteria growth increase was observed. The enhanced SAAs was accompanied with *Planctomyctes* related bacteria expansion (Sobotka et al., 2017).

Al-Hazmi et al. (2017) extended the tests to examine a short-term and long-term impact of DO concentration and the selected aeration strategies on enhancing the deammonification process. A special attention was paid to testing and comparing the process performance under the intermittent aeration vs. continuous aeration mode. In the batch tests, various DO concentrations were set at 0.3, 0.5 and 1.0 g O_2/m^3. The highest AUR along with the lowest nitrate accumulation were obtained for the short intermittent aeration mode (8 min on/22 min off), at the DO set point = 0.5 g O_2/m^3. Under the continuous aeration mode, the highest AUR along with the highest nitrate accumulation were observed at the DO set point = 0.3 g O_2/m^3. Sample results of the test are presented in Fig. 4. The long-term effect of the intermittent aeration on the deammonification process performance was investigated in an SBR operated at different DO concentrations (0.3, 0.5 and 1 g O_2/m^3, respectively). The NH_4-N load was in the range of 120–240 g N/m^3. At the concentration of 120 g N/m^3, the maximum NH_4-N removal efficiency of 76–89% and the minimum NO_3-N production of 24–25% were observed at the short intermittent aeration mode (10 min on/20 min off), at the DO set point = 0.3 g O_2/m^3. The maximum NO_3-N accumulation of 44–80% was achieved at the intermittent aeration mode (20 min on/20 min off) and at the DO set point = 1.0 g O_2/m^3. At the concentration of 240 g N/m^3, the maximum NH_4-N removal efficiency of 84–89% and the minimum NO_3-N production of 21–27% were obtained for the short intermittent aeration mode (5 min on/15 min off), at the DO set point = 0.5 g O_2/m^3. The maximum NO_3-N production of 25–34% and the minimum NH_4-N removal of 65–79% were observed for the intermittent aeration mode (15 min on/45 min off), at the DO set point = 0.5 g O_2/m^3. The results confirmed that the DO set point as well the aeration mode are explicitly important parameters that affect the deammonification process rate and efficiency. The most promising results were obtained when the short-term operation mode was applied (Al-Hazmi et al., 2017).

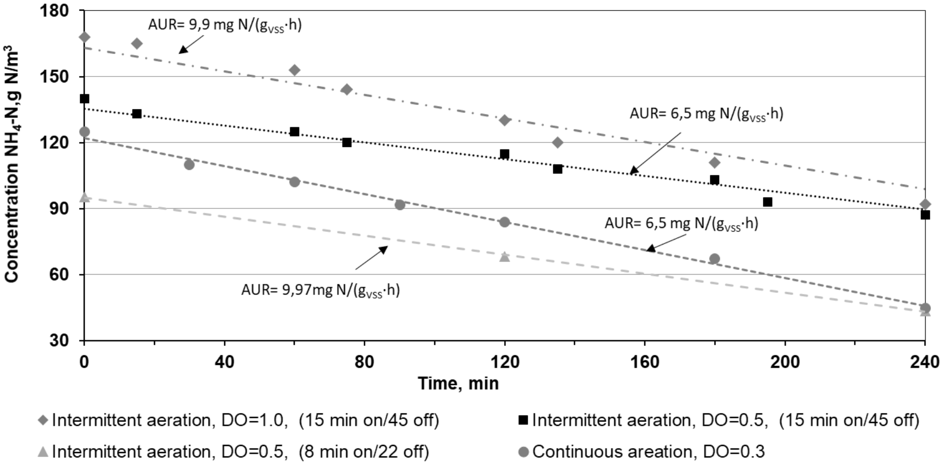

Fig. 4. Sample results of the AUR values in the experiments on the deammonification process carried out by Al-Hazmi et al. (2017)

4. *Incorporating the innovative nitrogen removal processes in the plant-wide models*

The implementation of the innovative nitrogen removal technologies allows to separate organic and nitrogen waste streams and thus maximize energy capture (Gao et al., 2014). Chemically enhanced primary wastewater treatment (CEPT) and the deammonification process give an opportunity for reduction of energy consumption. Primary sludge tanks removal efficiencies vary from 40% to 60% for total suspended solids (TSS) and from 25% to 40% for chemical oxygen demand (COD). By adding chemicals these efficiencies can be enhanced to about 80 to 90% for TSS and from 50% to 70% for COD removal (Kroiss and Cao, 2014). Gori et al. (2011) concluded that an increase in the particulate COD removed in primary clarifiers would result in reduction of the energy demand for aeration in the AS bioreactor, and the associated direct CO_2 emissions from microbial respiration and indirect CO_2 emissions from power consumption for aeration. However, it should be noted that care must be used during the process analysis since a fraction of COD is necessary for proper nutrient removal in the conventional AS systems.

A study on model-based evaluation of technological upgrades on the energy balance in a large biological nutrient removal WWTP in the city of Slupsk (northern Poland) was presented by Zaborowska et al. (2017). A simulation platform GPS-X (Hydromantis, Canada) was used for that purpose. Two strategies for upgrading the plant were considered. The first strategy predicted the impact of increased raw sludge withdrawal as a result of the CEPT. The second strategy employed the advanced nitrogen removal processes for sidestream treatment (deammonification via partial nitrification/anammox). The simulations revealed the potential for an increased biogas production in the anaerobic digester up to 56% and reduction in the electric energy demand for aeration up to 36%, while still maintaining the required total nitrogen (TN) effluent standard. The proposed upgrades improved the energy balance and could lead the studied WWTP from the energy deficit to the energy neutrality.

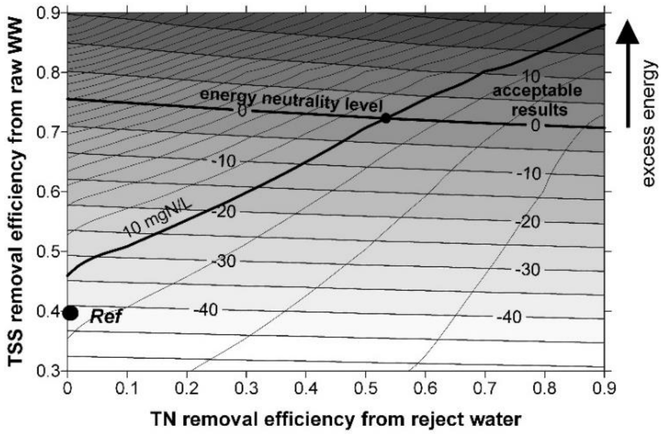

Fig. 5. Model predictions under the energy neutrality conditions (in %) for the studied technological upgrades (Zaborowska et al., 2017)

The energy positive area (exceeding the energy neutral point) was found for TSS removal efficiency higher than 75% and TN removal efficiency in the sidestream higher than 50% (Fig. 5). In the studied case, the operating cost balance depended mainly on the applied coagulants/flocculants and specific electric energy costs (Zaborowska et al., 2017).

5. *Energy neutral WWTPs implementing the innovative nitrogen removal processes*

It was discovered that 100% energy self-sufficient WWTPs are feasible by a combination of increased energy efficiency and energy harvesting from the wastewater. A few WWTPs have already achieved almost 100% (or higher) electricity self-sufficiency through energy efficiency and harvesting biogas and electricity (Wang et al., 2016). The Strass WWTP in Austria (200,000 PE) is one of the most commonly discussed facilities with the energy recovery efficiency as high as 108%. The approaches used to reach the goal included application anammox for sidestream ammonium removal and adoption of on-line sensor based dynamic control of intermittent aeration. These approaches were complemented with the two-stage AS to maximize COD fed to the anaerobic digesters to increase biogas production and adoption of high efficiency electricity generators (Kroiss and Cao, 2014).

6. Conclusions

Both the optimization of the aeration process control and the application of the innovative nitrogen removal technologies were overviewed in the study as the measures to improve the energy balance in WWTPs. The novel control of aeration based on the ABAC or the AVN system can provide significant savings in electricity consumption and support efficient nitrogen removal. However, it is expected that the future of the control systems in WWTPs belongs to smart process control systems cooperating with simulation models. Compared to the conventional nitrification-denitrification, the advantage of the shortcut nitrogen removal processes is the reduction in the electrical energy demand for aeration and the production of the sludge, as well as no need for organic carbon. Numerous researchers investigating the deammonification process emphasized the significance of pH, temperature, DO concentration, as well as the aeration patterns. The energy self-sufficient wastewater treatment can be achieved when a combination of various approaches is used. It was proved that the implementation of the innovative nitrogen removal technologies allows to separate organic and nitrogen waste streams and thus maximize energy capture. A shift towards energy neutral or even energy positive WWTP is possible without compromising wastewater effluent standards.

Acknowledgments

This study was supported by European Regional Development Fund within the framework of the Innovative Economy Operational Programme under the project no. UDA-POIG.01.03.01-22-140/09-01.

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Impact of water consumption patterns on accuracy of hydraulic modeling of water pipe network

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Abstract

Mathematical models are basic tools to simulate the work of Water Distribution System (WDS). Constructing of such prediction tool is a complex task, which requires the most detailed data. Information needed to create it can be divided into two categories: network data and WDSs operational data. The first category consists of attributes describing pipes and junctions, such as location (junction coordinates: x, y, z), length and diameter, also water demand data. The second category contains data, that specifies network performance such as: pumps hydraulic curves, water consumption patterns or control of WDS. Quality of data listed above will be reflected in simulation error, that will determine the compatibility of claimed hydraulic parameters with the real flow and pressure quantities. The error scale determines the usefulness of hydraulic model to administrate the current exploitation of WDS as well as undertaking strategically decisions concerning modernization directions and development of water supplying system for pointed municipality.

During dynamic modeling, daily water consumption patterns have the essential influence on simulation accuracy. Their structures will be different, depending on the customer type (domestic, industry), or analyzed period. Water consumption dynamics is the motion wheel determining workflow of the network. Changes in water consumption patterns are impacting calibration accuracy. There are three numeric simulation scripts presented in this paper, considering diversions of customers, water consumption patterns created for different periods of WDS exploitation: one year, six months and one month. Water consumption patterns were automatically created during data export from GIS database based information from the SCADA system.

Keywords

water distribution system, hydraulic simulation model, EPANET, water consumption pattern

1. Introduction

Mathematical models are commonly used in WDS as supporting tool for making decisions. The purpose of network modeling is to reflect WDS operational work in any period with the highest possible accuracy, independently of chosen period or occurred failure, which are analyzed. Before model using, it is necessary to calibrate it, which is determining physical and operational data of the WDS in order to obtain the slightest difference between real and simulated values (Shamir and Howard, 1977; Kaplan, 2002), therefore calibration is the most important step in modeling process. During modeling, all data that represent network graph and WDS performance are verified. The highest uncertainty of data are related to pipe roughness and water demand (Walski, 1983; Sanz and Perez, 2014). These are elements, which are verified and defined in micro-calibration (Walski, 1983; National Modelling Guidelines, 2009; Savic et al., 2009; AWWA, 2012), the final stage of creating mathematical model. In case of pipe roughness, this parameter is dependent to the pipe diameter, material, age and water quality, which can be defined by mathematical function or systemized. Demand data represent the total water supplied to the distribution system as well individual customers sale, group customers sale and water losses. The variable describing water consumption is a dynamic random factor, characterized by its requirement variation (daily as well as periodically) of different type of customers. In hydraulic analysis there are five main types of water demands: residential, commercial, industrial, wholesale demands, and water loss, however, this group can be expanded to include more specific customers classes such as military bases, prisons, universities, and hospitals (AWWA, 2012).

The essential role of the Extended-Period Simulation (EPS) is characterized the water consumptions patterns (diurnal curves), which determine customer character and the conditions of the WDS operational work. Diurnal curves represent the ratio of hourly demand to average demand in a given period of time. Water consumption patterns may vary depending on the customer classes (residential, industrial, commercial) and season. Residential areas consist of domestic and irrigation consumption, characterized by two maximum water consumptions per day, morning when people prepare for the day and early evening as people return home (Umapathi et al., 2012; CBCL, 2011; Carragher et al., 2012). Commercial areas consist of demands by stores, restaurants, gas stations and offices, are characterized by a maximum consumption in the afternoon, while industry water consumption patterns are determined by its character of work, the maximum water consumption may occur in the morning, afternoon or at night. Factories are increasingly using the internal water cycle, which results in less water demand during a day (or longer period), and only water shortages are replenished from the WDS. These water intakes can be a large part of the water demand for a given zone and can cause uncontrolled peaks in diurnal curves (for given zone) and pressure drops in the network. Figure 2 shown examples of water consumption patterns for residential, service, industry and business areas. The season also influences the appearance of the water consumption patterns, during watering the gardens, or filling swimming pools, there is a noticeable increase in water demand. Fig. 1 shows differences between maximum day of year (summer season) and typical winter day, the water demand and amplitude is greater in the summer than in the winter season.

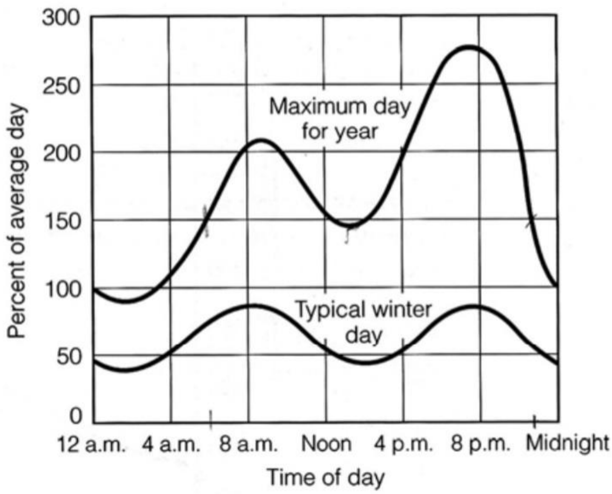


Fig. 1. Comparison of water consumption pattern for the maximum day of year and typical winter day (Chin, 2000)

Water consumption patterns can be created for a particular customers class or for a specific zones, but in the second case, the customers with high water intake will have the greatest impact on the course of the diurnal curve which may result in lower accuracy of the model.

Data used to create of hydraulic model are mostly based on billing databases, i.a. Customers Information System (CIS) and SCADA (Supervisory Control And Data Acquisition) systems. Water demand are exported from billings database (average, maximal or minimal value of water consumption for chosen period), while water consumption patterns are created form SCADA systems.

According to modeling algorithm, water consumption pattern is determined by formula below (P´erez et al., 2011):

$$d_i(k) = d_{basei} \cdot \text{pattern}(k) \quad (1)$$

where: $d_i(k)$ – i junction water consumption at any time k, d_{basei} – i water consumption (value from billing databases), pattern(k) – water consumption pattern exported from SCADA database.

The sum of momentary water consumptions should equate total daily water consumption in node:

$$\sum_i d_i(k) = d_{total}(k) \tag{2}$$

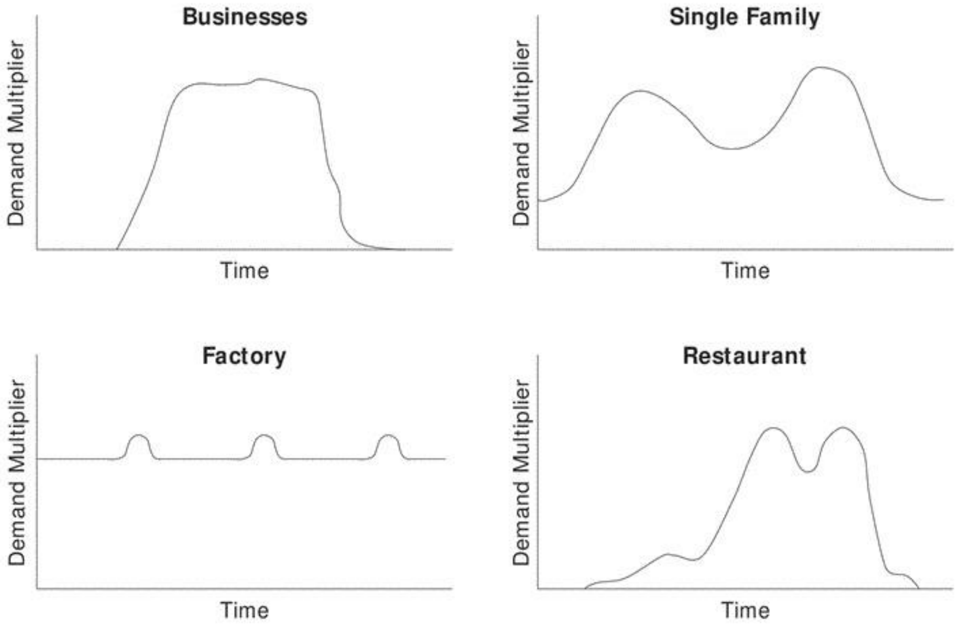


Fig. 2. Water consumption pattern for different type of areas (Walski et al., 2007)

In the modeling process of the WDS, the choice of the simulation period should be preceded by precise analysis of the network performance and water consumption, so that it reflects the normal operating work of the WDS.

2. *Research subject - selected area of the WDS*

The subject of the study is the selected water supply area of the large WDS. The water supply system consists of four Water Treatment Plants (WTP A, WTP B, WTP C and WTP D) with a total daily average production of 55,000 m^3 and four complexes of tanks (H, G, E and F Tanks) with a total capacity of 162,000 m^3 (Fig. 3). Tanks E are additionally supplied from a pumping station (PUMPING STATION I) located outside the considered area of WDS, in an average daily amount of 60,000 m^3. The average daily amount of supplied water in this area is 115,000 m^3. Daily water demand for this area is 102,000 m^3. Considered WDS is an wide network with a total length of 256 km. The water supply infrastructure is characterized by high variability of material and diameters from 55 mm to 1600 mm (Tab. 1). The WDS is mainly made of steel (73%) and polyethylene PE-SDR17 (10.6%), with a small share of ductile iron (Tab. 1). The oldest pipelines that build this distribution subsystem come from 1929 (steel wires) and the latest from 2016 (PE-SDR17).

The central point of the subsystem is the storage tanks E, which is supplied from two directions (WTP A and PUMPING STATION I) and supplies water to the largest number of customers, representing nearly 50%. Due to the boundaries of the subsystem, the storage tanks F are the water receiver (normally supplied water in five directions). Storage tanks G supplied the smallest area due to pipes failures occurred in considered period. WTP D works periodically, in situations of increased water consumption (summer time) (Fig. 3).

Table 1. Percentage of diameters divided by material.

Diameter [mm]	**Material**	**Participation [%]**
79–141	PE-SDR17	1.78%
200	Steel	2.53%
279	PE-SDR17	1.23%
300	Ductile iron	0.01%
354–398	PE-SDR17	3.31%
400	Steel	4.47%
400	Ductile iron	3.09%
443–496	PE-SDR17	4.02%
500	Ductile iron	6.50%
558	PE-SDR17	0.29%
600	Steel	4.10%
800	Steel	0.84%
1200	Ductile iron	2.31%
1400	Steel	22.67%
1600	Steel	42.85%

The WDS is supplying an urban-industrial area with a high prevalence of urban areas (93.8%). Industrial consumers often collect water irregularly or periodically, contributing to the maintenance of high network pressures around the clock at 75–100 m H_2O. Figures 4 and 5 show exemplary water consumption patterns for industry that show irregular water consumption. Figure 4 shows a customer receive water for 13 hours, and figure 5 shows the customer characterized by a certain regularity of water intake from morning to night.

In the case of domestic areas, the daily water consumption patterns are characterized by the standard regularity of the occurrence of two peaks in water consumption in the morning and in the evening (Fig. 6).

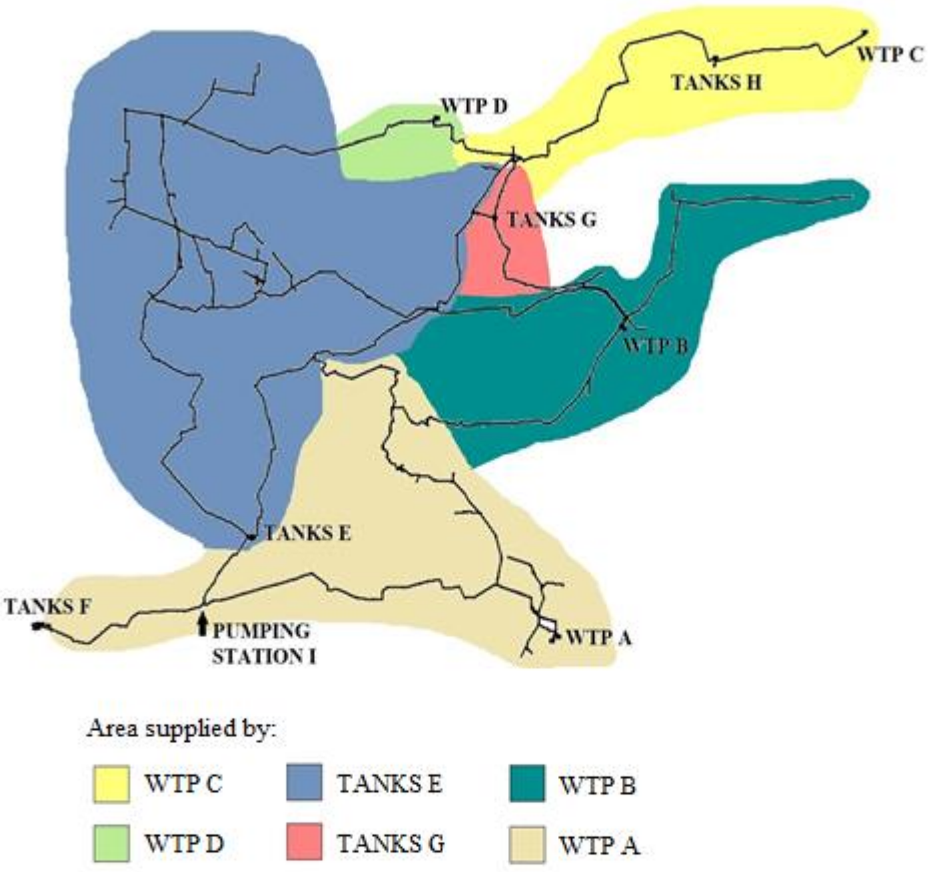


Fig. 3. Structure of WDS with water supply areas

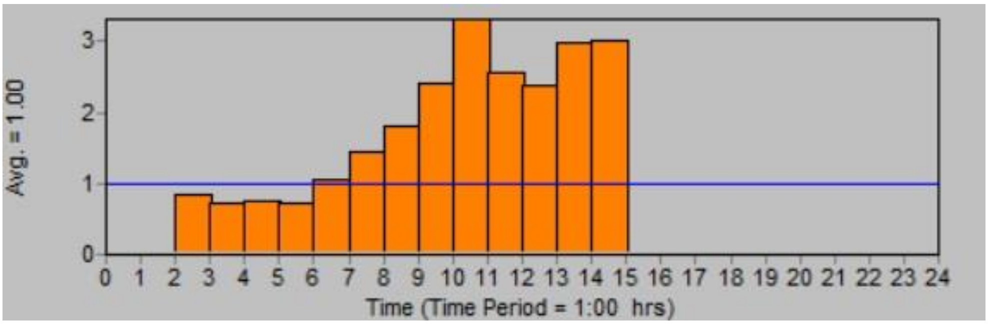


Fig. 4. Daily water consumption pattern for selected industrial customer

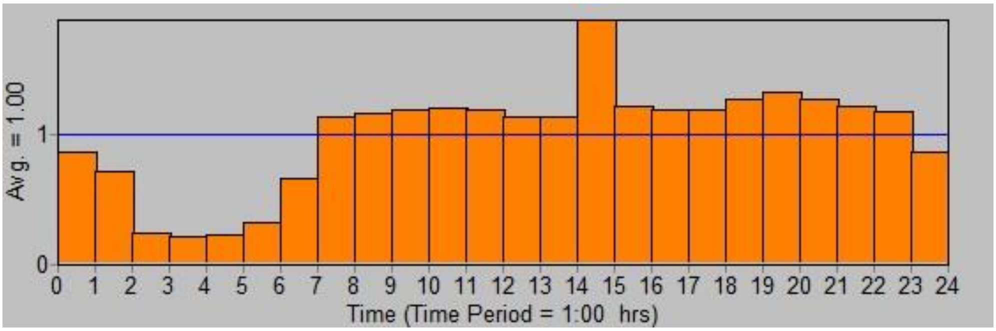
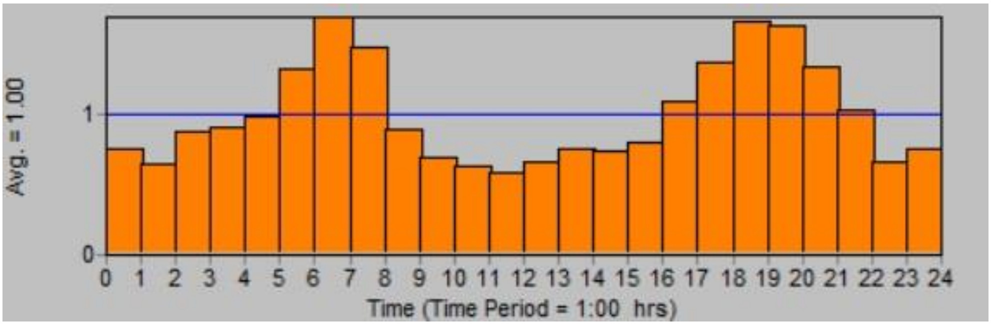


Fig. 5. Daily water consumption pattern for selected industrial customer



Rys. 6. Daily water consumption pattern for selected domestic customer

3. Assumptions of the simulation and results discussion

The calibrated hydraulic model of the WDS was used in the study, EPANET 2.0 was used for the simulation. The model was developed within the framework of NCBiR project POIG.01.03.01-14-034/12. The network graph was exported from the GIS (Geographic Information System) database, while the water demand data, from the one year period (2016), was exported from the available billing databases. The project uses automatic generation of water consumption patterns based on telemetry systems, at points covered by the system. For other consumption points, these patterns were created on the basis of water supply points (WTP, etc.). Model is built from 1488 pipes and 1989 nodes, 524 valves, 22 pumps, 4 tanks and 4 reservoirs. The model was calibrated for one-month data (October 2016). Fig. 7 and 8 show the daily variation of the flow rate for supplying points for two characteristic periods (maximum and minimum day). Small changes between the characteristic days (at the level of 20 000 m^3/day) are due to a short period of analysis, during which there was no variation in water consumption.

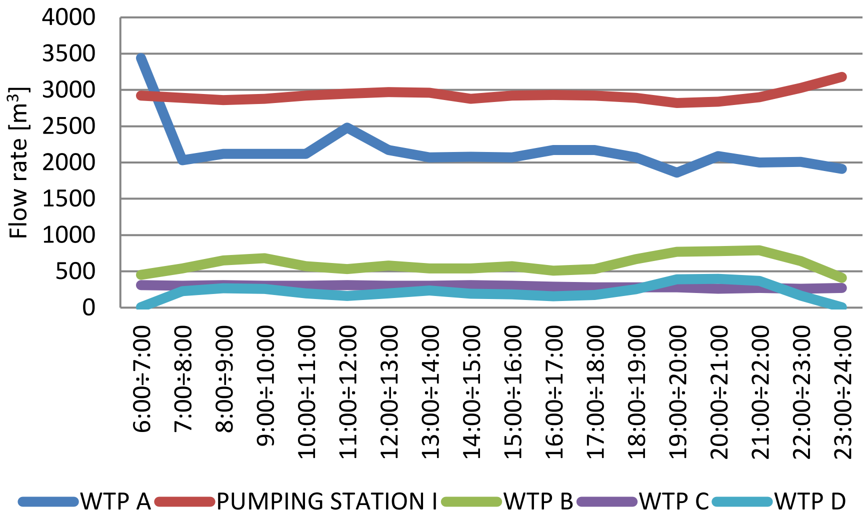


Fig. 7. Flow rate at supplying points for maximum values for considered period

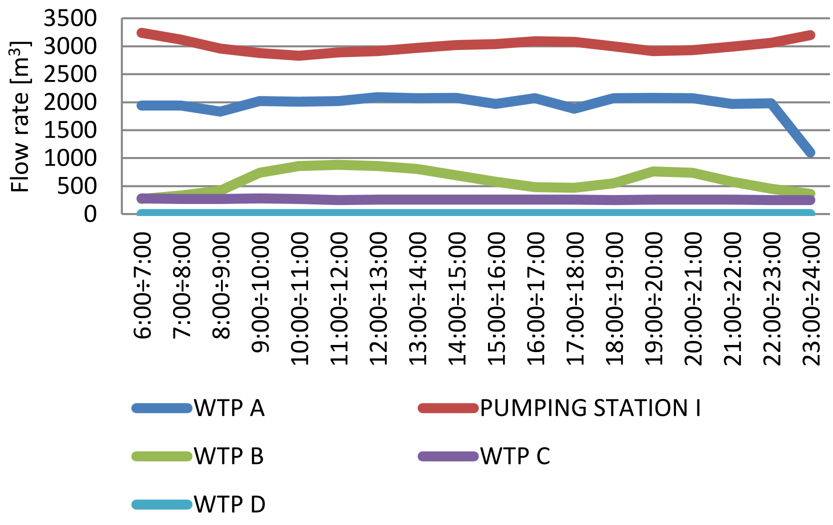


Fig. 8. Flow rate at supplying points for minimum values for considered period

Daily water consumption patterns were created from actual data collected from the SCADA telemetry system. Model validation covered a period of three days (17, 18, 19 October). Correlation of simulation results and actual measurements for subsystem is as follows: pressure 99.2%, flows 98.5% (Fig. 9 and 10).

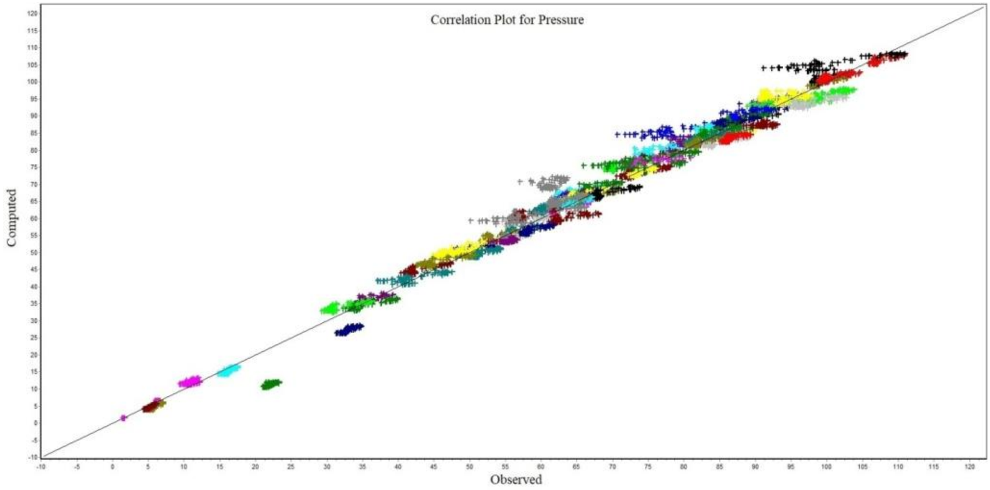


Fig. 9. Correlation plot for pressure, calibration results.

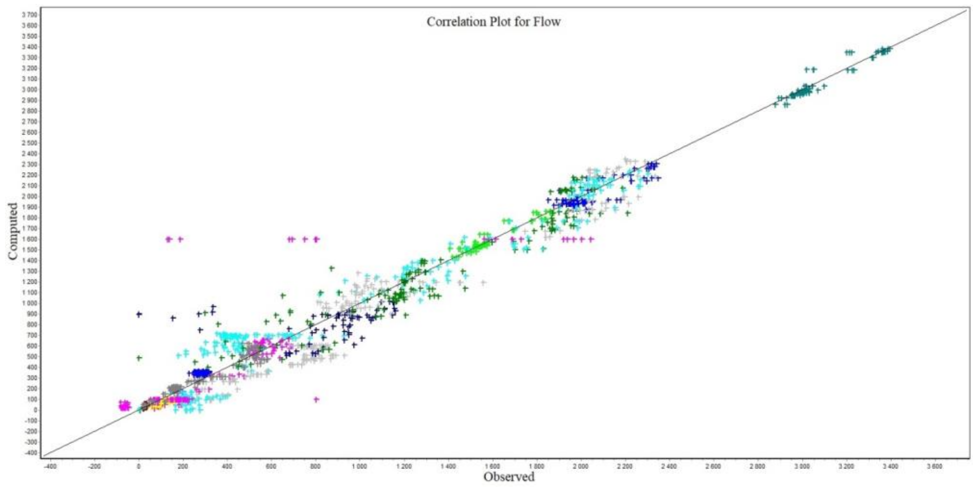


Fig. 10. Correlation plot for flow, calibration results

This calibrated model was the basis for further analysis. Simulations were performed for three periods for which the data were exported form available database:

- Scenario - simulation for one month (October)
- Scenario II - simulation for the period of 6 months (second half of the year)
- Scenario III - Simulation for one year

For each considered period, the water consumption patterns were created from the mean values.

Scenario I:

The nodal water demand and water consumption patterns were exported from the billing databases and the SCADA database from one month (October). This is the period for which the network model has been calibrated. In the simulation there are 267 nodes with a total average water demand 105.500 m^3/day. Compatibility of the simulation result with actual measurements: for flows is 98.5%, and for pressures 99.2%.

Scenario II:

The nodal water demand and water consumption patterns were exported from the billing databases and the SCADA database for 6 months (second half of 2016). In the simulation there are 275 nodal water demands with a total average water demand of 104.500 m^3/day. In this period, there are more nodal water demands due to periodic water intake from some water consumption points. Compatibility of simulation result and actual measurements: for flow is 98.7%, and for pressures 99.3%.

Scenario III:

The nodal water demand and water consumption patterns were exported from the billing databases and the SCADA database for a one year period (2016). In the simulation there are 281 nodal water demands with a total average demand of 123.900 m^3/day. Also during this period, more water demands were recorded, due to periodic water intake from some water consumption points. Compatibility of the model with actual measurements for flows is 97.5%, and for pressures 98.7% .

Considering the calibration criteria, established by Water Association Authorities and WRc (Tab. 2), and the simulation results it can be stated that the analyzed models have a high degree of calibration and can be the basis for various analyzes.

Table 2. Calibration criteria (Walski et al., 2007; Apaydin, 2013; EPA, 2005)

Flow Criteria
(1) Modeled trunk main flows (where the flow is more than 10% of the total demand) should be within ±5% of the measured flows
(2) Modeled trunk main flows (where the flow is less than 10% of the total demand) should be within ±10% of the measured flows
Pressure Criteria
(1) 85% of field test measurements should be within ±0.5 m or ±5% of the maximum head loss across the system, whichever is greater.
(2) 95% of field test measurements should be within ±0.75 m or ±7.5% of the maximum head loss across the system, whichever is greater.
(3) 100% of field test measurements should be within ±2 m or ±15% of the maximum head loss across the system, whichever is greater.

A detailed analysis was conducted for selected area supplied by Tanks E (Fig. 3, color blue). The following figures (Fig. 11–16) shows the results of the simulations results for two outlets from Tanks E (west and east directions).

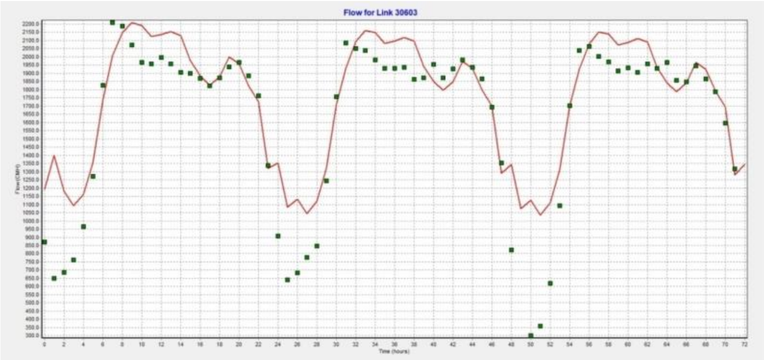


Fig. 11. Results of Simulation I for Tanks E, east direction

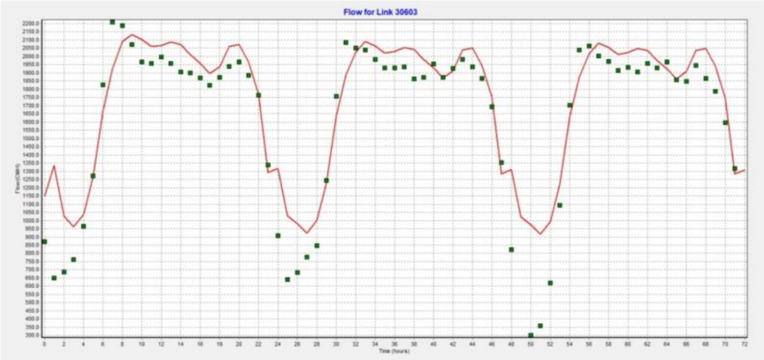


Fig. 12. Results of Simulation II for Tanks E, east direction

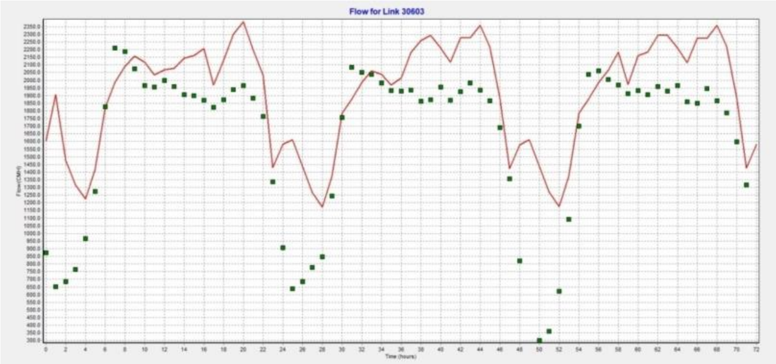


Fig. 13. Results of Simulation III for Tanks E, east direction

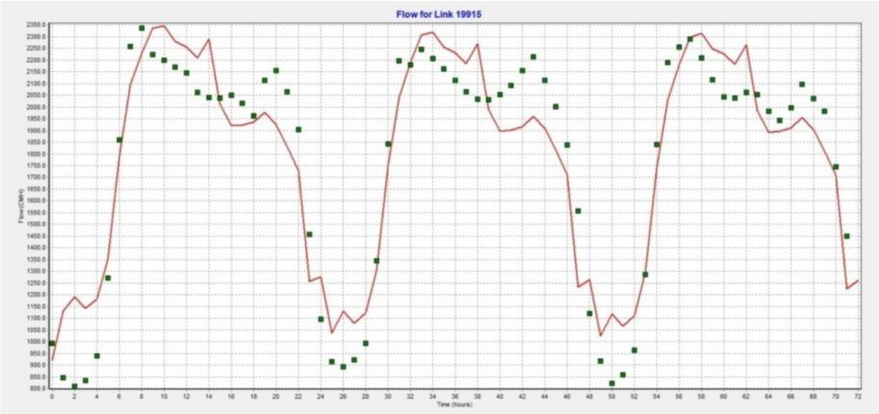


Fig. 14. Results of Simulation I for Tanks E, west direction

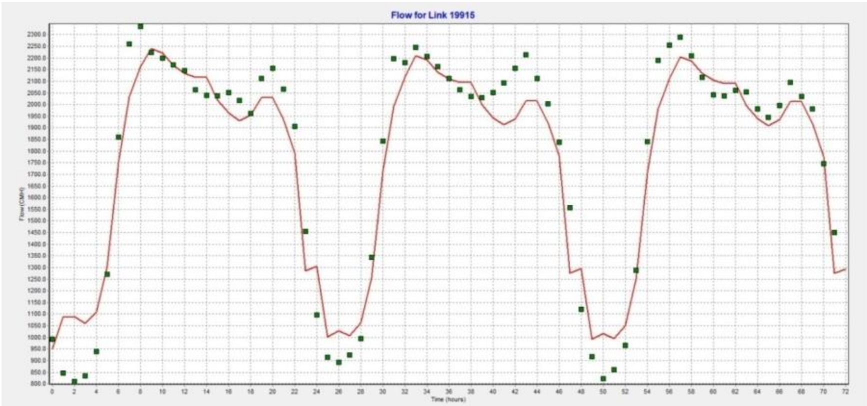


Fig. 15. Results of Simulation II for Tanks E, west direction

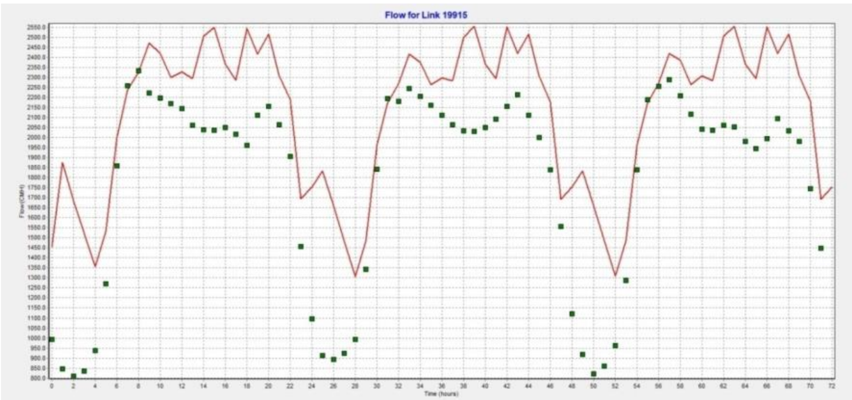


Fig. 16. Results of Simulation III for Tanks E, west direction

The simulation results for the 6-month data are better than one month and the whole year. The cause may be various random events occurring in the given periods. The period of one month may seem too short to analyze the operation of the network and for modeling, e.g. in the event of a failure of the measuring device, incorrect data can be obtained, but there is no seasonal variation in water consumption. For 6 months, even in the case of devices failure, reliable data can be obtained (e.g. from months when there are no disturbances), but seasonal variation occur (summer and winter). Yearly data includes all events that occurred on the network, including winter variability, spring (watering of gardens), and summer. Where nodal water demands and water consumption patterns are formed of average values, significant differences may be observed in relations to actual water consumption. The figures below show daily water consumption patterns for selected water intake points from the area supplied by Tanks E. Figure 17 shows the variability of the water consumption pattern, depending on the time period, for domestic area, and Figure 18 for industry area. The graphs show that the longer period, from which patterns are created, the hourly demand values are closer to the daily average demand value (line = 1), this means that they are more stable and correspond to the normal operation of the network.

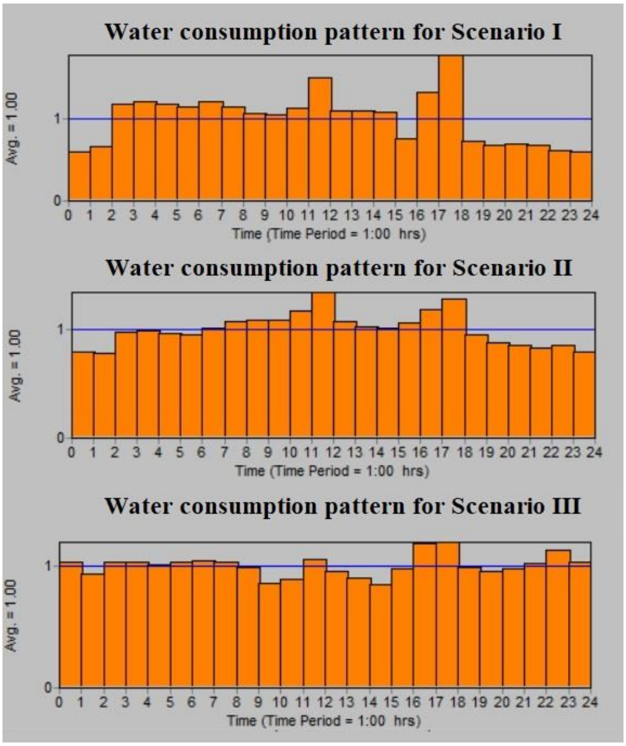


Fig.17. Changes of daily water consumption patter depending on the export period; domestic customer

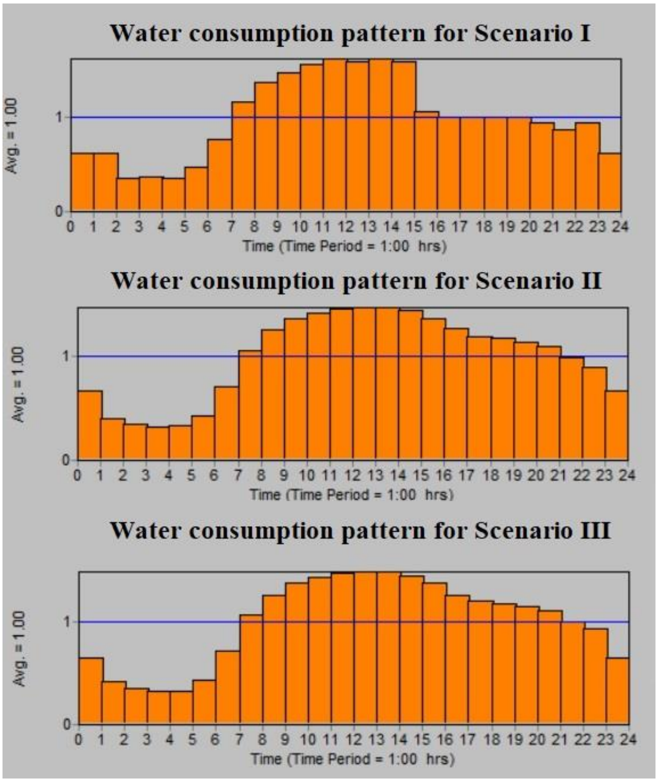


Fig.18. Changes of daily water consumption patter depending on the export period; industrial customer

4. Conslusions

All the data used to build the WDS model should be analyzed and verified, such as network graph, operational data (pump and valve settings) and water demand data (quantity and daily consumption pattern). Network graph information is constant (rarely changed) and should be checked periodically, while water demand data are changing over time. During model calibration and validation, it is important to choose a period that reflects the normal operation of the WDS. This period should not be too short, due to the possibility of failure of measuring devices, or too long to avoid seasonal variations in water consumption. While the analysis period is extended with more detailed data, it is important to remember that random events may occur at this time, which will not reflect the normal operation work of the system. During calibration should pay attention to all aspects of network performance, especially the water consumption patterns, to get the highest level of detail of the model.

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Microbiological contaminants of Legionella spp. in hot water in public buildings in Silesia province

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Abstract

Directive 98/83/WE, as one of its major goals, aims to introduce rules for risk assessment in water supply system (WSS) in Member States. This risk assessment should be based on water quality control as well as on condition estimation of the level of systems functioning. That is why a procedures of risk management have to be implement in daily operation practise in ecsploitatiom of water supply system. Realisation of this goal will improve safety of water supply, thus ensure better safety of water consumers. In Poland, this directive is implemented, among others, by Regulation of Ministry of Health on water for human consumption which also defines the extend of quality control of hot water based on *Legionella spp*. identification. That could be the cause of legionnaire's disease.

This paper presents the results of assessment of hot water quality as regards to contamination with *Legionella spp*. after actions undertaken by controlling units – Sanitary Inspection. A legal act from 2008 introduced an obligation to examine levels of *Legionella spp* in hot waters in communal housing facilities and closed health care facilities and from June of 2016 this rule also applies to swimming pools.

Using available water quality data disclosed by Sanitary Inspection, an analysis of the changes in sanitary quality of hot water was performed. In this analysis, both, type of used facilities as well as its special distribution thus affiliations to appropriate controlling unit – Sanitary Inspection in the Silesian province were considered. The data used in comparison analysis was collected directly after the imposition of the obligation to make such tests. For communal housing facilities and closed health care facilities the data from 2008, 2010, 2015 and 2017 was used. While for swimming pools the data from second half of 2016 and first half of the 2017 was used. Using performed analysis of hazard for humans caused by Legionella spp. contamination objections were pointed out to address inconsistencies of existing legislations.

Keywords

microbiological contamination, risk, hot water, Legionella spp

1. Introduction

In direct relation to drinking water, the Council Directive 98/83/EC of 3 November 1998 on the quality of water intended for human consumption (Counsil Directive, 1998) which defines, inter alia, the principles of water quality control as well as values of index parameters has been established. Its revisions (Commission Directive, 2015) introduce the change in the approach to safety of water supply systems. Safety provision shall be based on risk assessment. The means for realisation of the aforementioned is the implementation of the principles of responding to threats identification in the water supply systems (WSS) on the basis of both water quality control and evaluation of conditions of functioning of these systems. The application of risk management procedures in WSS is to ensure more effective provision of safety of water supply the effect of which is – the protection of consumers' health from adverse consequences of tap water contamination. In Poland, the matters of water supply are regulated by two acts: Water Law (Polish Law, 2001a) and the act on collective water supply and collective sewage disposal (Polish Law, 2001b). The two acts specify the responsibility of water companies for the quality of water fed to the water connection. The act defining the principles of water quality control both by water companies and the control authority is the regulation of the Minister of Health (Polish Law, 2015). This regulation also includes the principles of water quality control in terms of Legionella spp. The obligation to control water in this scope was introduced in the previous regulation of the Minister of Health (MH) (Polish Law, 2007) and included the obligation to test water only in twenty-four-hour health care centres (24hHCC) and multi-apartment residential buildings (MARB) as of 2008. The currently applicable legislation does not define the entity in charge for conducting controls in this scope. The regulation of 2015 (Polish Law, 2015) imposes the obligation to perform tasks in 24hHCC on the official control authorities, i.e. State Sanitary Inspection. In the second half of 2016, the legislator introduced the obligation of performance of water tests for Legionella spp. in swimming pools (Polish Law, 2016) through the new regulation. This obligation was assigned to the facility manager and the control authority performs the test minimum once a year.

Legionella spp. bacteria are gram-negative, non-endosporic rods, also forming polar, ciliated flagella, not generating capsules. They do not decompose sugars, do not reduce nitrates, and do not need vitamins for growth. This genus includes 49 species, for which over 70 serotypes have been identified (Grabińska-Łoniewsa i Sińskie, 2010). These microorganisms grow in aerophilic conditions in temp. 15–43°C, with optimum temperature 30–43°C. They can proliferate even in temp. 67°C, but they also adapt to low temperatures. The natural place of existence of Legionella spp. are underground, surface, thermal, sea waters and even soils. Such a broad scope of life-supporting temperatures facilitates settlement in ecosystems, e.g. through elimination of inter-species competition.

Legionella spp. can trigger a so-called Legionnaires' disease and Pontiac fever in a human. The main source of exposure to these microorganisms is the respiratory tract. The second way of penetration is the gastrointestinal tract. On the basis of the conducted literature review, it can be concluded that some countries keep register and list of diseases having their source in water. These diseases are called waterborne diseases. For example, such a register of this type of diseases with the source in water is kept by the organisation Waterborne Disease and Outbreak Surveillance System (WBDOSS) based in the USA. Its goal is to collect information on the occurrence and characteristics of waterborne diseases and epidemic focuses. WBDOSS is the result of cooperation of the Council of State and Territorial Epidemiologists (CSTE), Environmental Protection Agency (EPA) and American Center for Disease Control (CDC) (www.cdc.gov, 2017). The drafted reports, studies and analyses are used at different levels of state organisation (Mulik et al., 2015a; b). The study of years 2011-2012 shows that 32 events as the result of which 431 persons fell ill, including hospitalisation of 102 (www.cdc.gov, 2017) were observed in the USA due to waterborne diseases. In many European countries there are also institutions dealing with identification of waterborne diseases. For example: National Public Health Institute in Finland, the Health Protection Agency in the UK, the Robert Koch Institute in Germany, the National Institute for Public Health and the Environment in the Netherlands, the National Institute for Public Health Surveillance in France or the Swedish Institute for Infectious Disease Control (Beer et al., 2011–2012). Diseases caused by bacteria of Legionella spp. constitute a serious health problem. The report of CDC shows that, in the USA, in 2011–2012 that 66% of illness cases were actually caused by Legionella spp. (Beer et al., 2011–2012).

The water should be tested for Legionella spp. in the hot water plumbed system, including the heat distribution centre, plumbed water, including the storage tank and water treatment facilities, air-conditioning systems (with humidifier), cooling systems (with cooling tower and evaporative condensers), water swimming pool systems and, most of all, swimming pools with hydro massage as well as bubble bathtubs, in bathrooms, fountains, rainwater systems and tanks, sprinkler systems, wastewater systems (grey water), solar systems, fire-fighting systems and tanks (Toczyłowska, 2013).

Despite of obligations to test water for Legionella spp., there is no defined obligation to keep the register of the facilities in which water must be tested in this regard. The legislative provisions regarding multi-apartment residential buildings are construed so that the parameter value for Legionella spp. is defined, but there is no entity defined which shall perform the tests. There is lack of obligation to conduct control, e.g. in the multi-storey buildings with residential hot water system. There is also lack of data on the water quality in these types of facilities, which in combination with lack of register of waterborne diseases can pose an epidemic threat.

2. *Research subject*

The data obtained from water quality monitoring performed in the scope of internal control by the facility administrators and by the local competent body of State Sanitary Inspection (SSI) for the Silesian Voivodeship were subject to analysis. This voivodeship is the unit of the territorial self-government and the unit of administrative division of Poland with surface of 12,333.09 km², resided by ca. 4.6 million people. This voivodeship has the highest number of cities with 100 thousand citizens in Poland. This is the only voivodeship in Poland where there are fewer districts (17) than cities with district rights (19). This proves high urbanisation level the derivate of which is the number of facilities where the water quality control for Legionella spp. should be conducted. The ongoing sanitary supervision, including supervision of water quality, is conducted under the Act on State Sanitary Inspection (Polish law, 2017) by 20 State District Sanitary Inspectors (SDSI) and Silesian State Voivodeship Sanitary Inspector (SSVSI) (www.higienawody.wsse.katowice.pl, 2018). For the purposes of this research, the voivodeship was divided according to the competence of the local SSI.

Table 1. Characteristics of the local SDSI in the Silesian Voivodeship

No.	**State District Sanitary Inspectors**	**The number of residents**	**Area [km^2]**
1	SDSI in Bielsko Biała	303,000	582.16
2	SDSI in Bytom	379,000	751.73
3	SDSI in Chorzow	159,000	46.82
4	SDSI in Cieszyn	245,000	730.2
5	SDSI in Częstochowa	404,000	1679.1
6	SDSI in Dabrowa Gornicza	204,000	555.83
7	SDSI in Gliwice	464,000	877.67
8	SDSI in Jaworzno	93,000	152.2
9	SDSI in Katowice	455,000	255.79
10	SDSI in Kłobuck	84,000	889.15
11	SDSI in Lubliniec	72,000	822.13
12	SDSI in Myszkow	69,000	478.62
13	SDSI in Raciborz	162,000	543.98
14	SDSI in Ruda Śląska	161,000	77.59
15	SDSI in Rybnik	303,000	437.53
16	SDSI in Sosnowiec	207,000	91.26
17	SDSI in Tychy	338,000	943.29
18	SDSI in Wodzisław Slaski	229,000	372.36
19	SDSI in Zawiercie	122,000	1003.27
20	SDSI in Żywiec	85,000	1039.96

The results of water quality tests for Legionella spp. conducted in 24hHCC and MARB in years 2008–2010 and 2015 as well as in swimming pools in 2016 and the first six months in 2017 were analysed (SSVSI, 2008, 2010, 2015–2017). Table 1 presents the number of residents and the area subject to competent local SSI divisions in the Silesian Voivodeship. There is no obligation of identification and registration of the facilities in which the control of residential hot water system must be conducted for Legionella spp. The SSI authorities hold only the list of swimming pools covered with supervision. In 2016, the water quality control for Legionella spp. was conducted in 293 swimming pools. It is important to highlight that the sanitary supervision in 24hHCC and MARB is conducted only by 20 SDSI for the Silesian Voivodeship. Whereas the supervision over swimming pools is additionally conducted by SSVSI for the facilities for which the district is the founding body or in which the district is the dominant entity (Polish law, 2017).

3. Research methodology

In order to perform the classification of contamination level, the law recordings were used (Polish Law, 2007; 2015; 2016). Table 2 presents the classification of contamination level in relation to the number of colony forming units of Legionella spp. in 100 ml of water sample. Above mentioned classification of contamination evaluation is the same like in case of 24hHCC, MARB and swimming pools. However the procedure for taking corrective actions for a given contamination level is completely different for swimming pools. The goal of this research was to perform the fast and transparent evaluation of the tap water quality in facilities with residential hot water system on the basis of available data, i.e. for 24hHCC, MARB and swimming pools. In the proposed research methodology, the analysis of microbiological parameters such as Legionella spp. was conducted. At the first stage of this analysis, for each area of operation of the given SDSI, in division to 24hHCC and MARB, the probability of occurrence of lack of contamination or a particular contamination level (formula 1) in given specific periods of the analysed time was defined.

$$P_i = \frac{NS_i}{CS_i} \tag{1}$$

where: P_i – probability of occurrence of *i*- this contamination level in *i*- th facility type for *i*- th area of operation of PSI authorities (testing region), NS_i- number of water samples for which the occurrence of *I* – th contamination level in *i*- th area of operation of PSI authorities (testing region) was stated, CS_i – number of water samples tested for Legionella spp. in *i*-th facility type in *i*-th area of operation of PSI authorities (testing region).

In case of swimming pools, the methodology additionally includes the facilities directly supervised by SSVSI. In the analysed set of data, the number of control samples (CS) which were tested for Legionella spp. is defined. At the next stage, the number of samples (NS) for which a given contamination level was observed is identified. The determination of these values allows for determination of the probability of occurrence of a given microbiological contamination level in a given type of facilities in a given period of the research time.

Table 2. Classification of assessment of microbiological contamination level together with the procedures of immediately undertake corrective action

No. of Legionella spp.	**Assessment of microbiol. contamination**	**Procedures of undertake corrective action procedures***	**Test schedule**
<100	no contamination / negligible contamination	The system is under control - this state does not require to be taken special action.	After 1 year or after 3 years (if in two subsequent tests, no or negligible contamination was found at annual intervals).
100–1000	mean	If most of the samples are positive it should be water pipe-network considered as colonized by Legionella spp. - then you should find the cause of bacteria grow (make technical network review, check water temperature) and take an action to reduce the number of bacteria. Further actions (cleaning and disinfection) depend on the result of next sampling	After 4 weeks, if the test result does not change, you should clean and disinfect the hot water and water plumbed systems, then repeat the test after 1 week and after 1 year.
1000–10000	high	Immediately you should start the intervention action as above, including cleaning and disinfection - water is not suitable for showers.	After 1 week of cleaning and disinfection, then every 3 months the test of water should be done. If in the next two tests carried out at intervals of three months was found less than 100 CFU /100 ml, then the next test can be done after a year.
>10000	very high	Immediately you should disconnect domestic hot water system from operation and to start cleaning and disinfection.	After 1 week of cleaning and disinfection, then every 3 months the test of water should be done. If in the next two tests carried out at intervals of three months was found less than 100 CFU /100 ml, then the next test can be done after a year.

* the procedure does not apply to the swimming pool

4. Results and Discussion

The results of microbiological water quality analysis in research period for 24hHCC and in MARB are presented in table 3 and table 4 respectively.

Table 3. Characteristics of water quality in terms of the occurrence of Legionella spp. in 24hHCC in 2008-2010 and 2015

State District Sanitary Inspectors (SDSI) in	**2008**				**2010**				**2015**			
	Probability of contamination											
	Samples number	**mean**	**high**	**very high**	**Samples number**	**mean**	**high**	**very high**	**Samples number**	**mean**	**high**	**very high**
Bielsko Biała	-	-	-	-	3	0.333	0.333	0	-	-	-	-
Bytom	174	0.448	0	0	-	-	-	-	26	0	0	0
Chorzow	47	0.234	0.213	0.043	24	0.083	0.250	0	10	0.100	0	0
Cieszyn	66	0.242	0.182	0.015	63	0.032	0.063	0.016	29	0.069	0	0
Częstochowa	44	0.091	0.068	0	17	0.294	0.059	0	6	00	0	0
Dabrowa Gornicza	24	0.333	0.375	0	10	0.100	0.400	0	-	-	-	-
Gliwice	33	0.303	0.182	0	35	0.343	0.029	0	30	0.133	0	0
Jaworzno	20	0.250	0.200	0	28	0.036	0.143	0	10	0	0	0
Katowice	11	0.235	0.084	0	80	0.363	0.038	0	15	0.133	0	0
Kłobuck	9	0	0	0	8	0	0.125	0.375	-	-	-	-
Lubliniec	75	0.067	0	0	49	0.122	0	0	29	0.241	0	0
Myszkow	-	-	-	-	4	0	0	0	-	-	-	-
Raciborz	20	0.200	0.100	0	43	0.233	0.256	0.023	11	0	0	0
Ruda Śląska	25	0.280	0.400	0	22	0.227	0	0	3	0	0	0
Rybnik	32	0.094	0.500	0.031	22	0.045	0.182	0	10	0.100	0.100	
Sosnowiec	30	0.233	0.467	0	30	0.267	0.200	0	-	-	-	
Tychy	32	0.031	0.094	0	30	0.100	0	0	1	0	0	
Wodzisław Slaski	42	0.095	0.167	0	34	0.059	0.059	0	15	0	0.067	
Zawiercie	10	0.400	0	0	6	0	0	0	5	0	0	
Żywiec	-	-	-	-	15	0.467	0	0	-	-	-	
Total	802	0.243	0.132	0.005	523	0.182	0.092	0.010	200	0.085	0.01	

Table 4. Characteristics of water quality in terms of the occurrence of Legionella spp. in MARB in 2008-2010 and 2015

State District Sanitary Inspectors (SDSI) in	**2008**				**2010**				**2015**			
	Probability of contamination											
	Samples number	**mean**	**high**	**very high**	**Samples number**	**mean**	**high**	**very high**	**Samples number**	**mean**	**high**	**very high**
Bielsko Biała	-	-	-	-	9	0	0	0	-	-	-	-
Bytom	24	0.292	0	0	124	0.258	0	0	67	0.149	0.134	0
Chorzow	-	-	-	-	9	0.444	0.333	0	8	0.375	0	0
Cieszyn	6	0	0	0	1	0	0	0	36	0.111	0.056	0.028
Częstochowa	-	-	-	-	16	0.063	0	0	12	0.167	0.167	0
Dabrowa Gornicza	-	-	-	-	40	0.475	0.325	0	15	0	0	0
Gliwice	50	0.140	0.300	0.020	52	0.192	0.173	0	29	0.172	0	0
Jaworzno	20	0.100	0.250	0	35	0.229	0	0	5	0	0	0
Katowice	55	0.182	0.164	0	90	0.200	0.078	0	35	0.257	0.114	0
Kłobuck	3	0	0	0	3	0.333	0	0	-	-	-	-
Lubliniec	95	0.295	0	0	93	0.226	0	0	44	0.227	0.182	0
Myszkow	-	-	-	-	6	0	0	0	2	0	0	0
Raciborz	37	0.243	0.162	0	97	0.144	0.021	0	6	0	0	0
Ruda Śląska	50	0.380	0.040	0	58	0.207	0.086	0	34	0.176	0.235	0
Rybnik	39	0.103	0	0	44	0	0.068	0	25	0.120	0	0
Sosnowiec	10	0.100	0.900	0	40	0.175	0.125	0	19	0	0.105	0
Tychy	-	-	-	-	119	0.059	0.067	0	3	0	0	0
Wodzisław Slaski	43	0.047	0.116	0.047	32	0	0	0	6	0	0	0
Zawiercie	27	0.296	0.037	0.037	16	0	0	0	11	0	0	0
Żywiec	-	-	-	-	-	-	-	-	-	-	-	-
Total	459	0.211	0.113	0.009	884	0.174	0.062	0	357	0.146	0.098	0.003

In the first year of application of the regulation (Polish law, 2007) in the scope of performance of water tests for Legionella spp., 802 water samples were tested in 24hHCC and 459 in MARB. The lack of facility register in which the quality of residential hot water system should be tested for waterborne diseases does not allow for evaluation of the effectiveness in performance of this obligation. On the basis of the results of the conducted analysis, it was stated that, despite the specifics resulting from the nature of 24hHCC, i.e. with consideration of temporary stay of persons with immunodeficiency, no control of water quality was conducted in the places of relevant venue of SDSI in Bielsko Biala, Myszkow and Zywiec. In addition, such tests were not conducted in MARBs in the areas of operation of SDSI in Chorzow, Czestochowa, Dabrowa Gornicza and Tychy. In relation to the data presented in table 1, ca. 450.000 residents could be exposed to potential waterborne diseases in case of stay in 24hHCC. In case of MARB, the size of potentially exposed population increased up to ca. 1.4 million residents. This data does not include, e.g. the number of tourists who stayed in these areas in the analysed period of time. Analysing the structure of occurrence of particular water microbiological contamination level, it was stated that average values in 24hHCC and MARB are on the similar level. The average probability of occurrence of particular contamination level in the Silesian Voivodship in 2008 was the following:

- mean in 24hHCC 0.243 and in MARB 0.211,
- high in 24hHCC 0.132 and in MARB 0.113,
- very high in 24hHCC 0.005 and in MARB 0.009.

Situations in which the presence Legionella spp. in cell number of over 10 000 [UFC/100ml] was stated were sporadic and concerned in total 6 areas of operation for both 24hHCC and MARB in 2008. For areas of operation of SDSI in Bytom, Dabrowa Gornicza and Gliwice, the value of probability of average contamination level in 24hCC in 2008 was higher than in whole Silesian Voivodeship. In relation to the area supervised by SDSI in Bytom, it is necessary to pay attention to high intensity of conducted control which is confirmed with the largest number of tested water samples. The three above areas are characterised with high urbanisation and industrialisation level, which impacts the number of 24hHCC. In turn, for MARB in 2008, the highest value of probability of occurrence of average contamination level was observed in the area of operation of SPSI in Ruda Śląska. In 2008, for 24hHCC facilities, the values of probability of occurrence of high contamination level at 0.4–0.5 for areas of operation of SDSI were stated in Ruda Slaska, Rybnik, Sosnowiec. The occurrence of average contamination level was also stated in all these cases of statement of high probability of occurrence of high contamination level. Such a sanitary condition of water quality in the tested facilities results in the possibility of exposure of the population living in a given area to waterborne diseases. In MARB, in 2008, the total value of probability of occurrence of lack of conformity with the microbiological quality standard in the area of operation of SDSI in Sosnowiec shows that each tested sample was a potential health threat.

In the next period of the analysed time period, the average probability of occurrence of contamination in the Silesian Voivodeship in 2010 was the following:

- mean in 24hHCC 0.182 and in MARB 0.174,
- high in 24hHCC 0.092 and in MARB 0.0062,
- very high in 24hHCC 0.01 and none occurrence was stated in MARB.

The first conclusion is the reduction of the number of water tested performed in 24hHCC by ca. 35% with their simultaneous increase for MARB by ca. 52% in 2010 compared to 2008. In 2010, no controls were conducted in MARB in the area of operation of SDSI in Zywiec. The highest probabilities of occurrence of average and high Legionella spp. contamination in MARB are observed in the areas of operation of SDSI in Chorzow and Dabrowa Gornicza, where did not conduct any corrective actions in 2008.

The mean values of occurrence of probability of a given contamination level in the entire Silesian Voivodeship are lower in 2010 compared to 2008. The only exception is the observation of very high contamination level in 24hHCC. The highest values of probability of occurrence of medium microbiological contamination level in 24hHCC were observed in the area of operation of SDSI in Bielsko Biała, Gliwice and Żywiec. The microbiological quality of water in the analysed scope for SDSI in Gliwice was subject to minor fluctuations. The percentage of high contamination level decreased with simultaneous increase of values of probability of medium contamination level. In turn, the values obtained in the areas of operation of SDSI in Bielsko Biala and Zywiec reflect the delayed performance of the obligation to conduct controls of water quality in terms of Legionella spp. Regarding the SDSI in Bielsko Biala, the total value of probability of occurrence of non-conformity with the standard is ca. 0.66. An unquestionable, expected effect of improvement in the microbiological condition of water quality is its quality in MARB in the area of operation of SDSI in Sosnowiec. The total value of probability of occurrence of non-conformity with the standard was reduced to ca. 0.47. In 24hHCC in the area of operation of SDSI in Czestochowa and Katowice (the total size of population is ca. 800.000), the reduction in the number of controls as well as the increase of the value of probability of occurrence of average contamination level were observed. It can prove that the facilities where the water quality did not meet the established requirements in the tested scope in 2008 were selected for control. In turn, in case of MARB, in 2010, in the area of operation of SDSI in Klobuck, the value of probability of average contamination level at 0.33 was observed in comparison with year 2008 when non-conformity with the standard was not stated.

Analysing the data from the subsequent period, i.e. year 2015, the decrease of the value of probability was observed with simultaneous minor fluctuations of the values of high and very high contamination level. In 2015, the average probability of occurrence of particular contamination level in the Silesian Voivodeship was the following:

- mean in 24hHCC 0,085a and in MARB 0.146,
- high in 24hHCC 0.01 and in MARB 0.098,
- very high in 24hHCC -0 and in MARB 0.003.

At the same time, the reduction in the number of tested samples for both types of analysed facilities is observed. A contentious issue is the lack of controls in the areas of operation of SDSI in: Bielsko Biala, Dabrowa Gornicza, Kłobuck, Myszków, Sosnowiec and Zywiec in 24hHCC and SDSI in: Bielsko Biala, Myszkow, Zywiec in MARB.

In the subsequent analysed periods of the study, the fluctuations of the microbiological condition of water quality were stated within the SDSI in Katowice in 24hHCC. One of the major 24hHCC in Katowice was subject to analysis with the purpose of reference of the analysed data set to a randomly selected facility. In 2008, none or very low contamination level was found in 17 out of 18 water samples and 1 sample presented mean microbiological contamination level. In 2010, 12 water samples were tested and none or low microbiological contamination level was stated for all of them. In 2015, water was not tested for Legionella spp. In 2017, all obtained results of 4 tested water samples confirmed the effective use of the facility's of both domestic hot water and plumbed water system.

The next group of facilities on which the obligation to conduct tests for Legionella spp. is imposed are swimming pools. Table 5 presents the analysis of the results of microbiological water quality test.

The analysis of microbiological water quality in the swimming pools allows for confirmation of the effectiveness of accurately defined obligations specifying the entity obliged to conduct the tests as well as accurately defined test frequency. The authorities of SSI keep the register of swimming pools, but only of those facilities which have been reported by their administrators to the control authority. In 2016, water samples were tested for Legionella spp. in 293 swimming pools. At that time, even though the performance of these tests was not obligatory, 2186 water tests were conducted. In comparison with the number of tests conducted in MARB in 2015, the number of tests conducted in swimming pools is 10 times higher. Only three samples confirmed a very high contamination level (the area of SDSI in Dabrowa Gornicza). The obtained values of probabilities of occurrence of a given contamination level in the swimming pools were lower by one order of magnitude compared to 24hHCC and MARB in both analysed time periods. Only in case of SDSI in Klobuck, the obligation to test water for Legionella spp. was not fulfilled in the required scope. The acquired data set of results of control water tests for Legionella spp. allows to state that the use of swimming pool should not lead to waterborne diseases caused by bacteria of Legionella spp. The experience gained by the administrators of 24hHCC and MARB as well as by the control authority has impact on the described microbiological condition in the swimming pools.

Table 5. Characteristics of water quality in terms of the occurrence of Legionella spp. in swimming pools in 2016 and in 2017 (*the first half of* the year)

State District Sanitary Inspectors in	**2016**				**01.01.2017–30.06.2017**			
	Probability of contamination							
	Samples number	**mean**	**high**	**very high**	**Samples number**	**mean**	**high**	**very high**
Bielsko Biała	109	0	0	0	131	0	0	0
Bytom	170	0.006	0	0	154	0.065	0.013	-
Chorzow	76	0.013	0	0	70	0	0.029	0.014
Cieszyn	156	0.006	0	0	342	0.018	0.009	0.003
Częstochowa	60	0	0.017	0	48	0.042	0	0
Dabrowa Gornicza	129	0.101	0	0.023	109	0.009	0	0
Gliwice	278	0.011	0.004	0	286	0.063	0.038	0.007
Jaworzno	22	0	0	0	31	0	0	0
Katowice	214	0.023	0.005	0	164	0.012	0	0
Kłobuck	8	0	0	0	-	-	-	-
Lubliniec	1	0	0.000	0	15	0.067	0	0
Myszkow	9	0.222	0.222	0	8	0	0	0
Raciborz	49	0	0	0	57	0.018	0	0
Ruda Śląska	88	0.023	0.034	0	108	0.009	0	0
Rybnik	140	0.079	0	0	85	0	0	0
Sosnowiec	61	0	0.016	0	22	0.045	0.045	0
Tychy	214	0.009	0.009	0	142	0	0	0
Wodzisław Slaski	124	0.016	0	0	108	0.028	0.009	0
Zawiercie	62	0	0	0	44	0	0.000	0
Żywiec	52	0	0	0	49	0	0.020	0
SSVSI	164	0.024	0.030	0	182	0.044	0.022	0
Total	2186	0.022	0.007	0.001	2155	0.025	0.012	0.002

It is necessary to highlight that the administrators of the swimming pool have more possibilities to ensure safe water as the swimming pools have their own water treatment plants. The administrators of 24hHCC and MARB facilities which use the collective water supply system have much fewer options of action in the scope of provision of appropriate water quality. It does not mean that they do not have any impact at all. The procedure specified in the regulations of the Minister of Health (Polish Law, 2007; 2015) defines the obligation to conduct a technical inspection of the network and/or its disinfection. Disinfection can consist in increasing the temperature to over 70°C, which leads to reduction of Legionella spp. If this method is ineffective, it is necessary to conduct chemical disinfection.

5. *Summary and Conclusions (formatting as all numbered headlines)*

The conducted analysis of the collected data of microbiological water quality in 24hHCC, MARB and swimming pools in a given research time horizon allows to propose a thesis that the applicable legislation must be improved. Firstly, it is necessary to oblige the control authorities to keep the register of facilities with residential hot water system in which the water quality must be controlled for Legionella spp. Secondly, it is necessary to precisely define the obligation of both for the owners/administrators of facilities and for the control authority in the scope of performance microbiological tests as well as precisely define their frequency. It also seems justified to standardise the procedures in case of statement of non-conformity with the quality standard. The last stage is introduction of the obligation to register and conduct the epidemic analysis in the scope of waterborne diseases.

The proposed in this study evaluation method can be used for preparation of procedure scenarios in case of occurrence of epidemics with the purpose of securing the number of available places in hospitals of infectious diseases or evaluation of the effectiveness of preparation and effectiveness of operation of competent services at a few levels of organisation of the state system, i.e. voivodeship, district, commune.

Acknowledgments

This work was supported by Ministry of Science and Higher Education Republic of Poland within statutory funds – project no BK-286/RIE-4/2017.

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