VODOHOSPODÁŘSKÉ TECHNICKO-EKONOMICKÉ INFORMACE
(WATER MANAGEMENT TECHNICAL AND ECONOMIC INFORMATION)

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60 years ago in VTEI

In VTEI No. 4 from 1964, Ing. Otakar Melzer, CSc., from the Department of Chemical Water Technology at UCT in Prague, described research into wastewater treatment from breweries and malt houses. His article was peer-reviewed by Ing. A. Nejedlý, CSc., from WRI in Prague.

The knowledge and results gained since 1949 by researching the quantity, quality, and treatment of wastewater from breweries and malt houses, contained in the reports of the Research Institute of Brewing and Malting in Prague, Water Research Institute in Prague, and the Research Institute of Královopolská strojírna in Brno, show the following:

1. *Processing 1 ton of barley requires 6.8 m^3, with 6.45 m^3 discharged as wastewater. Producing 1 hl of beer requires 1.2 m^3, with 0.6 m^3 of polluted water discharged.*

2. *After removing coarse solids using screens with 1 mm diameter holes, the concentration of the resulting wastewater mixture averages 800–1,100 mg/l O_2 BOD_5, or 300–550 mg/l of suspended solids, or 300–500 mg/l O_2 in a four-hour test, or 7–10 ml/l of sludge settled within two hours in an Imhoff cone.*

3. *The volume of wastewater varies significantly throughout the day, week, and year. According to Ing. Pospíšil, the coefficient of daily inflow irregularity is $k = 3.0$.*

4. *A considerable amount of glass shards, bottle caps, and sand is discharged from the facilities. To protect the treatment plant equipment, it is necessary to install a grit trap for these materials. The quantity of captured solids is approximately 15 l per 1,000 m^3 of wastewater.*

5. *If the facility uses wooden transport casks, the wastewater from their washing – both before and after pitch removal – must be discharged separately and the pitch must be removed prior to joint treatment.*

6. *To prevent unnecessary overloading of settling tanks with large quantities of coarse suspended solids, the water must be screened using screens with holes ranging from 1.0 to 1.4 mm. With 1 mm diameter holes, an average of approximately 150 l of such materials is captured per 1,000 m^3 of wastewater, with a maximum capture of up to 800 l per 1,000 m^3 of wastewater.*

7. *As the water contains a substantial amount of sludge, it must be subjected to sedimentation. The settled sludge exhibits significant cohesiveness; therefore, the use of Imhoff tanks is not suitable. Instead, tanks with scraped surface should be employed. If excess biological sludge is returned to the primary settling tanks, the surface must be continuously scraped. Settling tanks with an average retention time of 2 hours and 30 minutes will reduce pollution by approximately 25 % – as assessed by BOD_5, 25 % – as assessed by the four-hour test, 40 % – as assessed by the settleable solids in a two-hour sedimentation test in an Imhoff cone, or 20 % – as assessed by gravimetric determination of suspended solids. All data are derived from a model with a settling space depth of $h = 1.15$ m.*

8. *A biological tower with a final sedimentation tank, operated without recirculation, reduced the concentration of wastewater, measured by BOD_5, from 877 to 681 mg/l O_2, with a loading of 11.85 kg/m^3 · day and a height of $H = 4.15$ m. The specified loading is considerably high. In the spring months, the filter emits a slight odour. The efficiency, expressed as the percentage reduction in BOD_5, is only 22 %. Therefore, a biological tower does not yet appear to be the most suitable system for treating this wastewater.*

9. *A biological rapid filter with a final sedimentation tank reduced the concentration of wastewater, measured by BOD_5, from 547 to 423 mg/L O_2, which corresponds to a 52 % reduction based on the raw water concentration; the volumetric loading was 8.11 kg/m^3 · day O_2 BOD_5, and the average recirculation ratio was $m = 3.45$. In the spring months, the filter also emitted a slight odour. With a significantly lower volumetric loading, the rapid filter will need to be re-tested.*

10. *The activation tank with a final sedimentation tank reduced the wastewater concentration measured by BOD_5 from 888 to 325 mg/l O_2 at a loading of 1.98 kg/m^3 · day and a retention time of approximately 4 hours and 30 minutes. This loading is somewhat higher than can be used in practice because, although the efficiency in terms of % reduction of BOD_5 is 63 %, the resulting activated sludge is poorly sedimented and easily flushed. Therefore, the final sedimentation tank must also be constructed with a scraped surface. Actual tanks will probably have to have significantly reduced volumetric loading and longer retention times.*

These results and findings were obtained at the experimental treatment plant in the Velké Popovice brewery. The treatment plant consisted of a grit, glass and cork chamber, perforated scraped troughs, a pumping station, and a device dividing the water into six equal parts. The water was further treated by shallow activation with a final sedimentation tank, on a biological tower with a final sedimentation tank and on a biological rapid filter with a final sedimentation tank and recirculation from the bottom of this tank. Each of the three biological treatment methods was preceded by its own Imhoff tank. The other three wastewater parts are dedicated to the natural treatment methods. The biological treatment methods with final sedimentation tanks were originally designed for an average flow rate of 0.2 l/sec at each plant. All results shown for the biological treatment methods so far are from a two-month period. They cannot therefore be used as definitive criteria. According to existing results, chemical coagulation or a second stage of biological treatment will have to be added where high demands are placed on the purity of the discharged wastewater.

From TGM WRI archives

VTEI Editorial Office

Content


Photo: J. Unucka

Dear readers,

the June issue of our professional journal VTEI is being published at a time when the whole world is commemorating World Oceans Day (8th June). This celebration, declared in 1992 at the UN Conference on Environment and Development in Rio de Janeiro, aims to remind people of the importance of the oceans and their life-sustaining role. For the second year in a row, the main theme of this celebration is 'Catalysing Action for Our Ocean & Climate', which responds to the global climate and biodiversity crisis. It is based on the realization that a healthy ocean is essential for a stable climate and calls on world leaders and corporations to honour the commitments arising from the Global Ocean Treaty and the 30 x 30 action plan, which aim, among other things, to move towards a more sustainable society and to protect 30 % of the world's oceans by 2030. Protection of the oceans begins with the protection of water resources, and here, each of us can make a difference.

The content of the VTEI June issue brings you a diverse selection of expert and informative articles. The first of the expert articles is "Wetland evapotranspiration". The authors, Karel Pátek and Jiří Bruthans, summarise not only the current state of knowledge but also present interesting research findings on evapotranspiration impact on wetlands in the western part of the Czech Cretaceous Basin.

Monitoring hazardous organic pollutants and heavy metals, substances that accumulate in both biotic and abiotic components and food chains, provides crucial information about environmental pollution. The expert article by Hedvika Roztočilová and her colleagues on monitoring xenobiotic substances in solid matrices of aquatic ecosystems presents the results of monitoring carried out by the Czech Hydrometeorological Institute (CHMI), highlighting the long-term burden of surface waters in the Czech Republic with these substances.

Grass strips in agricultural landscapes are generally considered an effective method for reducing surface runoff and preventing the transport of eroded particles further down the slope. The third expert article by Tomáš Laburda and his colleagues "The potential of grass strips for retaining surface water runoff and sediment", focuses on quantifying the effect of grass strip length on the capture of sediment transported by surface runoff on experimental plots.

It is a well-known fact that agriculture is the largest freshwater consumer in the world, accounting for up to 70 % of total water resource consumption. The expert article by Dagmar Vološinová, Libor Ansorge, and Lada Stejskalová, "Grey water footprint of malting barley production", explores the concept of the grey water footprint as an environmental indicator which helps assess the impact of agricultural production on water resources. The article also examines different approaches to including biologically active substances in grey water footprint models.

The informative section of the June VTEI issue cannot be complete without an interview. Ivan Tučník, Head of Group Sustainability Asahi Europe & International, which owns breweries such as Radegast, Plzeňský Prazdroj, and Velkopopovický Kozel in the Czech Republic, accepted our invitation. The interview focused not only on water consumption in beer brewing and water management in general, but also on research into the cultivation of crops essential for brewing beer, water 'neutrality,' and the development of beer culture in our country. Following the interview is an informative article about projects implemented by Radegast Brewery focused on water retention in the landscape.

On 27th March 2025, our colleague Ladislav Kašpárek suddenly and unexpectedly passed away, without a chance to say goodbye. He had been a key figure at our institution for many years. This happened just a few days after the launch of the book *Historical Floods on the Rakovnický Stream*, of which he was a co-author. Therefore, the final pages of the June issue are dedicated to the memory of this outstanding expert and friend.

Dear readers, we wish you enjoyable reading and a good start to the summer.

VTEI Editorial Office

Wetland evapotranspiration

KAREL PÁTEK, JIŘÍ BRUTHANS

Keywords: evapotranspiration — wetland — water balance — groundwater level fluctuation — water retention in the landscape

ABSTRACT

A wetland is an environment where water is readily available for vegetation, and therefore intensive evapotranspiration (ET) close to the potential ET value occurs. In addition, higher ET intensities can be expected in the future due to the observed increase in temperatures associated with climate change. The impact of wetland ET needs to be considered, for example, in restoration planning or hydrological modelling, and it is important to draw on the current knowledge provided by the large number of papers worldwide. Therefore, the first part of the paper is a brief review of existing research on wetland ET. The second part of the paper is a practical demonstration of the impact of ET on wetlands in the western part of the Bohemian Cretaceous Basin.

In the first (review) part of the paper, the articles were divided into several groups according to whether they were based on investigations of groundwater level (GWL) fluctuations, monitoring of wetland-influenced streamflow, tree transpiration measurements, or a combination of different methods. Thus, we can see where current research has moved since the original observations of GWL fluctuations, for example in White's 1932 paper [1]. In the second (practical) part of the paper, GWL fluctuations were monitored in a wetland in the upper part of the Liběchovka catchment (moderate climate, western part of the Bohemian Cretaceous Basin) in the summer of 2024. Four piezometers representing different parts of the wetland were installed in the wetland. From the measured data, periods in which significant diurnal GWL fluctuations occurred for several days were selected; these were 8 periods of 3 to 14 days. Fluctuations were evident in all parts of the wetland surveyed, with GWL maxima and minima occurring at similar times in different parts of the wetland; only the amplitudes of the fluctuations differed. Diurnal GWL fluctuation was most evident in the central part of the wetland (amplitude up to 14.5 cm in peak summer). As additional information on wetland conditions, soil moisture was measured at different depths in autumn and summer. It was observed that the soil profile changes between different locations, even several meters apart, due to the dynamic action of the flowing stream. The moisture content of sandy layers (around 40 %) differed significantly from that of clay-loam layers (where it was mostly between 70 and 80 %). Soil in the wetland was also found to be very close to saturated throughout the profile, both in autumn, when no significant ET takes place, and during summer. This means that the small amount of water added/consumed is sufficient to cause significant vertical movement of GWL and therefore diurnal variation of GWL may be more apparent. For the 8 selected periods in which significant diurnal GWL fluctuations occurred for several days, the average ET of the monitored wetland in the upper catchment of the Liběchovka stream was calculated as 20 $l \cdot s^{-1} \cdot km^{-2}$ by White's method [1] based on GWL fluctuation. The two parts of the paper together show that the topic of wetland ET is important and relevant and demonstrate that wetlands need to be seen as environments where water is intensively used by vegetation.

INTRODUCTION

Evapotranspiration (ET) is generally an important component of the water cycle. A wetland, as a waterlogged area with a groundwater level (GWL) close to the surface, provides an environment where water is readily available to vegetation. Actual ET is therefore very high and approaches potential evapotranspiration (PET), which is the maximum possible rate limited only by the amount of energy available for evaporation.

While primary ET occurs in the soil, in wetlands, emerging groundwater or incoming surface water evaporates that has already undergone primary ET. The influence of wetlands on GWL and streamflow is therefore particularly significant during dry periods, when there is a general lack of water in the landscape, but water remains available in the wetland due to groundwater inflow, allowing intensive ET to continue.

It is therefore necessary to consider the influence of ET not only in planning restoration projects, calculating water balance, and in hydrological models, but it should also be considered in legislation – for example, in relation to the concept of minimum residual flow or minimum groundwater level. According to current legislation, artificial water abstraction is restricted during dry periods in order to maintain the minimum residual flow. However, under high temperatures, water consumption through ET may become so high that it will not be possible to maintain the minimum residual flow (or minimum GWL), even if all other artificial water abstraction is halted, because wetland and riparian vegetation is capable of evaporating groundwater near the watercourse [2]. This issue is becoming particularly relevant in the context of ongoing climate change, which is leading to rising temperatures and, consequently, an even more pronounced influence of ET.

The aim of this paper is to summarise the main research conducted to date in the field of wetland ET and to demonstrate that wetland ET is an important topic that must be taken into account in Central Europe, along with the latest findings. The first, theoretical part presents a review of articles focused on the influence of ET in wetlands. The second part provides a practical demonstration by monitoring the influence of ET on fluctuations in GWL and streamflow of a small watercourse in a wetland in the upper catchment of the Liběchovka.

Brief review of previous research

Wetland ET was intensively studied as early as the last century, yet it remains a current and actively researched topic today. Published articles can be divided into several groups, which are discussed in more detail in the following paragraphs. The first group of articles examines in detail the influence of ET on GWL fluctuations. Another group focuses on the relationship between ET and streamflow. A different group combines observed GWL fluctuations with fluctuations in the flow of a stream running through the wetland. A further group of articles addresses the issue in a highly comprehensive manner, linking the influence of ET, GWL fluctuations, and streamflow observations with measurements of vegetation transpiration.

Influence of ET on GWL fluctuations

The relationship between wetland ET and GWL fluctuations was studied by White as early as 1932 [1]. He observed that, due to ET, GWL is lower during the day than at night. This results in a regular within-day GWL fluctuation (diurnal fluctuation). A typical pattern of diurnal fluctuation is well described, for example, in articles by Gribovszki from 2006 [3] and 2008 [4], and is illustrated in *Fig. 1*.

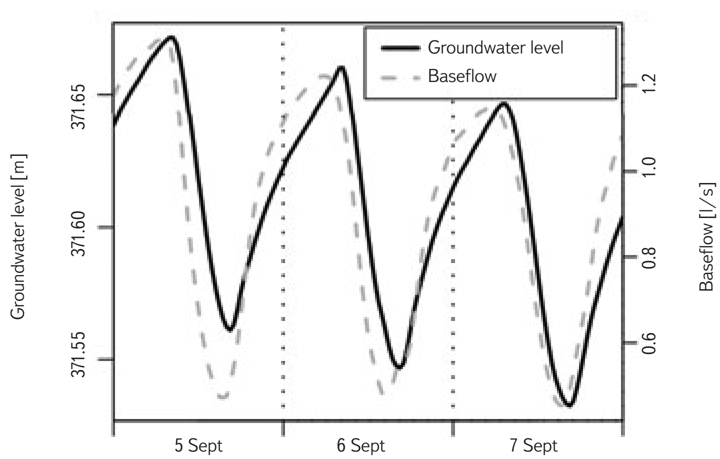

Fig. 1. Typical diurnal fluctuation of groundwater level and baseflow measured in the experimental catchment at the foot of the Alps (modified after [4])

White [1] developed a method to estimate the magnitude of ET on a daily scale based on diurnal fluctuations. The principle of the method is illustrated in *Fig. 2*. The vertical axis shows the GWL, and the horizontal axis represents time. For a given day, Grec presents a linear extrapolation of expected GWL development in the absence of ET-induced decline. The extrapolation is based on water level development during the night between midnight and 4 a.m., as ET can be considered negligible during this part of the day. ΔS represents the change in groundwater storage during the day (i.e., the change in water level between midnight of the current and previous day). Subsequently, ET is calculated as the sum of Grec and ΔS, multiplied by the specific yield value.

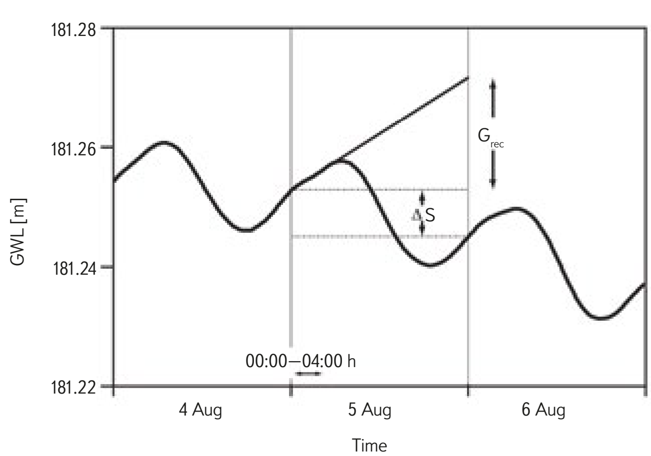

Fig. 2. ET calculation using the White method [1] (modified after [5])

The limitations and behaviour of White's method were studied in detail, for example, using numerical simulations in Loheide's 2005 article [6]. The most significant source of error was identified as uncertainty in determining specific yield. A specific yield of 2 % means that if the water level in rock environment drops by 1 m, the amount of water removed from the rock environment is equivalent to removing a 2cm water column from a water volume (e.g., a water tank).

Some authors have proposed alternative methods for estimating ET from GWL fluctuations. In 2008, Gribovszki [4] developed a modification of White's original method to improve the accuracy of the calculation. Another method was offered by Loheide in an article also published in 2008 [7]. Carlson Mazur, in 2014 [5], subsequently created a modification allowing its application across a wider range of natural conditions. Various methods for calculating ET from GWL fluctuations are discussed in the article by Fahle and Dietrich from 2014 [8]. The values were compared against reference measurements using a lysimeter; the highest correlation with the reference value was obtained using Gribovszki's method [4].

A significant advance was also made by Malama in 2010 [9]. He derived an analytical solution describing how ET (specified as an input "control function") manifests in GWL fluctuations (*Fig. 3*). Malama's solution [9] also explains the several-hour delay of GWL maxima and minima relative to the solar cycle, which had previously been observed in the field, for example, by Gribovszki [4]. The derived analytical solution can be applied in two ways: to model ET from measured diurnal GWL fluctuations given knowledge of the hydraulic parameters, or, inversely, to infer hydraulic parameters of the environment from diurnal GWL fluctuations given knowledge of ET. In Malama's article, the developed model was applied to measured diurnal GWL fluctuations, yielding information on ET, hydraulic conductivity, and river stage changes (*Fig. 3*). This demonstrated that, in principle, both ET and hydraulic parameters can be derived simultaneously from diurnal GWL fluctuations.

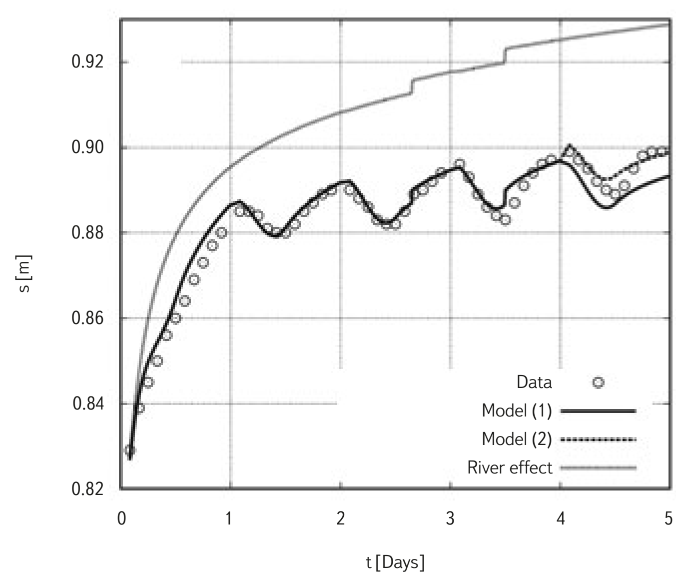

Fig. 3. Model fit of the measured values of diurnal groundwater fluctuation in the vicinity of the river. Model (1) uses the same ET amplitude for the 4th and 5th day, whereas model (2) takes into account the different ET amplitude for the 4th and 5th day. The dotted line represents the effect of fluctuating river stage

Relationship between ET and streamflow

The following group of articles focuses directly on the relationship between ET and streamflow fluctuations. Sometimes this relationship was assessed on a regional scale; an example is 1976 Daniel's article [10]. This paper addressed the relationship between ET from the aquifer and streamflow in a nearby watercourse, and described an analytical solution that can be used in rainfall-runoff models and GWL models. Another example is Wittenberg's 1999 article [11]. It determined hydraulic parameters from curves describing the long-term annual decline in streamflow, while observing how recession curves are influenced by ET.

Other articles approached the relationship between ET and streamflow at the catchment scale. An example is Zecharias [12] in 1988 which used a conceptual model to describe the relationship between runoff and aquifer. A significant advance was later made by Fonley [13], who in 2019 derived an analytical relationship for back-calculating ET from records of diurnal streamflow fluctuations (*Fig. 4*).

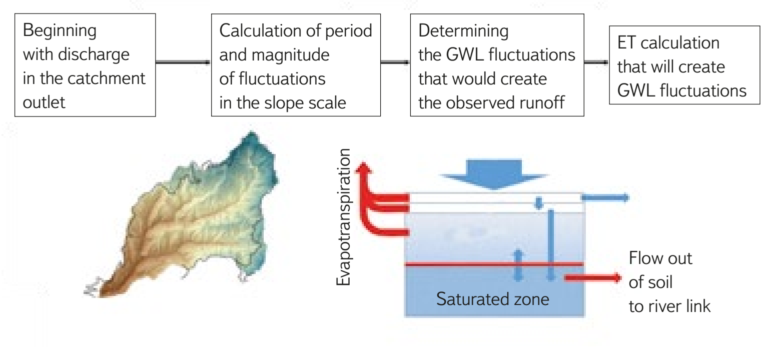

Fig 4. Schematic overview of the Fonley method of calculating ET from diurnal fluctuation of river flow (modified after [13])

ET and fluctuations of GWL combined with streamflow fluctuations

Another group of articles addresses GWL fluctuations caused by ET, while also including observations of the stream flowing through the wetland. Partially, the previously mentioned article by Gribovszki [4] could be included here, whose main contribution was improving methods for calculating ET from GWL fluctuations, but also addressing the relationship with stream level fluctuations. Another example is Yeh [14], whose 2008 article examined the regional-scale soil water balance using 19 years of monthly runoff observations, GWL, and soil moisture. In addition, changes were monitored in the thickness of the subsurface layer influenced by ET.

The relationship between soil moisture and streamflow fluctuations was examined in detail by Moore's 2011 study [15], conducted in the HJ Andrews experimental forest in western Oregon. He reached the interesting conclusion that, across all time scales, soil moisture correlates very well with the amount of water currently flowing in the stream. The correlation was strongest at high soil moisture levels.

ET, GWL fluctuations, and streamflow combined with independent measurements of vegetation transpiration

The most comprehensive studies of wetland ET are found in articles that monitor not only streamflow and possibly GWL fluctuations but also independently measure vegetation transpiration, for example through sap flow measurements in trees. These articles can be divided into two groups based on location. One group originates from the semi-arid climate of Arizona, the other from the Mediterranean climate of Oregon.

The first group consists of articles from Arizona. They form part of the extensive international SALSA (Semi-Arid Land–Surface–Atmosphere) programme, which focused on human-induced environmental changes in semi-arid regions [16]. ET from waterlogged areas along watercourses (riparian zones) represents an important component of the water balance and was therefore intensively studied within the project using various methods [17]. For example, transpiration of willows and poplars was measured using the sap flow method, and the results were compared with ET values indirectly derived from the water balance [18]. Canopy transpiration derived from sap flow measurements corresponded to values obtained using Raman lidar and was used to calibrate coefficients in the Penman–Monteith method for calculating ET. ET of grasses was also determined using the Bowen ratio method. A summary of these results, together with an estimate of the uncertainty in determining water balance components and a comparison with values obtained using a model, is discussed in detail in the paper by Goodrich (2000) [17].

The second group comprises studies from Oregon (HJ Andrews experimental forest). Important insights are provided in the paper by Bond (2002) [19], where streamflow was measured and correlated with sap flow data from the surrounding vegetation. The measurements were used to estimate the volume of water consumed by ET and to determine the width of the riparian zone from which ET occurred. The study also described in detail the times of year when diurnal fluctuations in streamflow caused by ET were observed.

An interesting contribution from Oregon is the 2007 paper by Wondzell [20], which examined in detail the delay between the daily ET cycle and streamflow fluctuations. It was found that the delay depends on streamflow velocity, more precisely on the speed at which the hydraulic pulse propagates through the channel. When streamflow was fast, the influence of vegetation from different parts of the wetland combined constructively, amplifying the original diurnal signal. In contrast, when streamflow was slow, the influence of vegetation from different parts of the wetland combined destructively (i.e. cancelled each other out), and the original diurnal signal was dampened. Wondzell [20] also mentions diurnal fluctuations in chemical indicators and suggests that they could be subject to a similar delay depending on streamflow velocity.

Another contribution from Oregon is the article by Barnard from 2010 [21]. The aim was to describe the relationship between transpiration and subsurface runoff. To this end, subsurface runoff from the soil, soil moisture, and tree transpiration were measured on a selected small plot. Artificial irrigation was also carried out, and the resulting development of changes was monitored. It was found that the delay of diurnal fluctuation relative to ET depends on soil moisture. In this case, however, Barnard [21] considers it unlikely that the signal from different parts of the catchment could synchronise in such a way that diurnal fluctuations would be observable.

An interesting article from Oregon is also that of Graham from 2013 [22], discussing the frequency of diurnal fluctuations caused by ET. Fifteen different catchments were monitored, and the results were compared with sap flow measurements. Diurnal fluctuations in streamflow caused by ET were observed in all years and across all fifteen catchments, suggesting that this is not merely a special phenomenon limited to a small number of sites.

Review summary

ET in wetland environments remains a highly researched topic today, leading to significant advances in understanding over the past 20 years. GWL diurnal fluctuations caused by ET have been shown to occur across many catchments, as demonstrated by Graham in 2013 [22]. Modifications of the original classic White method have been developed for calculating ET from GWL fluctuations, such as Gribovszki's method from 2008 [4]. Since the original observations and descriptions, research has progressed to modelling and deriving analytical relationships. An example in the relationship between ET and GWL fluctuations is the article by Malama from 2010 [9], and for the relationship between ET and streamflow fluctuations, the article published by Fonley in 2019 [13]. In the literature, there are also comprehensive studies combining observations of diurnal fluctuations with independent ET determinations by other methods, for example through sap flow measurements in trees. The main groups of articles adopting such a comprehensive approach come from the semi-arid climate of Arizona and the Mediterranean climate of Oregon.

Generally, the current state can be summarised in the following three points. Firstly, since the original observations of changes in water level or flow in wetlands, there has been a tremendous advance in knowledge. Secondly, despite this great progress, it remains necessary to conduct measurements in various locations. These measurements, employing ever-evolving technology, will help verify or supplement hydrological models [13]. The third and very important point is the need to transfer these already acquired insights into the general awareness of experts engaged in practical activities related to hydrology and ecology, who can then apply the new knowledge in everyday practice. This is an important step building on the previous research, which is essential to ensure that current findings are considered in everyday practice. An example of such an effort is this article on wetland ET, which combines a literature review with measurements of GWL fluctuations in a long-term studied wetland in the upper Liběchovka catchment.

Measurement of ET influence in the wetland on the Liběchovka

Following the literature review, a practical demonstration of the influence of ET on a wetland in the western Czech Cretaceous Basin was carried out. GWL fluctuations and runoff from a small wetland were monitored on a minor tributary in the upper catchment of the Liběchovka. It was demonstrated that even under these conditions, diurnal GWL fluctuations caused by ET can be observed. The measurements also provide information on what can be easily used for such monitoring, which conditions must be met, and how this diurnal fluctuation appears in practice.

LOCATION

The measurements were conducted in a wetland in the upper part of the Liběchovka catchment (*Fig. 5*). It is part of hydrological region 4522 Cretaceous of the Liběchovka and Pšovka. Average annual temperature (calculated based on the period 1981–2010) reaches 8.2 °C, and average annual precipitation totals 595 mm [23].

The site was chosen so that local conditions would not hinder the manifestation of ET. A small watercourse (Soví stream) flows through the wetland, whose main source of water bearing is groundwater from quaternary sandstones; rapid runoff is negligible. Therefore, the watercourse has a stable flow, allowing the impact of water consumption by ET to be clearly observed. The area of the wetland was determined through field survey to be 19,000 m^2 [24].

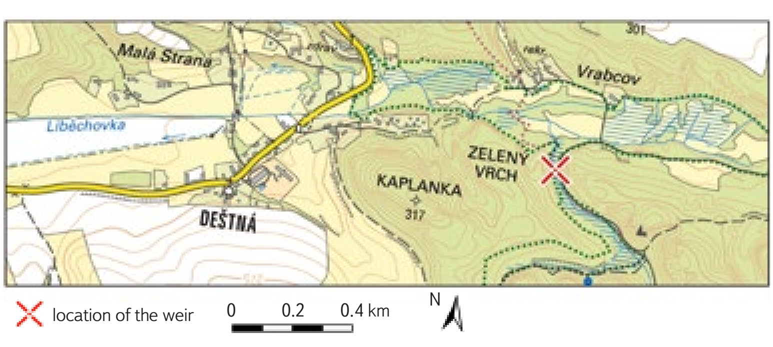

Fig. 5. Wetland area ([24])

METODOLOGY

As a practical demonstration of the influence of ET in the wetland, GWL and soil moisture were monitored. GWL was measured using piezometers, and soil moisture was determined by collecting soil samples.

GWL measurements

This current phase followed up on earlier measurements by the same authors [25] from the second half of summer 2022, which demonstrated that diurnal GWL fluctuations in the monitored wetland are caused by ET and occur on warm, rain-free days. The new study covered the entire summer period, with piezometers installed in different parts of the wetland, enabling observation of the gradual development of diurnal GWL fluctuations from the spring and summer of 2024.

Measurements from the beginning of 2024 until 9 September 2024 were used for the analysis. The location of the newly installed piezometers, which capture GWL spatial variability in the wetland, is shown in *Fig. 6*. The piezometer referred to as LI2 monitors the conditions in the middle of the lower part of the wetland, close to the constructed weir. The central area of the wetland is represented by the piezometer called Střední mokřad ("Central Wetland"). The piezometer called Boční mokřad ("Lateral Wetland") describes the edge of the wetland, where the terrain is 2 metres above the valley floor. However, based on the vegetation characteristics, this area is still considered a wetland with GWL close to the surface, which indicates that groundwater flows into the wetland from this side. This is also consistent with the clay layer found at a depth of about 60 cm below the surface, which may form a low-permeability layer along which water flows. Another inflow area to the wetland is represented by the piezometer designated U Studánky ("The Spring"). It was installed at the upper edge of the wetland, near the spring of one of the branches of the watercourse flowing through the wetland.

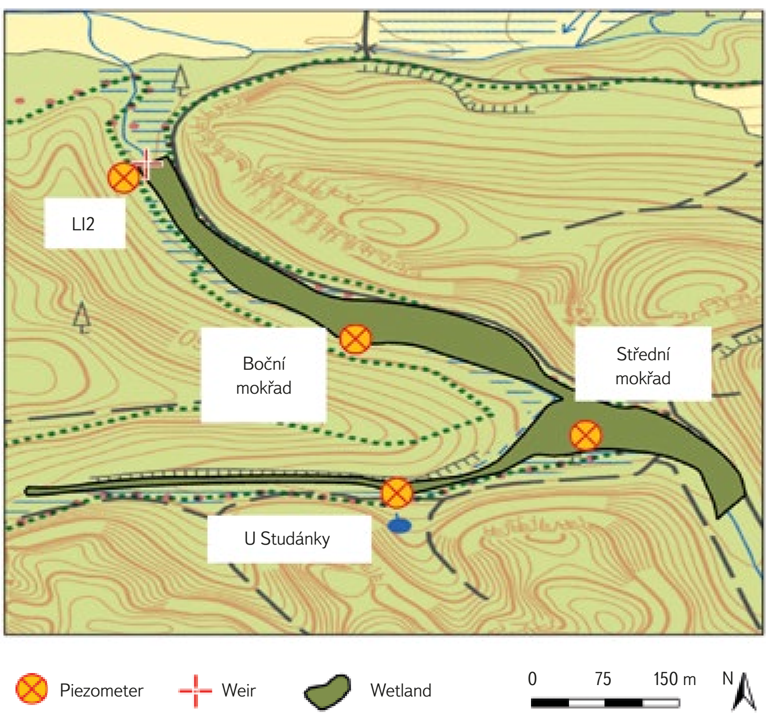

Fig. 6. Location of piezometers in the wetland; for a general overview, the location of the weir constructed by the same author during previous work ([25]) is also shown (background map: Base map 1 : 10,000 from [26])

The construction of the piezometers is described in detail in [24].They consisted of pipes with a perforated lower section, in which the GWL was measured using Solinst Logger pressure sensors. To convert the measured pressure to water column height, it was first necessary to subtract the atmospheric pressure from the measured values. Atmospheric pressure was monitored using a Solinst Logger pressure sensor located in the same area as the wetland. For measurement verification, the GWL was also manually measured with a water level meter at the beginning and end of the measuring period. The working names of the piezometers and the depth of their bases below the surface are shown in *Tab. 1*.

Tab. 1. Depth of the piezometer

Piezometer	Base depth below ground level [m]
LI2	0.92
Boční mokřad	1.2
Střední mokřad	0.85
U Studánky	1.16

Soil moisture measurement

Soil moisture was measured around all installed piezometers, thus covering different parts of the wetland. To avoid disturbance of the soil immediately surrounding the piezometer tube, sampling points were selected approximately 1.5 m from each piezometer. The first sampling took place in autumn 2023 (12–13 November 2023). This was a cool and wet period, during which it was expected that the soil profile would retain an above-average amount of water and the GWL would be high. The second sampling was conducted during hot summer days in a rain-free period (11th and 12th August 2024), when it was expected that the soil profile would contain a below-average amount of water and the GWL would be low.

First, a profile was excavated to a depth where water began to visibly seep from the walls, and the characteristics of the individual layers were roughly described. Subsequently, samples were taken. The sampling depths were adjusted to ensure that prominent soil layers were captured. During the second sampling, it was necessary to select a location near the piezometer that had not been disturbed during the previous sampling; therefore, the profiles for the same piezometer slightly differ between the first and second samplings. Kopecký cylinders with a volume of 100 cm³ were used for sampling, driven in horizontally. Within each profile, one sample was taken from each observed layer to determine specific yield. Immediately after sampling, the cylinders were sealed with lids, wrapped in foil, and placed in plastic bags to prevent moisture loss. As soon as the samples returned from the field, they were weighed on scales with an accuracy of 0.1 g. Subsequent weighing was conducted on the same scales.

The first step in sample processing was to determine saturated moisture content. Cylinders containing the samples were placed individually in a container, filled with water from the bottom, and left submerged for three days. They were then removed from the container and weighed. In two exceptional cases, a non-negligible loss of material occurred from the sample, which distorted the moisture content in the saturated state (a note has been added in the resulting graphs for the respective samples). The second step was to measure the weight of the dried soil. First, the sample was dried for several days at room temperature, then for 1.5 weeks in an oven at 105 °C, and finally it was weighed immediately after removal from the oven to prevent it from absorbing atmospheric moisture.

From the measured data, the volumetric moisture content of the sample at the time of collection (θ_{odb}) and the saturated (volumetric) moisture content of the sample (θ_{sat}) were calculated as follows:

$$\theta_{odb} = \frac{V_{v\,odb}}{V_{vzorek}} = \frac{m_{odb} - m_s}{\rho_v \cdot V_{vzorek}} \quad (1)$$

$$\theta_{sat} = \frac{V_{v\,sat}}{V_{vzorek}} = \frac{m_{sat} - m_s}{\rho_v \cdot V_{vzorek}}, \quad (2)$$

where:

$V_{v\,odb}$	is	the volume of water in the sample at the time of collection
$V_{v\,sat}$		the volume of water in the sample in the saturated state
V_{vzorek}		the volume of the collected sample (100 cm³)
m_{odb}		the weight of the sample at the time of collection
m_s		the weight of the dried sample
m_{sat}		the weight of the sample in the saturated state
ρ_v		the density of water (1,000 kg · m^{-3})

Subsequently, the resulting moisture values were converted from decimal numbers to percentages.

Estimation of ET from GWL fluctuations

In the first step, the specific yield of the soil layers in which diurnal GWL fluctuations occur was estimated based on soil moisture measurements. This allowed the measured GWL fluctuations to be converted into the amount of water gained or lost from the groundwater during level changes in subsequent data processing steps. For the estimation of specific yield, only those piezometers were selected where, during soil profile sampling, few heterogeneous layers were detected at depths around the GWL.

The calculation of ET from diurnal GWL fluctuations was performed in a simplified manner based on the original classical method by White [1]. The principle of the method is summarised in *Fig. 2* in the review section of this article. First, the average hourly change in GWL (r) was calculated based on the interval between midnight and 4 a.m. This value reflects the long-term GWL trend without the influence of daytime ET. Subsequently, the daily change in soil water storage (ΔZ) was calculated as:

$$\Delta Z = h_1 - h_2, \quad (3)$$

where:

h_1	is	the GWL at midnight of the current day
h_2		the GWL at midnight of the following day

A positive value of ΔZ indicates a drop in GWL during the day, meaning a loss of water from the environment.

In the next step, ET was calculated and expressed as the height of the water column per day:

$$ET = S \cdot (24 \cdot r + \Delta Z), \quad (4)$$

where:

S	is	the specific yield
r		average change in GWL without the ET influence
ΔZ		zchange in storage over a day

The obtained ET value was converted from the water column form to a flow rate form (volume of water consumed per unit of time). The resulting wetland ET value was calculated as the arithmetic mean of the individual values from each piezometer. This value was then compared with ET estimates based on streamflow fluctuations and the Oudin method for calculating PET, which had been conducted by the same authors on earlier data from the same site [25].

The average ET value calculated from the piezometers using White's method was designated as $ET_{prům}$. However, some piezometers were excluded from the ET calculation due to the presence of many different soil layers. Retrospective estimates were made for these piezometers to determine the specific yield required for the ET value calculated by the White method from GWL fluctuation to equal $ET_{prům}$. This calculation was performed using the Solver function in MS Excel.

RESULTS

GWL measurement

From the measured data, periods were selected during which significant diurnal GWL fluctuations occurred over several days (*Tab. 2*). There were eight such periods, with the duration of individual episodes ranging from three to fourteen days, and fluctuations were evident at all measured locations within the wetland. During these periods, the water level of the small watercourse at the weir also fluctuated. However, the amplitude of stream level fluctuations was significantly smaller – up to a maximum of 2.5 cm in peak summer – compared to GWL fluctuations, which reached up to 14.5 cm in peak summer at Střední mokřad piezometer.

Tab. 2. Periods with diurnal fluctuation in groundwater level in the wetland (2024)

From	To	Number of days
09. 05.	16. 05.	8
08. 06.	12. 06.	5
16. 06.	18. 06.	3
24. 06.	30. 06.	7
07. 07.	10. 07.	4
15. 07.	26. 07.	12
28. 07.	01. 08.	5
10. 08.	15. 08.	6
26. 08.	08. 09.	14

The first significant signs of diurnal GWL fluctuations were observed at the beginning of May (A in *Fig. 7*). During the eight-day period from 9 to 16 May, fluctuations were greater in the central part of the wetland (Střední mokřad piezometer) compared to fluctuations at other locations within the wetland. Another period with clearly visible GWL fluctuations was the first half of June, specifically from 8 to 12 June (B in *Fig. 7*) and from 16 to 18 June. The amplitude of fluctuations was more pronounced compared to May, especially in the case of Střední mokřad piezometer. An exception was U Studánky piezometer, where fluctuations were smaller compared to the other piezometers.

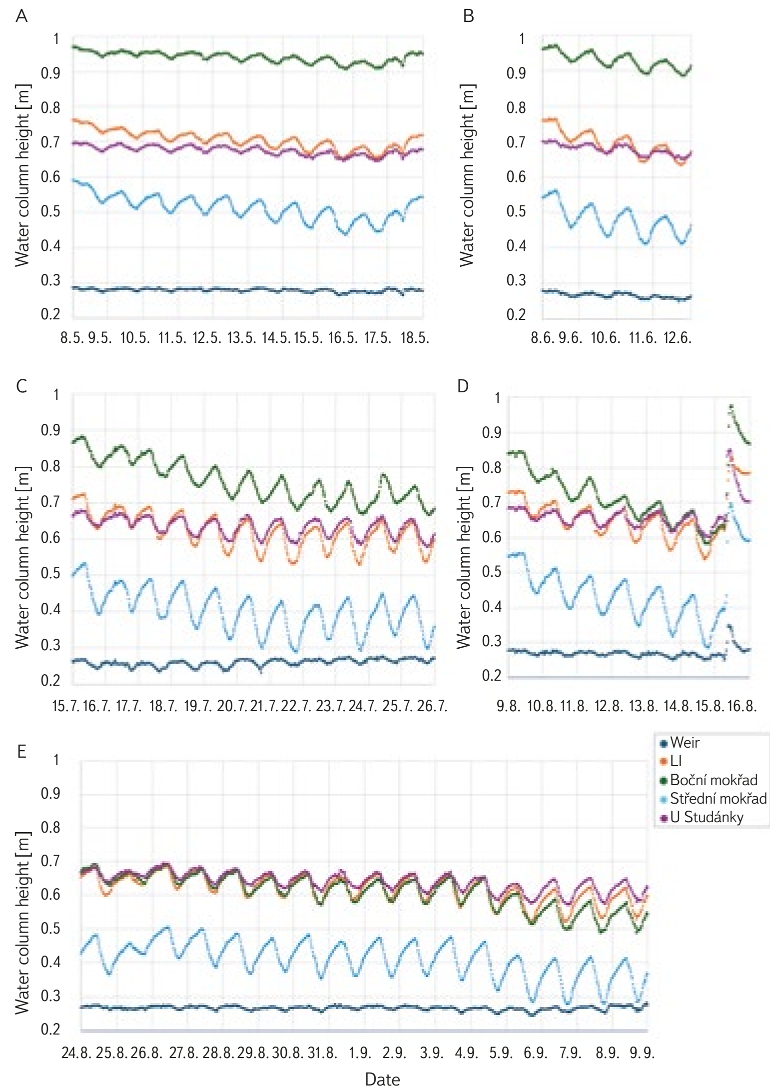

Fig. 7. Diurnal groundwater level fluctuation in early spring (A), late spring (B), mid-summer (C, D), and at the end of summer (E)

From the end of July, marked diurnal fluctuations in water level were observed in all piezometers (C and D in *Fig. 7*). These occurred during the periods from 24 to 30 June, 7 to 10 July, 15 to 26 July, 28 July to 1 August, and 10 to 15 August. However, in the case of the U Studánky piezometer, the amplitude of the fluctuations remained lower. The longest continuous period of diurnal GWL fluctuation occurred over 14 days at the end of summer, from 26 August to 8 September (E in *Fig. 7*).

Soil moisture measurements

The soil moisture measurement results are presented in *Tab. 3* and summarised in *Fig. 8*. The moisture content of sandy layers during sampling was around 40 %, while the moisture content of clayey-silty layers was significantly higher, generally between 70 and 80 %.

Soil profiles sampled in autumn and summer at the same piezometer were only a few metres apart, yet the layer composition often differed significantly. This indicates pronounced variation between soil profiles at different places within the wetland, which is a consequence of the watercourse dynamic activity. Simultaneously, this generally implies that it is difficult to determine representative hydraulic parameters for such environments, for example for the purposes of accurate modelling.

During autumn sampling, GWL was higher than in summer. In both periods, however, soil throughout the entire profile was very close to saturation. The difference between moisture content at the time of sampling and saturated moisture content was greatest in the surface layers, but even in this case it did not exceed 4 %.

The measurement error in determining soil moisture was estimated to range between 1 and 2 %. This is based on the fact that, in some samples, the measured moisture content at the time of sampling exceeded the saturated moisture content, or that layers below the GWL were not saturated. A difference value of 0 between saturated and sampled moisture content indicates that the sample's moisture content at the time of sampling was equal to or greater than the saturated moisture content.

In one case, it was found that a sample taken well below the GWL was 5 % below saturation; in another, the moisture content of the sample significantly exceeded the saturated moisture content (by 5 %). These values were considered to be processing errors caused by partial water loss from the sandy sample during handling prior to weighing. In the case of a third sample, it was found that just below the GWL the sample was 3 % below saturation. This was considered to be the result of GWL recorded from the nearby piezometer not exactly matching the GWL at the location of the sampled profile, and the respective sample was in fact taken from above the water level.

Tab. 3. Results of soil moisture measurements

Autumn				Summer				
Piezometer	**Depth below ground level [cm]**	**Moisture content at the time of sampling [%]**	**Saturated moisture content [%]**	**Piezometer**	**Depth below ground level [cm]**	**Moisture content at the time of sampling [%]**	**Saturated moisture content [%]**	**Difference between saturated moisture content and moisture content at the time of sampling [%]**
Boční mokřad	7	80	82	**Boční mokřad**	14	79	82	3
Boční mokřad	25	67	69	**Boční mokřad**	23	78	81	3
Boční mokřad	47	88	87	**Boční mokřad**	32	71	72	1
Boční mokřad	53	87	86	**Boční mokřad**	50	53	54	1
LI2	12	72	74	**Boční mokřad**	73	93	94	2
LI2	12	75	79	**LI2**	7	80	84	4
LI2	18	44	43	**LI2**	14	68	71	3
LI2	26	76	76	**LI2**	16	40	41	1
LI2	42	70	68	**LI2**	24	66	66	0
Střední mokřad	10	69	69	**LI2**	25	60	61	1
Střední mokřad	20	47	49	**LI2**	35	37	42	5
Střední mokřad	30	75	76	**Střední mokřad**	16	72	74	2
Střední mokřad	41	81	82	**Střední mokřad**	20	42	36	0
Střední mokřad	65	43	48	**Střední mokřad**	27	73	73	0
U Studánky	14	41	55	**Střední mokřad**	30	66	67	1
U Studánky	28	54	78	**Střední mokřad**	50	43	42	0
U Studánky	48	35	34	**U Studánky**	16	73	75	2
U Studánky	54	40	40	**U Studánky**	25	82	84	2
U Studánky	72	41	41	**U Studánky**	39	81	82	1
				U Studánky	49	75	78	3

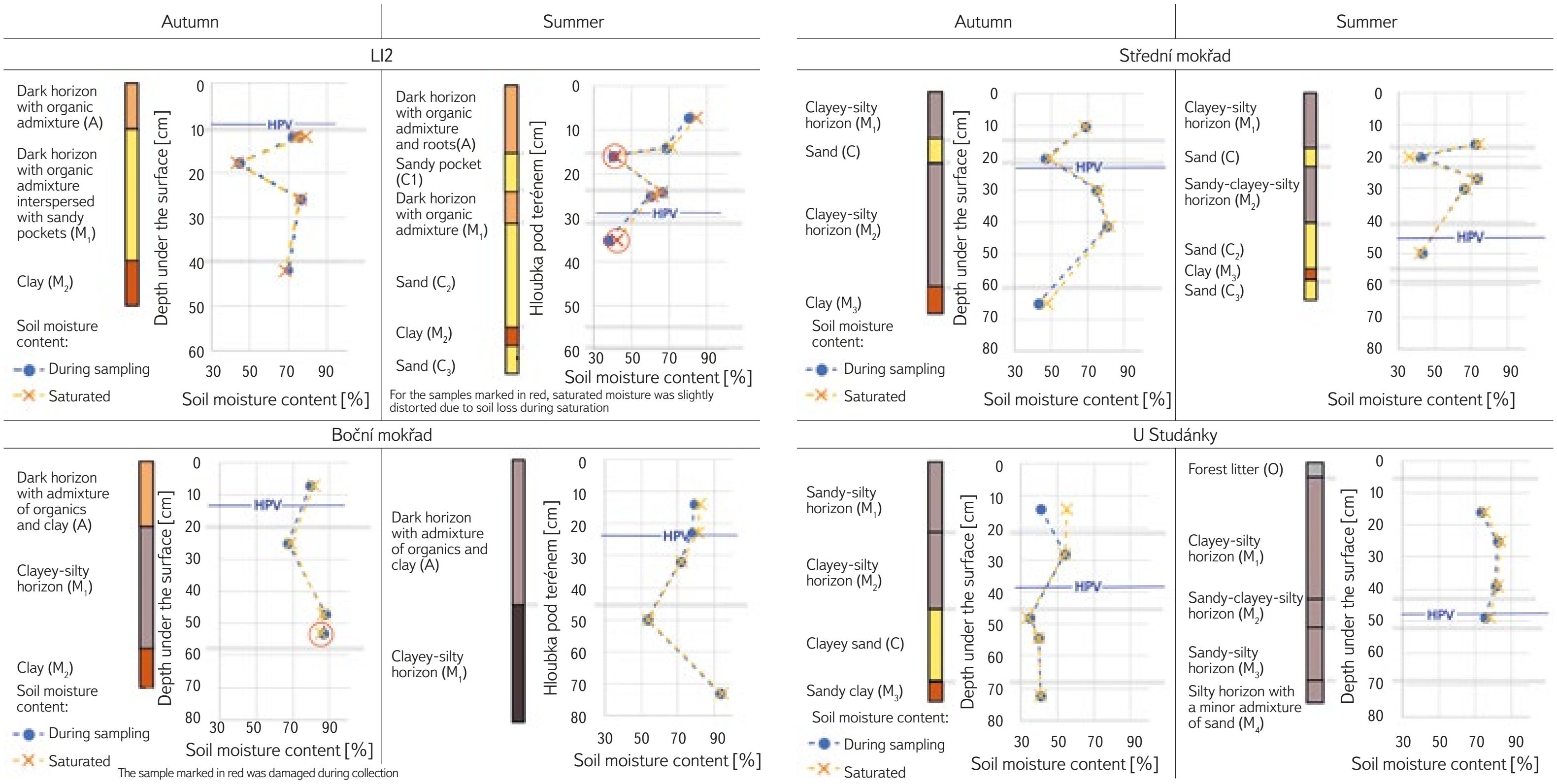

Fig 8. Measuring soil moisture content in the wetland (in the profile description, soil horizons are described in brackets according to the soil classification)

Estimation of ET from GWL fluctuations

The first step was to determine specific yield of the soil layers at the GWL depth. Summer moisture measurements were used, as diurnal GWL fluctuations were recorded during the warm part of the year. From the sampled profiles, the Boční mokřad and U Studánky piezometers were selected for analysis, as the soil composition at these sites showed little variation with depth. Samples were taken from layers above the GWL, and specific yield was determined as the difference between the saturated moisture content and the moisture content at the time of sampling. Using this method, specific yield was estimated at 2 %, with an absolute error of ±1 %.

In the second step, White's method [1] was applied to calculate ET based on the measured diurnal fluctuations in GWL at the U Studánky and Boční mokřad piezometers. The results are summarised in *Tab. 4*. The values presented are average ET calculated based on eight periods during which significant GWL fluctuations occurred over several days. The final ET estimate was calculated as the arithmetic mean of the values obtained from both piezometers and reached 20 $l \cdot s^{-1} \cdot km^{-2}$, with a range of ± 10 $l \cdot s^{-1} \cdot km^{-2}$ considering a specific yield variation of ± 1 %.

In the final step, the obtained ET value was compared with the results of ET estimation by a different method, carried out by the same authors in the same wetland [25]. In the referenced study, ET was calculated based on fluctuations in streamflow, under the assumption that the highest daily flow represents discharge unaffected by ET. Using this method, ET was determined to be 11 $l \cdot s^{-1} \cdot km^{-2}$. Based on the calculation of PET using Oudin's method, the study reported an average ET value of 25 $l \cdot s^{-1} \cdot km^{-2}$. These results from 2022 are consistent with the ET values obtained using White's method – 20 $l \cdot s^{-1} \cdot km^{-2}$ – within the accuracy limits of the specific yield estimate.

In order for the ET value of 20 $l \cdot s^{-1} \cdot km^{-2}$, as determined by White's method, to be valid also for the Střední mokřad piezometer, it was retrospectively calculated that the specific yield for this piezometer would need to be approximately 2 %. In the case of the LI2 piezometer, the same retrospective approach indicated a specific yield of around 1 %. Both values fall within the expected range of specific yield. In future measurements, it will be beneficial to compare the determined specific yield value with specific yield obtained experimentally by other means – for example, through a pumping test with a small volume of water extracted from piezometers, while monitoring the drawdown and geometry of the cone of depression using additional temporarily installed piezometers in the vicinity. This approach should lead to a significantly more accurate determination of specific yield.

Tab. 4. ET in the wetland calculated using the White method

Boční mokřad				U Studánky			
Specific yield	**ET [$mm \cdot den^{-1}$]**	**ET [$l \cdot s^{-1}$]**	**ET [$l \cdot s^{-1} \cdot km^{-2}$]**	**Specific yield**	**ET [$mm \cdot den^{-1}$]**	**ET [$l \cdot s^{-1}$]**	**ET [$l \cdot s^{-1} \cdot km^{-2}$]**
1%	0.94	0.2	11	1%	0.78	0.2	9
2%	1.89	0.4	22	2%	1.56	0.3	18
3%	2.83	0.6	33	3%	2.33	0.5	27

DISCUSSION

The measurements presented build upon previous research steps carried out by the same author at the same site. The results are consistent with findings from international studies presented in the first (review) section. In Pátek's 2022 study [24], diurnal GWL fluctuations and streamflow were detected at a single location within the wetland. Daily amplitude of fluctuations (difference between maximum and minimum GWL on a given day) increased with higher temperatures. This relationship was more apparent when only rain-free days were considered, and it was most pronounced on days with more than nine hours of sunshine. These findings confirmed the assumption that the detected fluctuations were caused by ET influence. In the subsequent article by Pátek and Bruthans from 2023 [25], diurnal streamflow fluctuation was used to estimate the amount of water consumed by vegetation, and the result was compared with the theoretical method of calculating PET using Oudin's approach. A delay was also observed between the timing of streamflow minimum and maximum and the solar cycle. The typical timing of the daily streamflow maximum (around 08:00) and minimum (around 16:30) is consistent, for example, with the findings of Gribovszki [3], obtained from an experimental catchment in the foot of the Alps, where in August the maximum flow occurred around 07:00 and the minimum around 16:00.

In the monitored wetland, the previous research steps were expanded by simultaneously observing the GWL across the entire area, which enabled an assessment of the spatial variability of diurnal GWL fluctuations. The fluctuations occurred uniformly and synchronously throughout the wetland, including its peripheral parts, with the most pronounced effect observed in the central area.

Soil moisture measurements yielded further interesting findings. The soil above the GWL was close to saturation, even during the summer months when ET was intense. It is surprising that, despite the presence of an unconfined aquifer, specific yield is very low – only a few per cent. This is due to the predominance of fine-grained material, in which the vast majority of pores are filled with water (capillary fringe), and only a very small volume contains air.

The high moisture present throughout the entire soil profile allows ET to have a more pronounced effect on GWL fluctuations. The primary reason for this is the easy availability of water to plant roots. The second reason is that only a small amount of water is needed to fully saturate the soil above the GWL, causing the GWL to rise (specific yield is therefore low, ranging from 1 to 4 %). As a result, a small volume of added or removed water produces pronounced vertical movements of the GWL. High soil moisture in summer also means that water is readily available even during dry periods, when ET from areas outside the wetland is limited by water scarcity. Consequently, the relative impact of secondary ET from the wetland on the landscape's water balance is greater.

The measured data allowed only an approximate estimate of specific yield. Differences between the moisture content of saturated samples and that of samples at the time of collection were in the single digit per cent range, roughly comparable to the accuracy of the method used. In addition, the properties of the individual soil profile layers varied significantly. This strong spatial variability in specific yield values suggests that even taking multiple Kopecký cylinders from a single specific depth within a given profile would not lead to a significant improvement in accuracy. This indicates that, in future studies, it will be more appropriate to use other methods for determining specific yield in the wetland, such as a miniature pumping test (pumping in $ml \cdot s^{-1}$ and water level drawdown in decimetres around the piezometer). An additional advantage of this method is that it reflects the overall behaviour of a larger volume of the environment and is therefore less sensitive to the heterogeneous soil profile composition in the wetland, which is influenced by the watercourse dynamic activity.

Diurnal fluctuations caused by ET were observed in the wetland both in streamflow (i.e., a decrease in flow during the day compared to night) and in GWL fluctuations (a drop in GWL during the day compared to night). The connection between the fluctuations of these two variables stems from the fact that the main source of water for the small stream flowing through the wetland is groundwater. Measuring fluctuations in GWL was easier compared to measuring fluctuations in streamflow because the vertical range of GWL movement was greater than that of the stream's surface level fluctuations, making the measurements relatively more precise. In environments where the GWL is close to the surface and the time lag between GWL fluctuations and streamflow is small, this finding suggests an alternative, simple method to measure streamflow with minimal instrumentation requirements. The close relationship between GWL and streamflow is also consistent with the results of Moore [15]; here, a strong correlation was described between current streamflow and soil moisture, with the relationship found to be more accurate at higher soil moisture levels.

The conducted measurements also demonstrate how diurnal GWL fluctuations can be detected using relatively simple and low-cost equipment. The measurements required only piezometers consisting of pressure sensors inserted into perforated pipes buried in the ground, whose installation involved only minimal environmental disturbance. The low maintenance requirements enable long-term monitoring, which could be used, for example, to assess the wetland condition, including the early detection of changes such as drought or tree dieback. Adding a simple weir to the measurement network allowed manual volumetric streamflow measurement. A flow rating curve was established, and the water level records captured by the sensors could be converted into streamflow records [25].

CONCLUSION

The literature review section of the article contains an overview of research focused on the impact of ET in wetlands on the water balance. The research can be divided into four groups, describing a spectrum of articles ranging from those examining detailed fluctuations of groundwater level (GWL) to more comprehensive studies combining GWL fluctuations with streamflow and, in some cases, also with measurements of vegetation transpiration by other methods, such as measuring sap flow in trees.

Theoretical knowledge was complemented in the second part by practical measurements illustrating the situation in the western part of the Czech Cretaceous Basin. The ET influence on the wetland in the upper catchment of the Liběchovka was monitored. Consistently with the results of the review conducted in the first part of the work, three main observations were made:

- a significant influence of secondary ET in the wetland on water balance,
- diurnal GWL fluctuations which, due to ET, occur simultaneously throughout the wetland during the summer months,
- a temporal delay of the diurnal GWL fluctuations and streamflow relative to the solar cycle.

Based on diurnal fluctuations, the wetland ET was determined to be $20\ l \cdot s^{-1} \cdot km^{-2}$. This is an average value representing the warm periods of the summer, during which significant diurnal GWL fluctuations were observed over several days. This is consistent with the results of previous measurements at the same site by the same authors [25], where ET was derived from fluctuations in the flow of the small watercourse passing through the wetland and from Oudin's method for calculating PET.

The necessity of considering the influence of wetland ET on the water balance in Central Europe was thus supported, as demonstrated by Bruthans in his 2020 study [2], based on streamflow measurements. From this study and other similar studies, the following important general conclusion emerges, carrying significant implications for hydrological practice. Wetlands and similar environments are not elements that retain water in the landscape; rather, they are environments where water is intensively consumed by vegetation and, under high summer temperatures, rapidly lost to the atmosphere.

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Xenobiotic substances in solid matrices of aquatic ecosystems

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Keywords: POPs — mercury — accumulation — solid matrices — rivers of the Czech Republic — biota — sediment

ABSTRACT

Monitoring of substances such as halogenated and other hazardous organic pollutants or heavy metals provides valuable information about environmental pollution. These persistent substances accumulate in both biotic and abiotic compartments, as well as in food chains, and many of them act as human carcinogens and endocrine disruptors. The Czech Hydrometeorological Institute's annual monitoring results show long-term contamination of surface water ecosystem in the Czech Republic by these substances. Mercury contamination was documented practically in all evaluated samples with consistently elevated concentrations exceeding environmental quality standard (EQS) in adult fish. Perfluorooctane sulfonate (PFOS) was identified at above EQS concentrations in 50 % of monitored profiles in juvenile fish. Concentrations of dichlorodiphenyltrichloroethane (DDT) and polychlorinated biphenyls (PCB) show a slightly declining trend in some cases. For selected contaminants, their distribution in biotic (benthic organisms, fish, biofilm) and abiotic (sediments, sedimentable solids, suspended solids) matrices was evaluated. In addition, results from passive samplers and surface water were also included.

INTRODUCTION

Persistent substances arise as a result of various industrial and other anthropogenic activities. Some of them have been produced deliberately (pesticides, brominated flame retardants, polychlorinated biphenyls (PCBs), per- and polyfluoroalkyl substances (PFAS)), while others arise as unintended by-products (polycyclic aromatic hydrocarbons (PAHs), dioxin compounds). These substances may also be released from various consumer products, which serve as their source (flame retardants used in furniture, household appliances or textiles, nanomaterials, chemicals used to create non-stick surfaces, plasticisers, phthalates, etc.). From their point of origin, contaminants can be transported through the atmosphere and subsequently distributed globally into other environmental components. Important pathways for their entry into the environment is through wastewater, contaminated soils, and waste landfills [1]. The high chemical stability and lipophilic nature of these substances lead to their sorption onto solid particles, accumulation in organisms, and subsequent transfer through food chains. Due to their ability to be transported over long distances from the source of pollution, some persistent organic pollutants (POPs) tend to contaminate even remote ecosystems and negatively affect the health of organisms on a global scale. For example, in polar bears, they can disrupt hormonal processes [2].

In aquatic ecosystems, contaminants are distributed among different matrices. Depending on their physicochemical properties, some substances have a higher affinity for organic carbon and therefore primarily accumulate in sediments or suspended solids, while others tend to accumulate in the fatty tissues of organisms or bind to proteins – e.g., PFAS [3]. In water, most POPs are found only in minimal concentrations due to their very low solubility. For this reason, to assess the pollution status of an aquatic ecosystem by certain contaminants (such as mercury, phthalates, DDT, or PCBs), it is more appropriate to monitor solid matrices. Passive samplers also play a significant role here, as they concentrate dissolved substances directly from the water column, allowing their effective detection even at very low concentrations [4, 5].

In addition to well-characterised environmental contaminants (DDT, PCBs, PAHs), a number of relatively new, so-called emerging pollutants are also entering the environment, whose toxic effects have not yet been fully explored. This group includes a wide range of chemical substances, such as pharmaceuticals, personal care products, pesticides, and their metabolites. Many of these substances are characterised by high mobility in the environment due to their solubility in water, which can result in their presence even in drinking water [6]. It is also important to consider degradation products or synergistic interactions between different pollutants, which can induce toxic effects even at concentrations individually considered safe [7].

Tab. 1. Evaluated matrices

Matrix	Sediments	Suspended solids	Sedimentable solids	Biofilm	Benthic organisms	Fish – juvenile	Fish – adult	SPMD	Water
Number of samples per year	2	2–4	6	1					12
Units	[µg · kg^{-1} dry weight]				[µg · kg^{-1} wet weight]			[µg · kg^{-1} triolein]	[µg · l^{-1}]

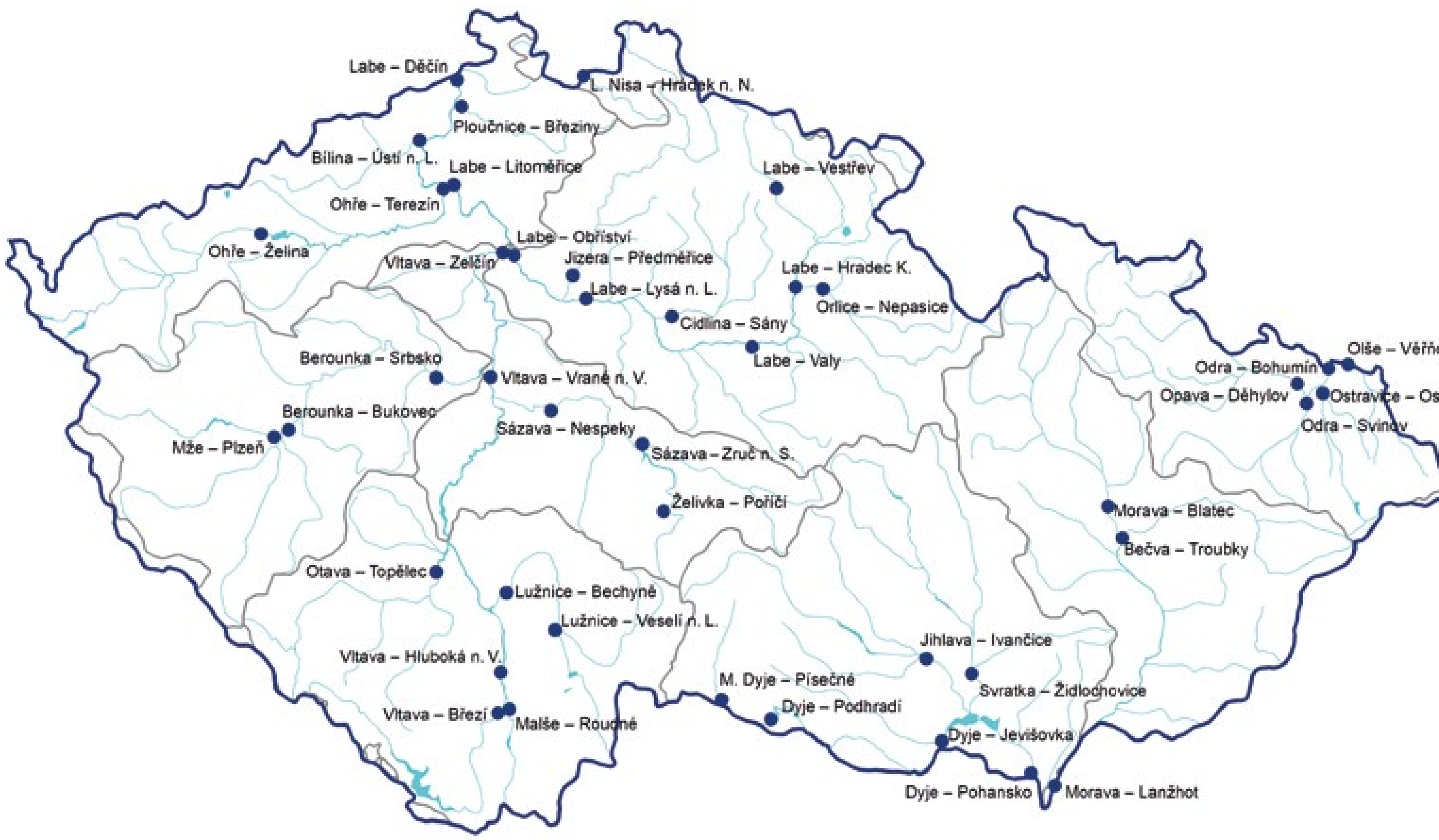

Fig. 1. Map of monitored profiles

As part of the regular annual monitoring of solid matrices, the Czech Hydrometeorological Institute (CHMI) tracks the content of more than 90 substances that have the potential to accumulate in both biotic and abiotic components of aquatic ecosystems. The main aim of this article is a comprehensive assessment of water pollution by hazardous substances from various perspectives, focusing on differences between individual matrices, including long-term trends and the influence of specific profiles.

METHODOLOGY

A total of 43 profiles of the main rivers in the Czech Republic were selected for the assessment, where all monitored matrices are sampled in the long term (*Fig. 1*). In the case of biotic matrices, these profiles are divided into two sets, which alternate every three years. The list of monitored matrices, the number of samples per year, and the corresponding units are provided in *Tab. 1*.

For benthic organism analyses, the main samples are leeches (*Erpobdella* spp.), caddisflies (*Hydropsyche* spp.), and amphipods (*Gammarus* spp.) For adult fish, the species is the common chub (*Squalius cephalus*). Semipermeable Membrane Device (SPMD) passive samplers, used for monitoring non-polar organic micropollutants, are filled with triolein fat and exposed to water for three weeks. Sedimentable solids are sampled for four to eight weeks, depending on the specific location, using sediment trap boxes, and the suspended solids are actively collected with a mobile centrifuge.

The substances selected for analysis of aquatic ecosystems contamination include: benzo(a)pyrene (B(a)P) and fluoranthene (FLU) as representatives of polycyclic aromatic hydrocarbons (PAHs), di(2-ethylhexyl) phthalate (DEHP), polybrominated diphenyl ethers (PBDEs), PCBs, DDT and its metabolites, perfluorooctane sulfonate (PFOS), and mercury (Hg). Of the selected contaminants, only PAHs are analysed in the biofilm. In the case of abiotic matrices, concentrations are not normalised to organic carbon content. Chemical analyses are conducted in external laboratories depending on the matrix and the group of substances monitored. For the determination of metals and PFAS in adult fish, muscle tissue samples were used, while other organic substances were analysed in muscle tissue with skin.

Boxplots were used to interpret the data, incorporating results from 2006–2023 depending on the type of matrix and the substance monitored. Selected substances have a limit concentration, known as the environmental quality standard (EQS) for biota, established by Government Regulation No. 401/2015 Coll., against which the measured concentrations are compared.

RESULTS AND DISCUSSION

Occurrence of selected contaminants in monitored matrices

The distribution of substances differs between the various biotic and abiotic matrices (*Fig. 2*). Specific differences in the distribution of contaminants across matrices reflect their differing physicochemical properties and interactions with the environment. Analytical method parameters, such as the limit of quantification (LOQ), may also play an important role.

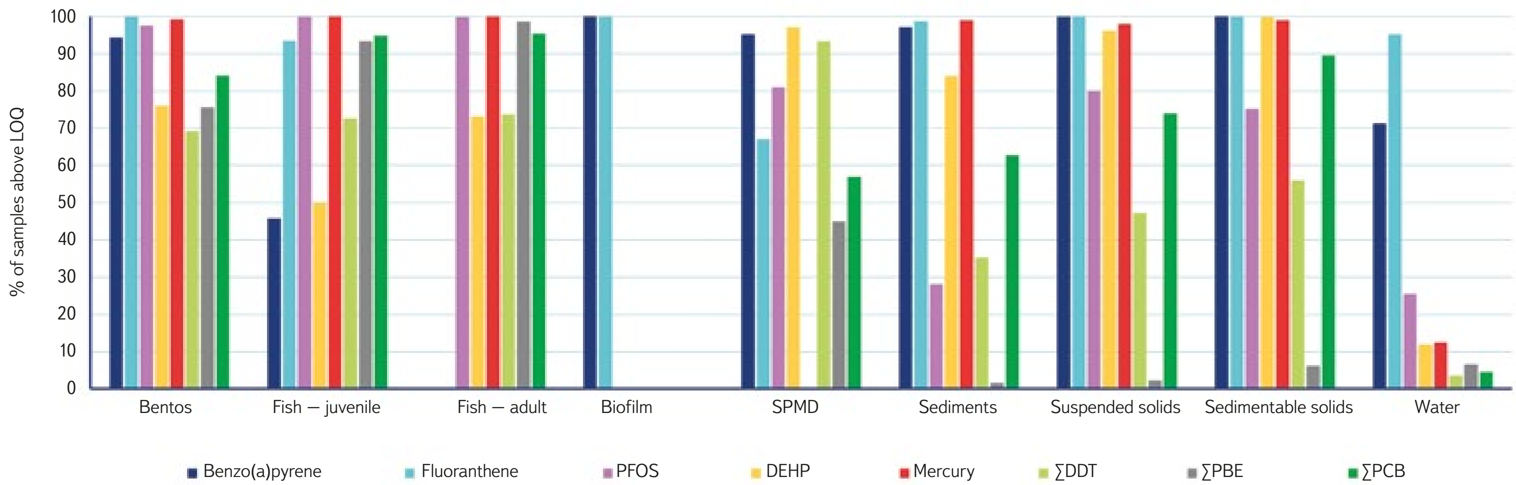

Fig. 2. Occurrence of selected substances above LOQ [%] in individual matrices for the period 2006–2023 (according to specific matrix and substance)

PAHs were found above the LOQ in 100 % of biofilm, suspended solids, and sedimentable solids samples. Currently, PAH analyses are not conducted in adult fish, as these substances can undergo significant metabolism within the fish organism [8]. This also partly affects the occurrence of PAHs in juvenile fish, where, for example, B(a)P was detected in less than half of the samples. PBDEs were found in more than 75 % of biota samples, in contrast to abiotic matrices and water, where they were rarely detected above the LOQ. PFOS was present in nearly 100 % of biota samples, with slightly lower occurrence in abiotic matrices, except for sediments, where – similarly to water – it was detected in only 25 % of samples. Mercury was detected in nearly 100 % of solid matrices, while in water it was recorded in only 10 % of cases.

Distribution of substances in solid matrices of surface waters

Benzo(a)pyrene (B(a)P) and fluoranthene were assessed as representatives of PAHs, with concentrations in biota ranging two to three orders of magnitude lower than in abiotic matrices (*Fig. 3*). An exception is biofilm, which, unlike most animals, lacks a metabolic transformation mechanism for PAHs, so their concentrations are comparable to those in abiotic matrices. However, biofilm may also contain a certain amount of inseparable abiotic fraction, which can influence the resulting concentrations. In juvenile fish, PAH concentrations are orders of magnitude lower than in benthos, which can be attributed not only to differences in metabolic capacity but also to the fact that benthic organisms are exposed to significantly higher PAH levels from sediments than fish. Although both benthic organisms and juvenile fish metabolise PAHs through similar mechanisms involving cytochrome P450 enzyme systems, this capacity is considerably limited in some benthic species [9]. However, the lower measured concentrations of parent PAHs in organisms may be due to their rapid transformation into potentially more toxic metabolites, whose concentrations can be higher compared to the original substances [9]. In abiotic matrices, the concentrations of B(a)P and FLU were comparable in magnitude, with FLU detected at higher concentrations in all matrices. This difference can be explained by the greater amount of FLU released during combustion processes and its higher environmental stability [10].

Another substance evaluated was DEHP, which accumulates most in benthic organisms among the biota (*Fig. 4*). It is also the substance found in the highest amount in benthos of all the contaminants monitored. In abiotic matrices, DEHP concentrations are highest in suspended solids and lowest in sediments, directly proportional to the total organic carbon content. According to Huang et al. [11], a positive correlation was demonstrated between certain water parameters, such as chemical oxygen demand and ammonium

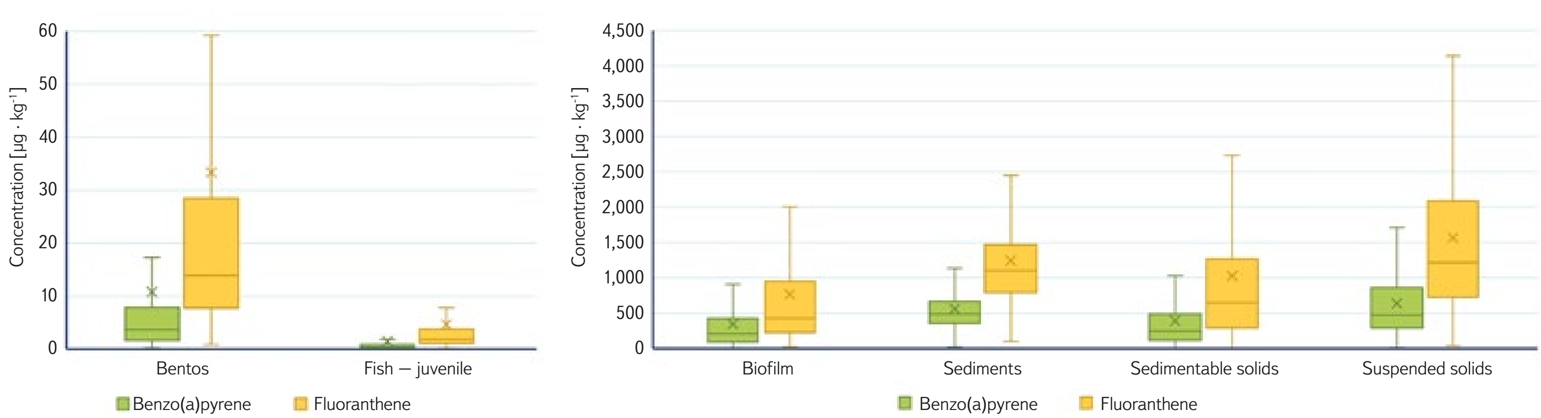

Fig. 3. Long-term concentration of selected PAHs: a) Biotic matrices without biofilm; b) Abiotic matrices, biofilm. Individual boxes include data from all monitored profiles for selected years (benthos: 2012–2023, others: 2006–2023). Medians (–), means (x), quartiles (box boundaries), and "maximum/minimum" (line endpoints) are indicated, excluding outliers.

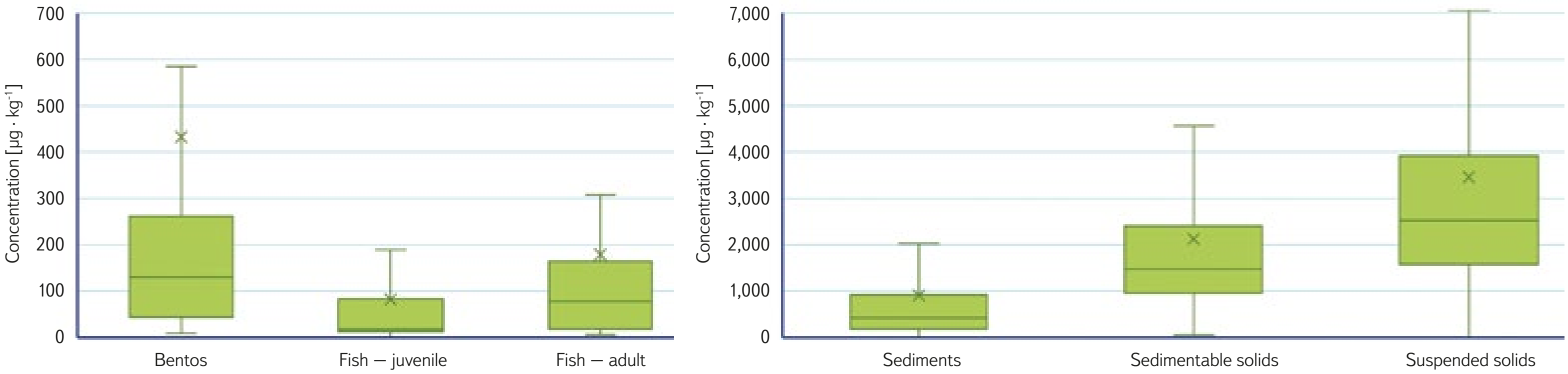

Fig. 4. Long-term concentration of DEHP: a) Biotic matrices; b) Abiotic matrices. Individual boxes include data from all monitored profiles for selected years (adult fish: 2012–2023, others: 2010–2023). Medians (–), means (×), quartiles (box boundaries), and "maximum/minimum" (line endpoints) are indicated, excluding outliers.

nitrogen concentration, and DEHP concentration in sediments; however, no effect of water temperature was observed. In contrast to our results, the mentioned study measured higher DEHP concentrations in fish than in benthic organisms, although the fish were predatory species (the chub is omnivorous).

Mercury concentrations (*Fig. 5*) show a different distribution across matrices compared to DEHP. In fish, mercury occurs at significantly higher concentrations over the long term than in other biotic matrices. In adult fish, the majority (up to 95 %) of total mercury may be present in the form of neurotoxic methylmercury (MeHg), which primarily binds strongly to muscle tissue, where it accumulates in the long term. In contrast, inorganic mercury Hg(II) tends to accumulate in the digestive system and liver, from where it is more easily eliminated [12]. An important property of MeHg is biomagnification, where its concentration increases with the trophic level of the organism; therefore, MeHg accumulates at demonstrably lower concentrations in benthic organisms, which occupy a lower level in the food chain, compared to predatory fish species. Unlike in biota, most mercury in abiotic matrices is present in inorganic form (MeHg represents at most a few percent of total mercury [13]), which may explain the negative correlation with organic carbon content confirmed by our results.

From a long-term perspective, concentrations of DDT, PCB, and PFOS occur in all matrices within a similar order of magnitude (*Fig. 6*). Slight differences were measured for PCB, which are found at the highest concentrations in sedimentable solids, and for DDT, where the highest accumulation was recorded in SPMD passive samplers. In SPMDs, concentrations are expressed only relative to fat content, which confirms the high affinity of DDT for lipids. The highest concentrations of PFOS in biotic matrices are regularly found in juvenile fish. This may be because, unlike the substances mentioned above, PFOS has an amphiphilic character and, besides fatty tissue and muscle, it is also present in high concentrations in blood, where it binds primarily to plasma proteins [3].

Contamination of individual profiles

In the long term, specific trends in the concentrations of hazardous substances are evident in the monitored profiles. The load on individual profiles is influenced not only by current industry but also by legacy environmental burdens, which include river sediments, where excavation can lead to remobilisation of contaminants. *Tab. 2* summarises the profiles most heavily burdened over the long term by selected substances. PAHs were regularly found at elevated concentrations in profiles from the Moravian-Silesian Region, particularly in the area of the Ostrava-Karviná coal basin. DDT and PCB exhibited the highest concentrations in the downstream profiles of the Elbe, specifically in the Děčín area, which may indicate cumulative transport of these POPs from the upper parts of the basin. In the Ústí Region, in the profiles of the Bílina and Ohře rivers, PFOS was found in the highest concentrations. Compared to other locations, the highest concentrations of DDT were measured in the Bílina – Ústí nad Labem profile in samples of not only biota but also other matrices (*Fig. 7*).

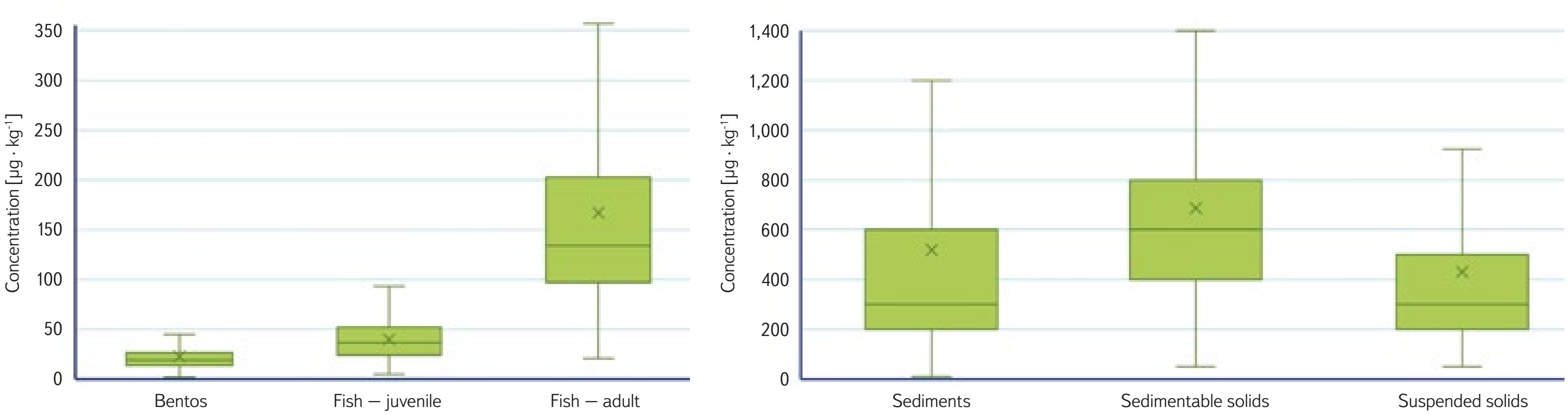

Fig. 5. Long-term concentration of mercury: a) Biotic matrices; b) Abiotic matrices. Individual boxes include data from all monitored profiles for selected years (2006–2023). Medians (–), means (×), quartiles (box boundaries), and "maximum/minimum" (line endpoints) are indicated, excluding outliers

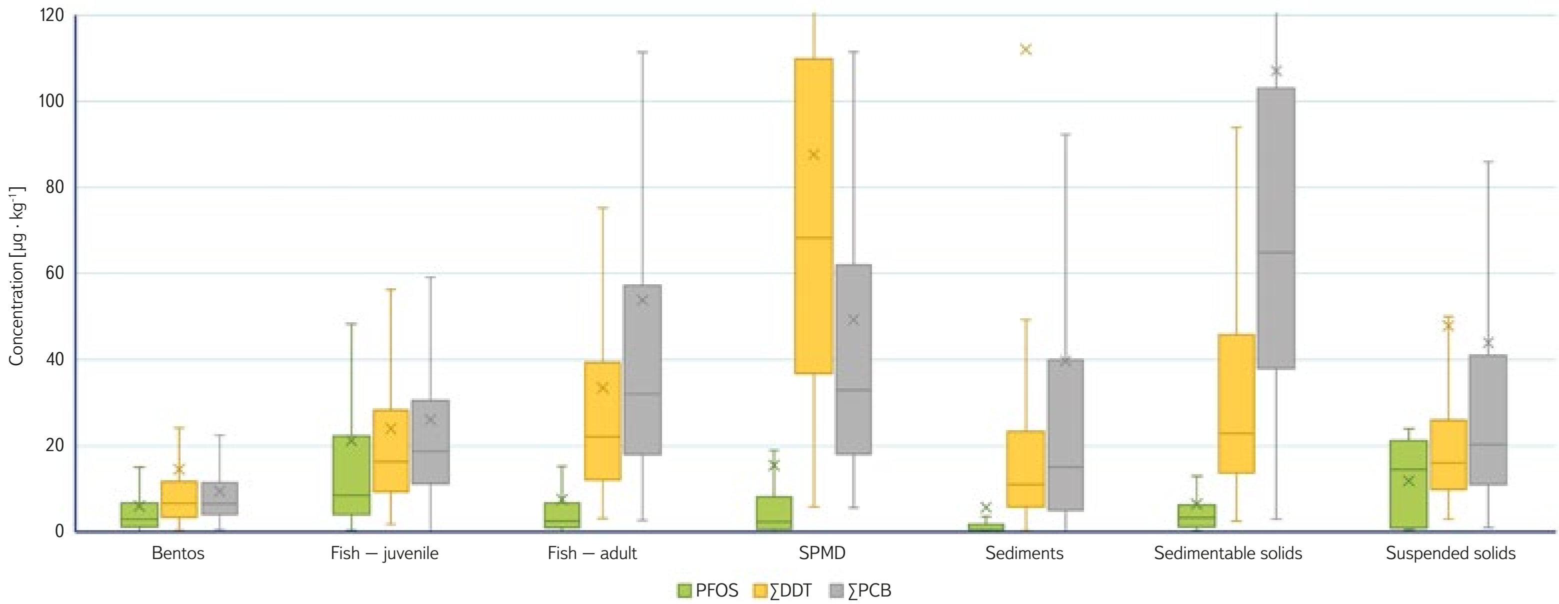

Fig. 6. Long-term concentration of selected POPs. Individual boxes include data from all monitored profiles for selected years (PCB, DDT: 2006–2023, PFOS biota: 2010–2023). Medians (–), means (×), quartiles (box boundaries), and "maximum/minimum" (line endpoints) are indicated, excluding outliers

For a more detailed assessment, bioaccumulation of PFOS in juvenile fish was analysed, where concentrations regularly exceeded the EQS limit of 9.1 $\mu g \cdot kg^{-1}$ (*Fig. 8*). Among the biotic matrices evaluated, the highest frequency of this value exceedance was recorded in juvenile fish samples, with more than 50 % of analysed samples exceeding the EQS. In contrast, exceedance of the EQS was recorded in only 20 % of benthic organism and adult fish samples during the monitored period.

An overview of mercury loads in fish at individual profiles, along with a comparison to the EQS value for all biotic matrices, is summarised in *Fig. 9*. The EQS for mercury, set at 20 $\mu g \cdot kg^{-1}$, was exceeded in 100 % of adult fish samples. However, within the framework of the European assessment of surface waters in the Czech Republic (similarly to many other countries), due to non-standardised evaluation procedures, most profiles show good chemical status in terms of mercury contamination, although this status was calculated from mercury concentrations in water, not in fish. In contrast, in Sweden, which uses mercury concentrations obtained from biota to assess chemical status, all measured profiles indicate poor status, even though mercury concentrations in fish there may be lower than ours [14, 15].

Long-term trend

The long-term development of concentrations was also assessed over the monitored period. Trends do not differ among individual representatives of biotic and abiotic matrices; however, a difference between these two groups was observed for certain substances. In some profiles, a decreasing trend was identified for biotic matrices only in the case of DDT and PBDE. Mercury concentrations in all matrices, as well as B(a)P in abiotic matrices, have remained broadly stable over the years. In contrast, B(a)P in biota and PFOS in abiotic matrices show a more fluctuating development of concentrations without any

Tab. 2. Profiles exhibiting maximal contamination by target substances in environmental matrices

<table>
<tr><th></th><th>Benzo(a)pyren</th><th>PFOS</th><th>∑ PCB</th><th>∑ DDT</th></tr>
<tr><td>Benthos</td><td>Odra – Bohumín</td><td rowspan="3">Bílina – Ústí nad Labem
Bečva – Troubky
Ohře – Želina
Labe – Obříství</td><td>Labe – Děčín
Jizera – Předměřice</td><td>Bílina – Ústí nad Labem</td></tr>
<tr><td>Fish — juvenile</td><td>Morava – Blatec</td><td>Labe – Litoměřice
Svratka – Židlochovice</td><td>Dyje – Pohansko
Bílina – Ústí nad Labem</td></tr>
<tr><td>Fish — adult</td><td>Neměřeno</td><td>Labe – Děčín</td><td>Labe – Děčín</td></tr>
<tr><td>SPMD</td><td>Odra – Bohumín</td><td>Bílina – Ústí nad Labem</td><td>Labe – Valy</td><td>Dyje – Pohansko</td></tr>
<tr><td>Sediments</td><td>Odra – Bohumín</td><td>Bílina – Ústí nad Labem
Ohře – Želina
Ohře – Terezín</td><td rowspan="3">Labe – Děčín</td><td rowspan="3">Bílina – Ústí nad Labem
Labe – Děčín</td></tr>
<tr><td>Suspended solids</td><td rowspan="2">Odra – Bohumín
Bečva – Troubky
Morava – Blatec
Lužická Nisa – Hrádek</td><td rowspan="2">Labe
Bílina – Ústí nad Labem</td></tr>
<tr><td>Sedimentable solids</td></tr>
</table>

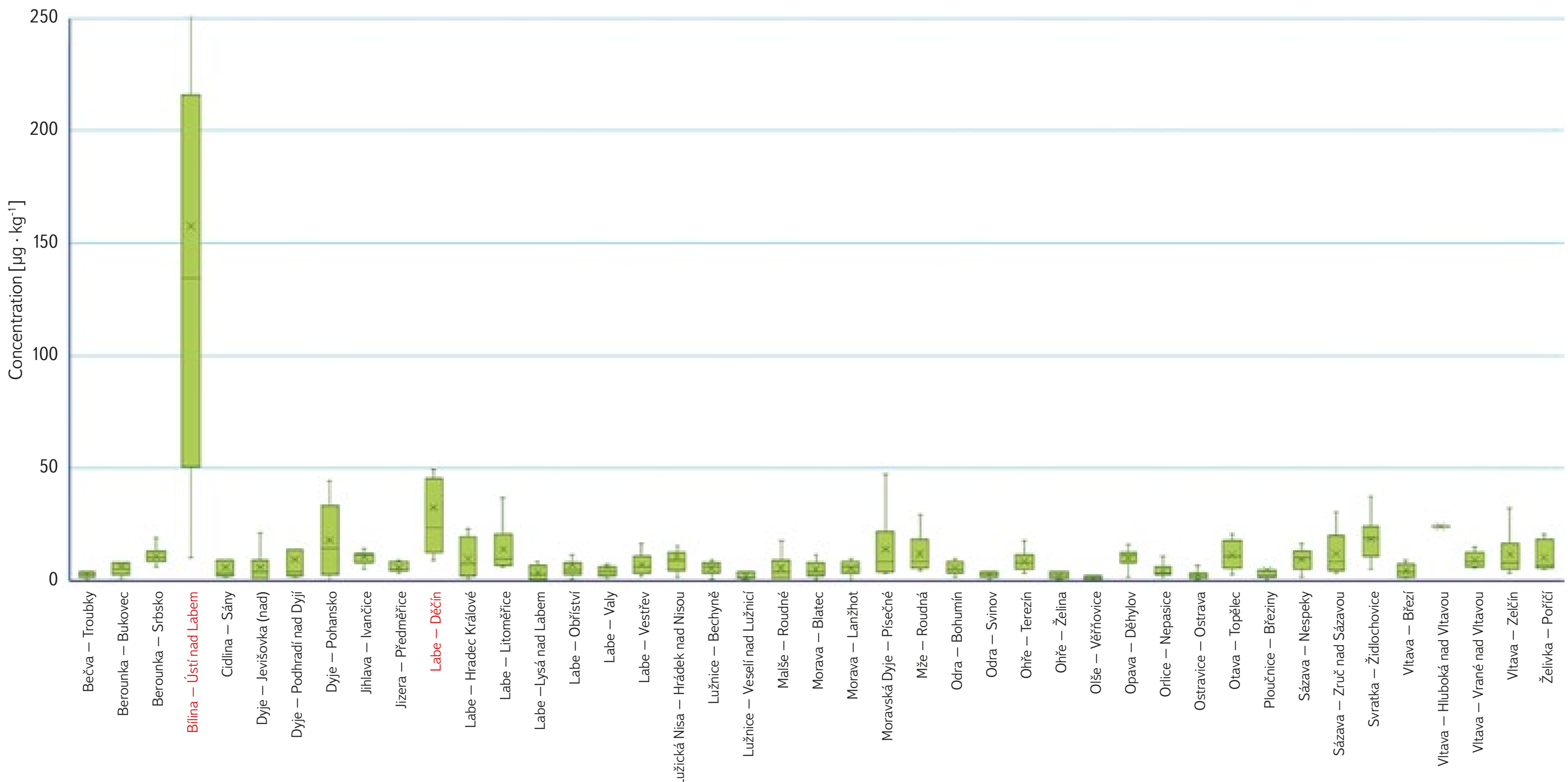

Fig. 7. DDT Concentrations in benthic organisms across monitored profiles for the period 2006–2023; profiles with the highest measured concentrations are marked in red

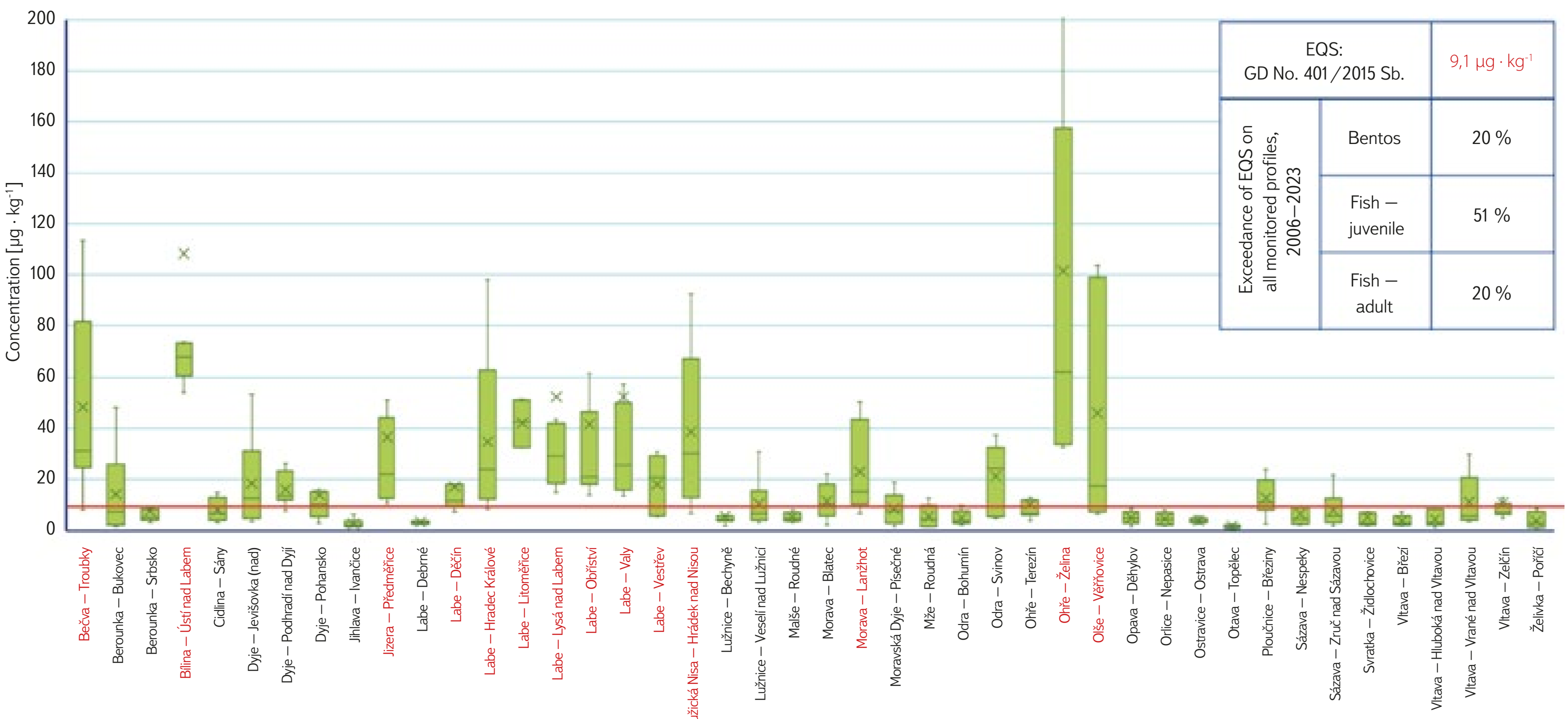

Fig. 8. PFOS concentrations in juvenile fish across monitored profiles for the period 2010–2023; profiles marked in red indicate locations where the EQS limit (red line) was exceeded in almost all samples (the table presents the percentage of profiles exceeding the EQS during the monitored period in biotic matrices)

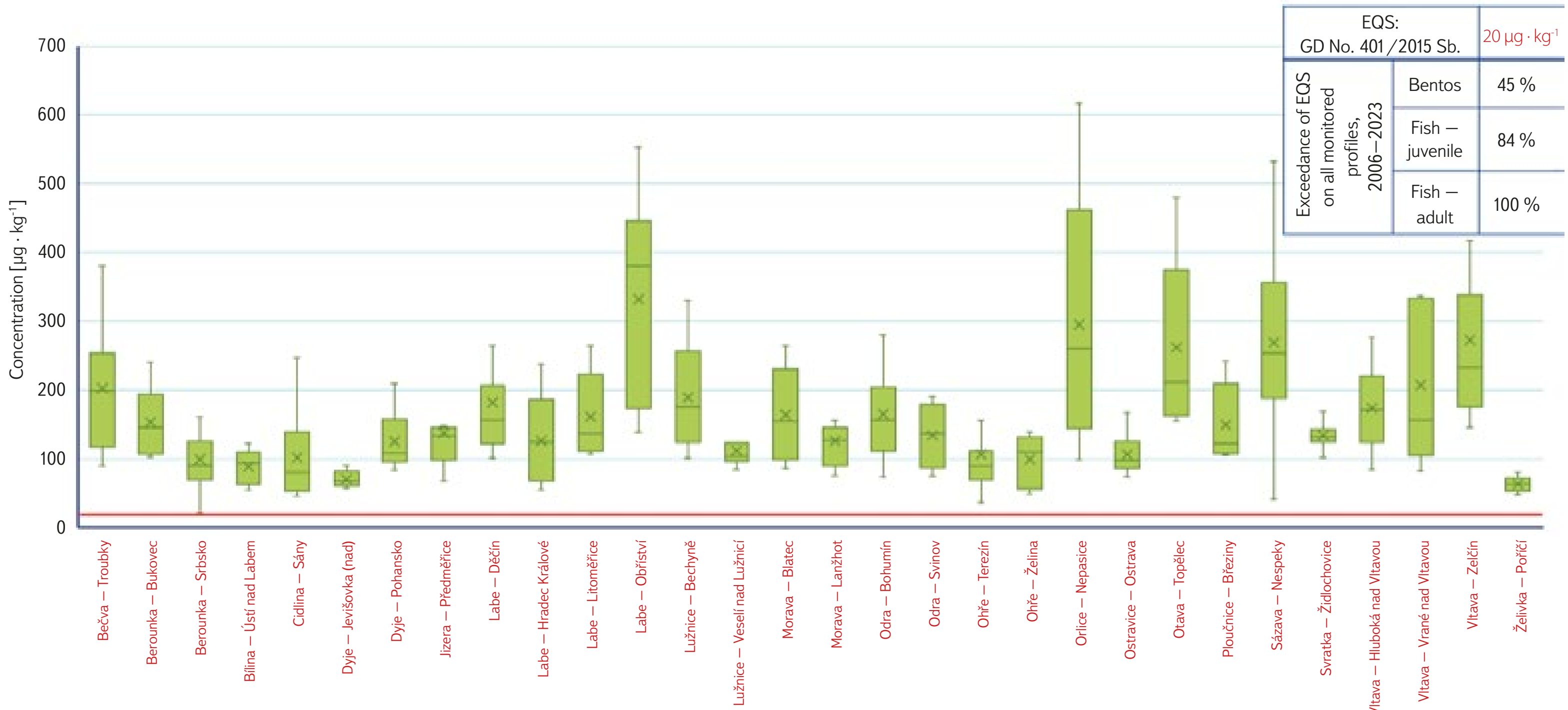

Fig. 9. Hg concentrations in adult fish across monitored profiles for the period 2006–2023; profiles marked in red indicate locations where the EQS limit (red line) was exceeded in almost all samples (the table presents the percentage of profiles exceeding the EQS during the monitored period in biotic matrices)

clear systematic pattern. For example, no decreasing trend has been observed in PFOS concentrations in juvenile fish, despite its inclusion in the Stockholm Convention in 2009, which significantly restricted its production [16] (*Fig. 10*). The historically highest concentration of PFOS (409 µg · kg^{-1}) was recorded in juvenile fish at the Bílina – Ústí nad Labem profile in 2016.

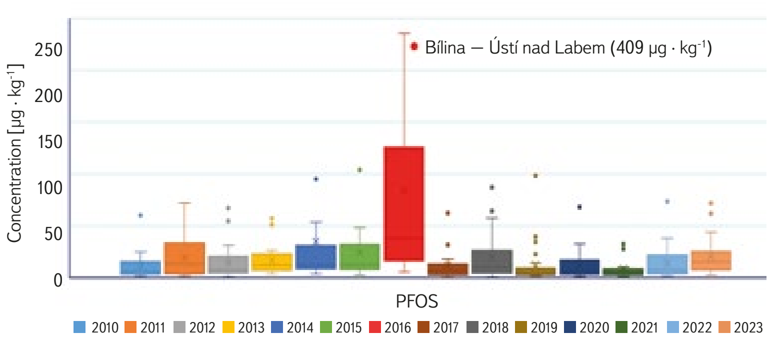

Fig. 10. Long-term trend of PFOS in juvenile fish with indicated maximum concentration

CONCLUSIONS

For a comprehensive assessment of aquatic ecosystem contamination, systematic monitoring of all matrices is necessary due to the uneven distribution of contaminants among them. Among the biotic matrices, the highest concentrations of mercury were measured in adult fish, with accumulation directly proportional to the trophic level within the food chain. In contrast, PAHs and DEHP were detected at the highest concentrations in benthic organisms, which are unable to metabolise these substances effectively. PFOS predominated in juvenile fish, where it accumulates significantly not only in fat and muscle tissue, but also in blood. In abiotic matrices, elevated concentrations of substances were detected in suspended solids (DEHP, PFOS) and in sedimentable solids (DDT, PCBs). In sediments, the concentrations of these POPs are lower, which may be related to the lower organic carbon content in this matrix.

The continuous development of analytical methods enables monitoring of an ever-wider range of xenobiotic substances which, combined with promising technologies for the elimination of toxic substances, green manufacturing processes, and ongoing updates to environmental legislation, can lead to the gradual minimisation of anthropogenic pollution. However, evaluating current results remains challenging because limits ensuring a good status of aquatic ecosystems are set for biota for only a limited number of substances, and not at all for abiotic solid matrices (despite the large number of measured indicators).

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The potential of grass strips for retaining surface runoff and sediment

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Keywords: grass strips — vegetative filter strips — surface runoff — sediment — soil erosion

ABSTRACT

The use of grass strips in agricultural landscapes is widely recognized for their ability to effectively reduce surface runoff and the transport of eroded particles, while simultaneously enhancing biodiversity and landscape stability. This study aimed to quantify the impact of grass strip length on sediment retention in surface runoff. Experimental measurements were conducted on enclosed plots measuring 8 × 1 metres, each with varying proportions of grass cover to simulate different grass strip widths under real-world conditions. Four treatment variants were tested: variant 1 with 0% grass cover (8 m bare soil); variant 2 with 25% grass cover (6 m bare soil and 2 m grass); variant 3 with 50% grass cover (4 m bare soil and 4 m grass); and variant 4 with 100% grass cover (8 m grass). Each variant was tested in triplicate. To simulate surface runoff with a high sediment load, an artificial suspension composed of water and finely ground sand with an average particle size of 27 µm and a target concentration of 40 $g \cdot l^{-1}$ was applied to the plots via a distribution system at a flow rate of 1 $l \cdot s^{-1}$ for 20 minutes from the onset of runoff. Results indicated that relative runoff volume decreased progressively with increasing grass cover, from 100% in the bare soil variant to 91%, 76%, and 71% in the 25%, 50%, and 100% grass cover treatments, respectively. Sediment transport was reduced even more substantially, from 100% in bare soil conditions to 51%, 24%, and 15% with increasing grass cover. Additionally, the velocity of surface runoff within the grass-covered areas was approximately 6.4 times lower than in bare soil conditions. The mean grain size of transported sediment decreased from 36 µm in the bare soil variant to 6.6 µm in the 100% grass cover treatment. These findings demonstrate that, under the given experimental conditions, increasing the proportion of grass cover significantly reduces both surface runoff and sediment transport. Moreover, vegetation plays a crucial role in promoting selective deposition of coarser sediment particles due to the substantial reduction in runoff velocity within the vegetated area.

INTRODUCTION

Grass strips in agricultural landscapes are generally considered an effective method for reducing surface runoff and preventing the transport of eroded particles further downslope [1] (*Fig. 1*). For this reason, they are commonly used either as a standalone agrotechnical measure or as part of a broader system of buffer strips within the standards of Good Agricultural and Environmental Conditions (GAEC) and the EU Common Agricultural Policy (CAP).

The benefits of grass strips lie not only in soil protection, but also in their positive impact on the quality of aquatic ecosystems and landscape stability [2–4]. By providing a suitable habitat for various plant and animal species, they support biodiversity and become a key element of sustainable landscape management [5]. Moreover, plants with deep root systems help stabilise soil structure and increase its resistance to erosion [6]. The effectiveness of these measures lies not only in slowing down runoff and increasing water infiltration [7, 8], but also in the sediment retention effect that occurs before water enters the grass strip, which gradually leads to the formation of terraces and a reduction in the longitudinal slope of the hillside, thereby slowing further erosion [9]. Various experimental methods are used to assess the effectiveness of grass strips, including the use of natural rainfall [10, 11], rainfall simulation using rainfall simulators [12], and direct simulation of surface runoff [13, 14]. Some studies even combine the use of rainfall simulators with the release of surface runoff to create the most realistic conditions possible using multiple methods, in order to analyse the effects of vegetation on erosion and sedimentation as accurately as possible under real-world conditions [12, 15]. In addition to several-metre-wide grass strips, there are also narrow grass barriers; the sturdy stems of selected plant species with lower spatial requirements effectively trap sediment and may even be more efficient in conditions of concentrated surface runoff [16, 17].

The presented study focuses on assessing the effectiveness of grass strips in reducing soil erosion, runoff, and sediment transport under controlled conditions. Its contribution primarily lies in verifying the methodological approach for the most realistic quantification of the effects of grass strips. The main objectives of the study are (1) developing and testing a system simulating surface runoff and sediment transport on agricultural land, and (2) applying the tested methodology to assess the impact of different lengths of grass cover (or grass strip width) on the ability to retain surface runoff and sediment. However, the set objectives represent only a partial step towards assessing the applicability of this measurement in more extensive research, which should follow from this pilot activity.

METHODOLOGY

The experimental measurements were carried out at Řisuty in the Czech Republic, located in Central Bohemia, approximately 30 km northwest of Prague (50.2173N, 14.0169E), at an altitude of 310–315 m above sea level. The area has a humid continental climate with an average annual temperature of 8°C and an average annual precipitation of 500 mm. The topsoil layer contains

Fig. 1. Example of an erosion event at the boundary between a grass strip and arable land (photo: T. Laburda)

9% clay, 55% silt, and 36% sand, which, according to the USDA-NCRS classification system, corresponds to silty loam. The dominant grass species was timothy (*Phleum pratense*), with meadow fescue (*Festuca pratensis*) and perennial ryegrass (*Lolium perenne*) present in smaller amounts.

The experimental plots measured 8 × 1 m and were created in four variants (*Fig. 2*) according to the grass cover ratio: 0% (variant 1), 25% (variant 2), 50% (variant 3), and 100% (variant 4). In real-world conditions, these variants would correspond to a field with bare soil without a grass strip (variant 1), or fields with grass strips of 2, 4, and 8 m in width (variants 2–4). Each variant was created and tested in three replications to ensure statistical relevance.

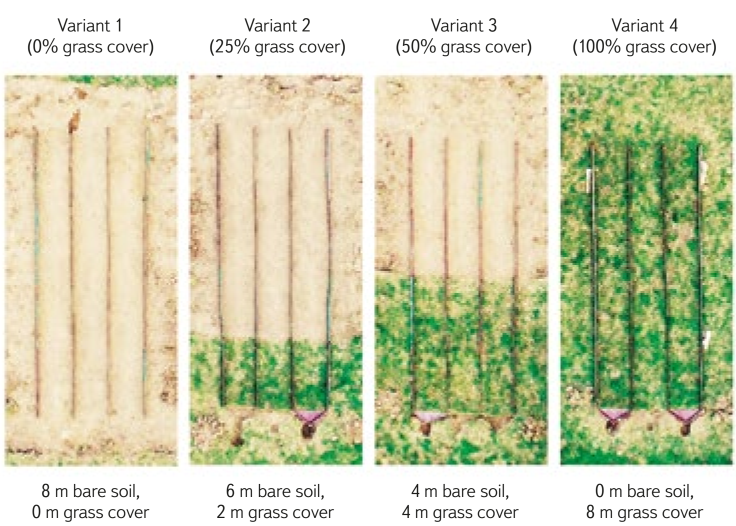

Fig. 2. Orthophoto images of the experimental plots for tested variants 1–4

The experimental measurements involved the release of a prepared suspension of solid particles simulating eroded sediment and water into the experimental enclosed area, followed by the retention of surface runoff at the discharge point. The target concentration of the suspension was 40 g · l^{-1}, and the material used was finely ground sand with a median grain size of 27 μm. The inflow at the upper edge of the plot was set to 1 l · s^{-1}. These values were selected based on steady-state runoff rates observed during previous repeated measurements using a rainfall simulator at the same site. They therefore represent realistic values that may occur during actual erosion events. Finely ground sand was chosen as a well-defined granular material whose grain size and bulk density closely corresponded to the values of eroded material observed during real erosion experiments using a rainfall simulator on actual field plots. The suspension was prepared in a 500-litre tank, into which water was continuously supplied in order to maintain a constant water level (to ensure steady gravitational discharge of flow onto the plot). The specified sediment was added to the tank at short intervals and kept in suspension by a sludge pump operating continuously within the tank. The suspension homogeneity was monitored through repeated sampling from the tank and at the inflow to the experimental plots.

Each experiment lasted 20 minutes from the onset of surface runoff at the closing profile. Surface runoff was measured at one-minute intervals during the first ten minutes, and at two-minute intervals during the following ten minutes. The sampling time was always recorded to determine the flow rate over time. Further analysis of the collected samples was carried out in the laboratory, where the samples were filtered, dried, and the amount (weight) of sediment was determined.

A selected number of samples (three samples from each measurement, taken at the 4th, 9th, and 20th minute of surface runoff) was further analysed using a Mastersizer 3000 laser diffractometer (Malvern Panalytical) to determine particle size distribution.

Surface runoff velocity was measured on each variant and replication three times in succession after the 15th minute of the experiment. Measurement was carried out using a coloured solution (Brilliant Blue), which was applied at the beginning of a continuous section of bare arable soil and grass cover, while the time taken to reach the end of this section was recorded.

The experimental measurements were conducted based on previous experiences gained within the international project LTAUSA19019, during which a device for discharging a suspension, replacing surface runoff, was tested at several other sites under identical conditions.

RESULTS

Based on the monitoring measurements, actual average flow rate and concentration of suspension at the inflow to the experimental areas were calculated. Average inflow to the area reached a value of 1.02 ± 0.13 $l \cdot s^{-1}$, and the average concentration of the created suspension was 33.5 ± 3.7 $g \cdot l^{-1}$.

Surface runoff

The graph in *Fig. 3* shows average runoff values from individual replicates of the plots for variants 1–4 from the onset of surface runoff. The fastest increase in runoff was observed in variant 1, which lacked grass cover. In the other variants, runoff increase was slower, depending on the proportion of grass cover. After 20 minutes of surface runoff, variant 2 reached an almost identical runoff rate of approximately 0.95 $l \cdot s^{-1}$ as that of variant 1 without grass cover. In contrast, variants 3 and 4 also reached very similar values of approximately 0.85 $l \cdot s^{-1}$ at the end of the experiment.

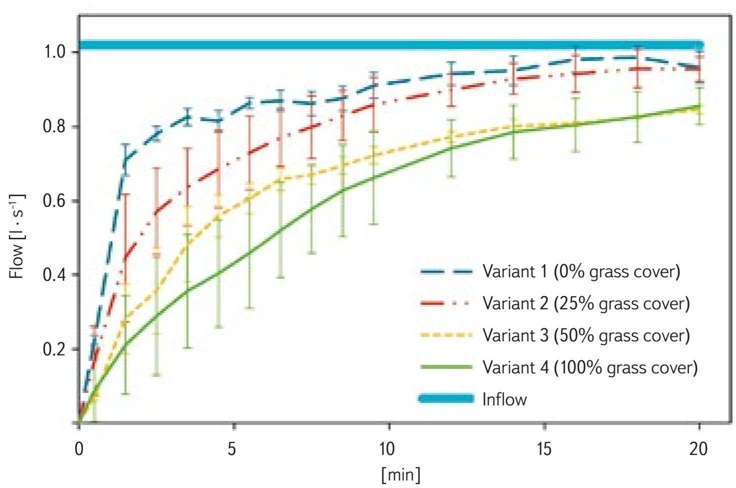

Fig. 3. Surface runoff progression for tested variants 1–4. The values shown are the averages of three replicates for each variant; error bars represent the standard deviation of the individual replicates

Average surface runoff velocity values for each plot variant are presented in *Tab. 1*. Total average surface runoff velocity on bare soil reached 0.58 ± 0.04 $m \cdot s^{-1}$, with very little deviation between the individual variants. On the grass-covered section, average surface runoff velocity was 0.09 ± 0.01 $m \cdot s^{-1}$. On average, runoff velocity on the grass-covered plot decreased approximately 6.4 times compared to the plot without vegetation cover.

Tab. 1. Average surface runoff velocities of variants 1–4

Surface	Average surface runoff rate of individual replicates and their standard deviation [$m \cdot s^{-1}$]			
	Variant 1	**Variant 2**	**Variant 3**	**Variant 4**
Bare soil	0.60	0.59	0.55	-
Grass cover	-	0.09	0.09	0.08

Sediment

The graph in *Fig. 4* shows average runoff concentration values from the plots of variants 1–4 from the onset of surface runoff. Variant 1, without grass cover, reaches very high values – up to 160 $g \cdot l^{-1}$ – within the first two minutes of runoff, followed by a rapid decline to a steady value of approximately 33 $g \cdot l^{-1}$. This development indicates very high erosion of unprotected soil at the beginning, followed by an almost complete inability to retain additional sediment from the discharged suspension. In contrast, variant 2 shows that initial erosion from bare soil is significantly reduced thanks to – even minimal – grass cover. In the first two minutes of runoff, the concentration reaches a maximum value of only 19 $g \cdot l^{-1}$. This is followed by a rapid decline and then only a slight increase as the capacity of the grass cover to retain sediment particles from the discharged suspension becomes gradually exhausted. At the end of the experiment, a concentration of approximately 27 $g \cdot l^{-1}$ is achieved. Variant 3 also shows a local increase in concentration at the beginning due to erosion of bare soil, but the subsequent rise is very gradual, and by the end of the experiment, it reaches a value of approximately 13 $g \cdot l^{-1}$. In variant 4, no local increase in concentration is observed at the beginning of runoff due to the absence of a bare, unprotected soil area. Nevertheless, even in this variant, there is a very slow increase in runoff concentration, reaching a value of approximately 9 $g \cdot l^{-1}$ by the end of the experiment.

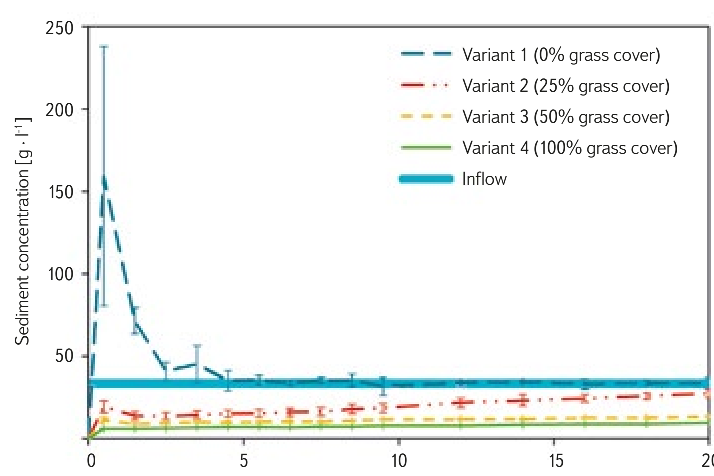

Fig. 4. Surface runoff concentration progression for tested variants 1–4. These values represent the averages of three replicates for each variant, with error bars indicating the standard deviation of the individual replicates

Sediment retention efficiency and runoff reduction

Based on the measured values throughout the experiments, cumulative values of runoff and sediment quantities for each variant were calculated, as shown in *Fig. 5*. As expected, sediment concentration in the runoff from the plots of variant 1, without grass cover, was the highest and therefore it was considered as 100%. Reductions in the other variants were then calculated relative to the values of variant 1.

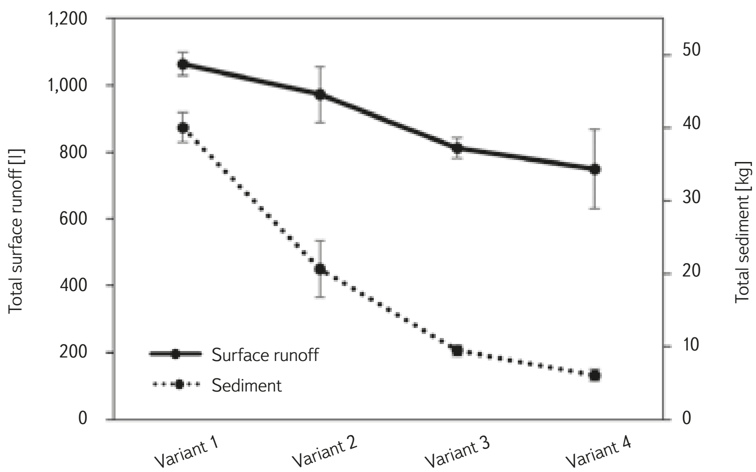

Fig. 5. Total amount of surface runoff and sediment for variants 1–4. These values represent the averages of three replicates for each variant, with error bars indicating the standard deviation of the individual replicates

The above graph shows that with a higher proportion of grass cover, both the total amount of runoff and erosion decrease. Runoff was reduced by 9% in variant 2 (25% grass cover), in variant 3 (50% grass cover) by 24%, and in variant 4 (100% grass cover) surface runoff decreased by 29%. In total, runoff decreased from 1,063 l to 972 l, and from 812 l to 750 l. The amount of sediment decreased even more significantly due to the effect of grass cover. In variant 2 (25% grass cover), the amount of sediment decreased by 49%, in variant 3 (50% grass cover) by 76%, and in variant 4 (100% grass cover) sediment decreased by 85%. In total, sediment quantity reduced from the original 40 kg to 21 kg, and from 9 kg to 6 kg.

Grain size distribution

The graph in *Fig. 6* shows the representation of individual particle size fractions in the outflow for variants 1–4, along with the mean grain size d50. By converting the overall grain size distribution of the sediment into the categories of clay (particles smaller than 2 μm), silt (particles from 2 μm to 50 μm), and sand (particles from 50 μm to 2 mm), the effect of grass strips in terms of selective sedimentation is illustrated. Most nutrients that negatively affect watercourses and reservoirs, such as phosphorus, nitrogen, and potassium, are mobilised primarily with clay particles, i.e. the finest fraction. A comparison of individual variants shows that the proportion of sand fraction decreases significantly in variant 2 and is almost negligible in variants 3 and 4. This marked decrease can also be observed in the silt fraction in variant 2; however, the subsequent reduction is no longer as pronounced. In the case of the clay fraction, there is an average decrease of 1 g (33%) between variant 1 and variant 2, but the change in the following variants is negligible.

This aspect is also reflected in the d50 value. In variant 1, a value of 36 μm was recorded at the outflow from the area due to the high erosion of unprotected soil. In variant 2, the filtering effect of transported material by the grass cover had already begun to take effect, resulting in a gradual reduction in the mean grain size – to 17.9 μm in variant 2, 8.3 μm in variant 3, and 6.6 μm in variant 4. The above findings show that grass strips effectively slow down the movement of coarse particles; however, they have a significantly smaller impact on the mobility of clay particles, which pose the greatest risk in terms of qualitative pollution. It should be reiterated that these are preliminary results aimed at verifying the experimental methodology. The retention ratio will strongly depend on the width of the strip and the duration of the runoff event, as well as on the volume of water discharged.

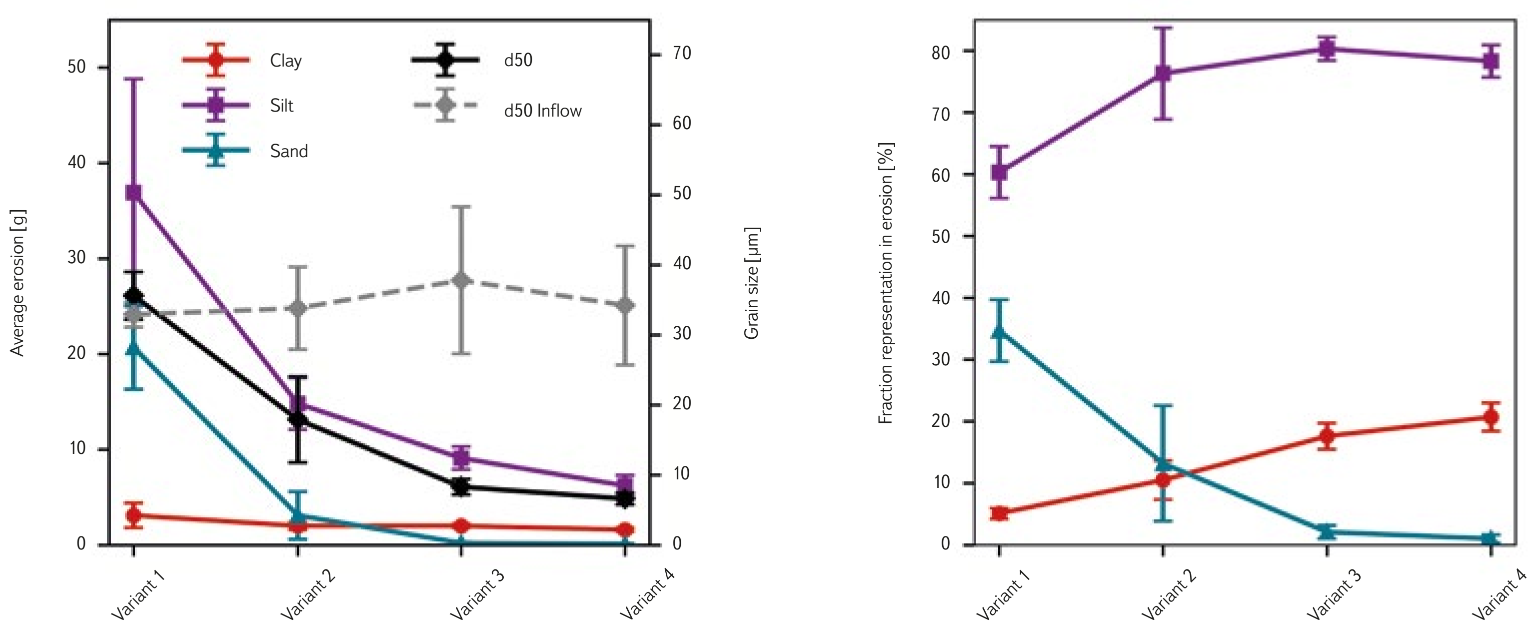

Fig. 6. Average soil erosion (g) and particle size distribution (%) of clay, silt, and sand for variants 1–4, including median grain size d50 of both the transported material and the inlet (prepared suspension); error bars represent the sample standard deviation

DISCUSSION

The results show that grass strips can be a highly effective measure for reducing surface runoff and sediment transport. However, the overall effectiveness heavily depends on the grass strip width and varies for runoff and sediment reduction. For example, the values for the fully grassed variant 4 reached only 71% of total surface runoff and 15% of total sediment quantity compared to the overall values of variant 1. The results clearly demonstrated that with an increasing proportion of vegetation, both runoff and sediment are reduced more, which is consistent with the findings of other studies [4, 18]. Although the presented study did not test more slope variants, other studies [19, 20] suggest that the primary factor influencing the effectiveness of runoff and erosion reduction is the length of the grass strip, rather than the slope on which the strip is located.

Based on the analysis of the grain size distribution, a decrease in the mean grain size of the eroded material was observed with an increasing proportion of vegetation, indicating the ability of grass strips to effectively capture only certain particle fractions. This effect has also been confirmed in other studies [4, 8, 10]. The effect of selective sedimentation in the area of vegetation cover is crucial with regard to nutrient transport, which is primarily associated with the transport of clay particles (< 2 μm). In this regard, it can be said that, under the tested conditions with this species composition and vegetation density, grass strips represent only a minimal obstacle to the transport of clay particles.

CONCLUSION

The above experiments were conducted based on the requirement to determine easily comparable parameters for different variants of grass strip lengths. Their results raise a number of questions related to the impact of flow rates, slope, vegetation density and species composition, duration of runoff, and the grass strip width. However, this pilot study with a limited number of experiments demonstrated that even relatively narrow grass strips can significantly reduce surface runoff and sediment quantity. With complete grass cover, a reduction of up to 29% in runoff and 85% in sediment was achieved, highlighting the potential of these measures in protecting agricultural land from erosion and water resources from sedimentation. The expected effect on reducing nutrient transport is lower, as grass strips primarily retain larger particles, which alters the enrichment ratio. Nevertheless, grass strips can be an effective solution both for agricultural production and for protecting water quality. The presented pilot study, together with subsequent research, can significantly contribute to further development and understanding of all the benefits, as well as to optimization of the design, sizing, and management of grass strips. To obtain presentable measurements, it is advisable to consider various configurations of experimental plots. In addition to the chosen discharge of artificially prepared suspension, another option could be the use of bare arable land in front of the grass strip, which would provide a sufficient amount of eroded material without the need for additional discharge of suspended particles. In this case, a rain simulator capable of generating the required eroded material could prove useful. Another option is the application of the suspension directly onto the grassed areas, in which case the sedimentation effect in front of the grass strip would not be utilised (variant 4). Experimental verification would also be necessary for different longitudinal slopes, varying flow rates of discharged suspension, and discharge duration, with the aim of achieving steady-state conditions. Last but not least, these approaches could also be tested on different types of vegetation with varying species composition, age, density, which could provide further insights into the effectiveness of vegetation in reducing soil particle transport.

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Grey water footprint of malting barley production

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Keywords: grey water footprint — micropollutants — pesticides — fertilizers — malting barley

ABSTRACT

Agriculture is the world's main freshwater consumer; it also contributes to its contamination through fertilizers and pesticides. This article focuses on the grey water footprint (GWF) as an environmental indicator assessing the impact of agricultural production on water resources. The study analyses the GWF of malting barley production on an area of 9,674 ha in different regions of the Czech Republic. Special emphasis is placed on including pesticides in the GWF calculation, as their impact on freshwater ecosystems and human health may exceed the impact of fertilizers. The analysis shows that insecticides have the highest GWF, especially deltamethrin, whose GWF is an order of magnitude higher than that of other agrochemicals. The study highlights the importance of including pesticides in future GWF assessments to better assess the environmental impacts of agricultural production and optimize sustainable water resource management strategies. At the same time, the study discusses different approaches to including biologically active substances in grey water footprint models.

INTRODUCTION

Agriculture is the largest consumer of freshwater in the world, accounting for approximately 70 % of total water resource consumption [1, 2]. Intensive agricultural practices, including the excessive use of pesticides and fertilizers, have a significant impact on aquatic ecosystems by leaching excessive amounts of these substances into the aquatic environment. Leaching of nutrients, especially nitrates, into groundwater often contributes to exceeding permitted limits for drinking water. In surface waters, elevated nitrate concentrations promote the growth of phytoplankton, dominated by algae and cyanobacteria. These reduce the dissolved oxygen in water and consequently lead to hypoxia or anoxia (process of eutrophication). These changes cause a loss of biodiversity and can lead to massive mortality of some aquatic organisms [3].

Pesticides, which are applied to protect crops from pests and diseases, leach into soil and water bodies, where they can threaten aquatic ecosystems and human health. Long-term exposure to these substances has been linked to endocrine system disruption, increased risk of cancer, and other health problems [2]. Water contamination by pesticides is particularly problematic due to the persistence of some of these substances, their ability to spread in the aquatic environment, and effect areas at high distances from sites of their application.

Various methods have been developed to quantify the environmental impact of agriculture, including the ecological footprint [4], the nitrogen footprint [5], and the water footprint, specifically the Grey Water Footprint (GWF) [6, 7]. The water footprint [8] consists of three components. The blue and green water footprints represent the physical volume of freshwater consumed for production. Consumption refers to the unavailability of the consumed water to other users in a given catchment and within a given period of time; this distinguishes the water footprint from other environmental indicators that reflect any water use, regardless of its availability to other users. The grey water footprint represents the theoretical volume of water required to dilute pollutants entering water to a level that meets the water quality standards in the recipient at a given location. It also represents the "consumption" of water, as a given volume of water is no longer available to dilute the same pollutant. This indicator allows an assessment of the level of water resource pollution and provides a basis for decision-making on sustainable water use.

The GWF calculation in this study focuses on identifying the amount of water needed to dilute the pollutants, mainly nitrogen, phosphorus, and pesticides, used in malting barley production in the Czech Republic. Previous studies have focused mainly on fertilizers when calculating the grey water footprint of crops, while the impact of pesticides was/and is often underestimated.

Nutrient runoff into surface waters leads to eutrophication and subsequent deterioration in water quality [9]. Nitrogen is highly mobile and its presence in surface and groundwater can cause significant ecological problems. The lack of data on the persistence of pesticides in the aquatic environment and their cumulative impacts on ecosystems makes it difficult to accurately quantify their contribution to GWF. However, a recent study by Yi et al. [10] and this study highlight the need to include pesticides as their environmental impact can be much more significant than that of fertilizers.

In areas with limited water resources and vulnerable ecosystems, the negative impact of contamination may be more pronounced than in regions with a higher capacity of natural systems to dilute pollution. Therefore, monitoring and reducing GWF is of critical importance not only for agriculture but also for downstream industries that use agricultural products as feedstock, such as the food and beverage industry. Quantification of GWF [11] allows the identification of critical points in the supply chain and in the production process. GWF assessment in barley production thus provides important information for environmental policy, agricultural practice, and the downstream food and beverage industry. This approach allows for a more efficient use of water resources and minimisation of their pollution, as well as environmentally sustainable production of food, beverages, and other agricultural products.

The methodology used provides a comprehensive approach to calculating the GWF of malting barley and allows a detailed analysis of the impact of agricultural production on water resources. The results of the study may be key to the design of more sustainable agricultural practices and better management of aquatic ecosystems. GWF monitoring and optimization is an important tool for farmers, industrial producers, and environmental policy makers to minimize negative environmental impacts and increase the efficiency of water resource use.

METHODOLOGY AND DATA SOURCES

This study focuses on the GWF analysis of malting barley grown on an area of 9,674.05 ha in different parts of the Czech Republic, specifically in the districts of Bruntál, Frýdek-Místek, Hodonín, Jeseník, Karviná, Kroměříž, Nový Jičín, Olomouc, Opava, Ostrava-city, Prostějov, Přerov, Rychnov nad Kněžnou, Semily, Svitavy, Šumperk, and Ústí nad Orlicí. To calculate the GWF of malting barley production, detailed data on fertilisers and pesticides used were obtained directly from growers supplying malting barley to Radegast Brewery. A questionnaire was prepared to collect the data, and Radegast Brewery representatives arranged for their suppliers to complete it. The collected data were provided to the study authors in aggregated form, i.e., as an average amount of applied substances per hectare of cultivated area.

The questionnaire survey focused on detailed information on the types and quantities of fertilisers and pesticides applied in the cultivation of malting barley. Based on the products used and their volume, the amount of active substance applied was determined.

To calculate GWF in cubic metres per tonne of crop grown, the Hoekstra and Hung equations [9] and *Water Footprint Assessment Manual* [8] were used:

$$WF_{grey\ li} = \frac{\frac{\alpha \times AR_{li}}{c_{max\ li} - c_{nat\ li}}}{Y}$$

$$WF_{grey\ l} = \max\{WF_{grey\ l,1}, WF_{grey\ l,2}, \ldots WF_{grey\ l,i}\}$$

$$WF_{grey} = \sum_{l=1}^{n} WF_{grey\ l}$$

where:

α	is	is proportion of fertiliser and pesticide losses (%), the so-called leaching factor
AR		amount of fertilisers and pesticides applied to each crop (kg/ha)
c_{max}		critical concentration of the monitored substance from fertilisers and pesticides in the recipient (g/m^3)
c_{nat}		natural (backround) concentration of the monitored substance from fertilisers and pesticides in the recipient (g/m^3)
Y		crop production (t/ha)

The average leaching factor α was determined based on the official *Water Footprint Network* methodology [12]. It has the following values: 0.1 for nitrogen fertilisers, 0.03 for phosphate fertilisers, 0.7 for potassium fertilisers, and 0.01 for pesticides. The leaching factor for pesticides was set at 0.01 due to the lack of detailed data on the soil properties at the monitored sites. The necessary data for calculating the regionalized α factor according to the methodology [12] were not provided.

The difference between the c_{max} and c_{nat} represents the assimilation capacity of the watercourse. For nitrogen, phosphorus, and potassium fertilizers, the following assimilation capacity values were determined: nitrogen 3 g/m^3, phosphorus 0.1 g/m^3, and potassium 5 g/m^3 [12]. For pesticides, the c_{nat} value was set to zero, while c_{max} values were derived from the lowest Predicted No Effect Concentration (PNEC) freshwater values from the NORMAN database [13]. PNEC values are commonly used as c_{max} in wastewater GWF studies [14–17], and can also be used in calculating GWF of pesticides in agriculture [18]. The PNEC values used for this study are listed in *Tab. 2*.

Information on the malting barley Y production in the studied districts was provided by representatives of the Radegast Brewery based on information from a questionnaire survey among farmers. All data are valid for the reference year 2022.

RESULTS

Tab. 1 shows the GWF values of different fertilisers applied to malting barley fields. The highest GWF values were found for phosphorus. *Tab. 2* shows the GWF values for individual pesticides applied to malting barley fields. Insecticides reach the highest GWF values due to their high ecotoxicity to aquatic organisms. The insecticide deltamethrin has the significantly highest GWF, even at very low concentrations. The GWF of deltamethrin is an order of magnitude higher than the GWF of two other important insecticides (gamma-cyhalothrin and esfenvalerate), three orders of magnitude higher than the GWF of fungicides (prothioconazole), herbicides (2,4-D 2-EHE), fertilisers (phosphorus), and four orders of magnitude higher than the GWF of a morphine regulator (trinexapac-ethyl).

Tab. 1. Grey water footprint of nutrients – malting barley

	GWF-N	GWF-P	GWF-K
		[m^3/t]	
Organic fertilizers	18.65	57.96	63.72
Industrial fertilizers	318.40	801.85	318.37
Total	337.05	859.81	382.09

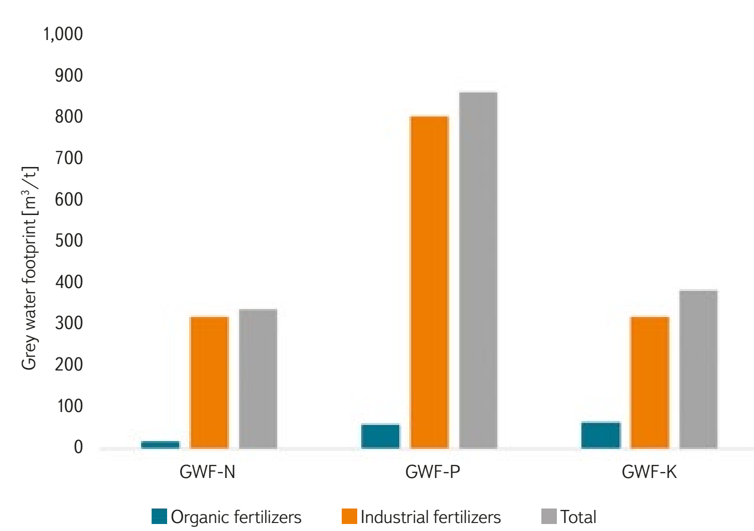

Fig. 1. Grey water footprint of nutrients – malting barley

Tab. 2. Grey water footprint of pesticides – malting barley

Main active substance	Type of pesticide	Quantity applied to the soil [kg/ha]	Amount of erosion into the water [kg/ha]	PNEC (c_{max} - c_{nat}) [mg/m^3]	GWF [m^3/t]
2,4-D -2-EHE	herbicid	276.768	2.768	0.051	714.923
Tribenuron-methyl	herbicid	67.449	0.674	0.100	88.856
Fluroxypyrmeptyl	herbicid	70.357	0.704	0.179	51.780
Diflufenican	herbicid	193.485	1.935	0.010	34.917
Mefenpyr-diethyl	herbicid	37.400	0.374	1.650	2.986
Prothioconazole	herbicid	8.557	0.086	0.330	3.416
Florasulam	herbicid	2.350	0.024	0.062	4.993
Metsulfuron-methyl	herbicid	0.450	0.005	0.010	5.930
2-Ethylhexyl phosphate	herbicid	44.266	0.443	17.100	0.341
Dimethylammonium 4-chloro-o-tolyloxyacetate	herbicid	29.352	0.294	41.300	0.094
Tritosulfuron	herbicid	0.021	0.000	0.750	0.004
2-Methyl -2,4-pentanediol	herbicid	4.787	0.048	822.000	0.001
Prothioconazole	fungicid	1,471.041	14.710	0.330	587.251
Tebuconazole	fungicid	758.947	7.589	0.240	416.594
Spiroxamine	fungicid	598.644	5.986	0.630	125.182
Metconazole	fungicid	368.477	3.685	0.290	167.389
Azoxystrobin	fungicid	134.723	1.347	0.200	88.741
Prochloraz	fungicid	42.643	0.426	1.560	3.601
Proquinazid	fungicid	20.420	0.204	0.180	14.945
Pyraclostrobin	fungicid	147.911	1.479	0.200	97.428
n,n-Dimethyldecanamide	fungicid	42.832	0.428	1.940	2.909
Boscalid	fungicid	13.036	0.130	12.000	0.143
Metrafenone	fungicid	25.455	0.255	4.500	0.745
Deltamethrin	insekticid	3.903	0.039	0.0000017	302,440.814
Gamma-cyhalothrin	insekticid	17.779	0.178	0.0000220	106,461.850
Esfenvalerate	insekticid	12.134	0.121	0.0001000	15,984.660
Cypermethrin	insekticid	1.094	0.011	0.00008	1,800.702
Trinexapac-ethyl	morforegulátor	381.223	3.812	1.100	45.656
Chlormequat chloride	morforegulátor	1,689.165	16.892	10.000	22.253
Ethephon	morforegulátor	801.548	8.015	4.700	22.467
Prohexadione-calcium	morforegulátor	40.265	0.403	10,000.000	0.001
1,1-Dimethylpiperidinium chloride	morforegulátor	2.030	0.020	260.000	0.001029

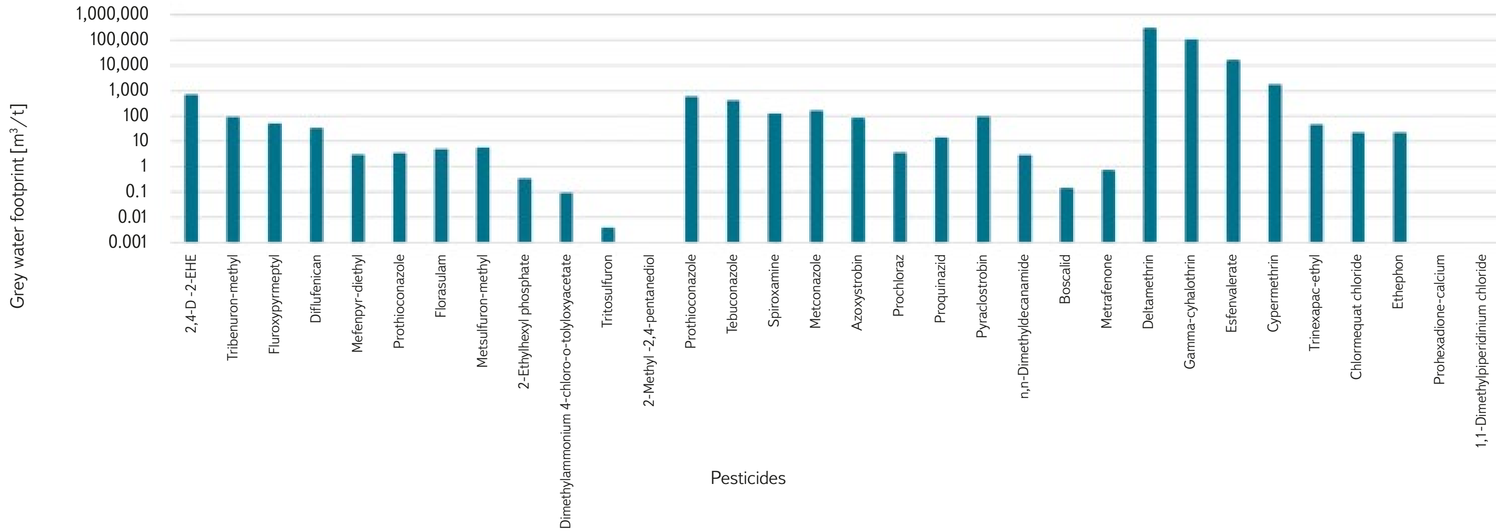

Fig. 2. Grey water footprint of pesticides – malting barley

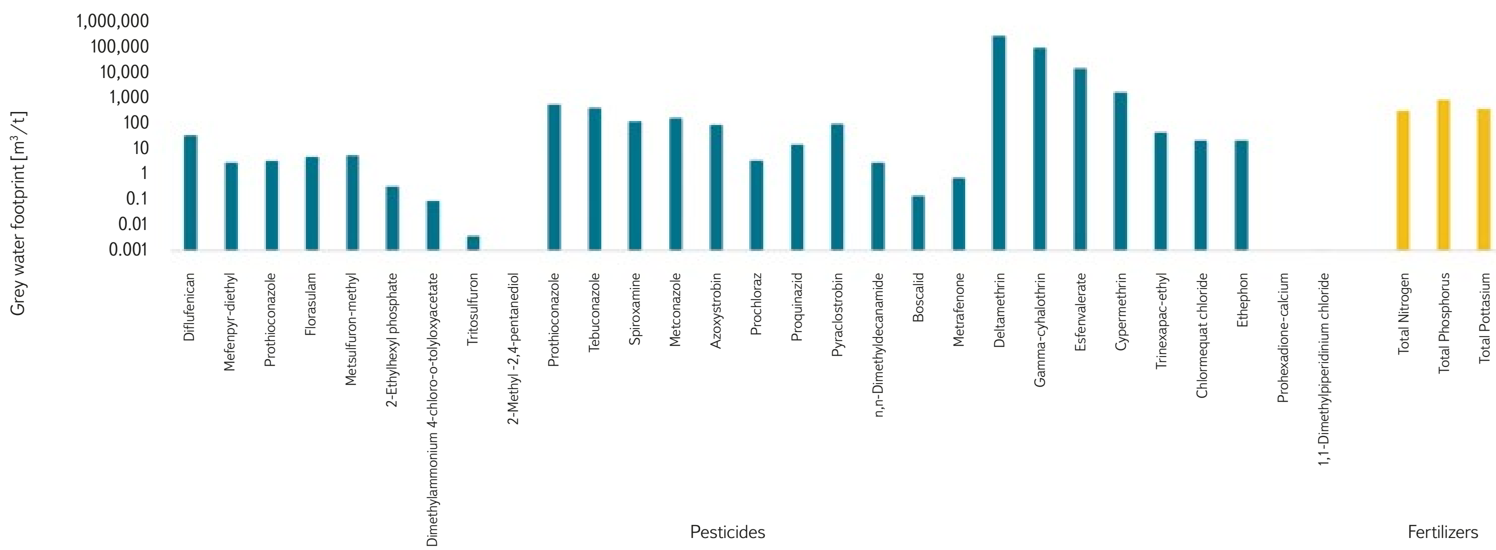

Fig. 3. Grey water footprint of malting barley production

Fig. 3 provides summary values of the GWF associated with fertiliser and pesticide use in malting barley production. Insecticides show the highest GWF values, which is related to their high ecotoxicity to aquatic organisms. Among them, deltamethrin dominates, with a GWF approximately one order of magnitude higher than the other two major insecticides (gamma-cyhalothrin and esfenvalerate). Also, it is three orders of magnitude higher than the GWF of fungicides (prothioconazole), herbicides (2,4-D 2-EHE), and phosphate fertilisers, and even four orders of magnitude higher than that of a morphoregulator (trinexapac-ethyl). Although only small amounts of deltamethrin have been applied, its overall impact on aquatic ecosystems is most significant. The total GWF associated with malting barley production amounts to 302,440.814 m^3/t, with insecticides with the active substance deltamethrin accounting for the most significant part of the pollution.

DISCUSSION

While the application of fertilisers and pesticides has a noticeable positive effect on boosting crop yields, the massive use of these substances causes environmental contamination both locally and globally. Studies published to date have generally focused on GWF caused by fertilisers, which are generally used in large quantities. Pesticides have not been included in most studies, both because of their relatively small quantities (compared to fertilisers) and because of methodological issues associated with their inclusion in the GWF model.

Pesticides usually break down very slowly; their residues remain in agricultural soil for many years after application. Their negative effects on water quality are evident at significantly lower concentrations than those of nutrients. Humans exposed to water poluted with pesticide residues are at risk of diseases such as cancer, endocrine disruption, etc. Aquatic ecosystems are even more sensitive to the effects of these substances.

The results described above show that for a correct assessment of the GWF of crops, it is necessary to assess not only the GWF of fertilisers but also the GWF of pesticides. Based on current knowledge, crop GWF studies can no longer be considered representative if they only focus on the GWF of fertilisers. There is a need to compare the GWF of fertilisers with the GWF of pesticides in future crop GWF studies is evident. Without such a comparison, the results are incomplete and may be misleading.

On the other hand, it is important to note the possible limitations of our results. The first limitation is the application to a single crop species grown on 9,674.05 ha. The amount of fertilizers and pesticides applied and their composition vary depending on the crop grown, soil characteristics, as well as on management practices. These variable factors influence the GWF value, as demonstrated in the study by Borsat et al. [19]. The second limitation is the use of a constant leaching factor α, which is in accordance with TIER 1 according to Franke et al [12]. The use of a constant leaching factor α represents a certain simplification of the heterogeneous conditions prevailing in agriculture. Such a simplification is therefore appropriate for large-scale studies or, in the absence of basic data, for more detailed approaches to the expression of the leaching factor (TIER 2 or TIER 3). In our case, it was used due to the lack of supporting information for the application of a more detailed solution.

A final simplification that we used due to the lack of detailed data is the composition of the individual mixtures applied to each field within the study area. The data obtained from individual farmers and provided by the Radegast Brewery representatives only gave the total amounts of the product applied in the area of interest, not in particular fields. Therefore, we considered the application rate applied to the entire area of interest of 9,674.05 ha. The mixture of products shown in *Tabs. 2* and *3* thus represents a kind of 'common average mixture' used in production.

The problem in determining the GWF of pesticides lies in the common application of pesticides in the form of mixtures of different active ingredients. All pollutants entering water from human activities are mixtures of several substances. The *Water Footprint Assessment Manual* [8] assumes that the individual substances in the mixture do not interact with each other, and the GWF is determined by the substance with the highest value. However, this assumption of the GWF model is often not met in reality. When different bioactive substances are mixed, they interact with each other, and their toxicity and impact on the receiving water body change depending on the mixture composition. Therefore, some researchers have proposed alternative approaches to address GWF mixtures.

One approach is to modify the GWF model. Paraiba et al. [18] proposed a model that assumes that the toxicity of a mixture is the sum of the toxicities of each substance in the mixture. De Lavor Paes Barreto et al. [20] compared such an approach with the original approach described in the *Water Footprint Assessment Manual* [8] and found that the model proposed by Paraiba et al. [18] is usually more precise. This is a logical conclusion, considering that in the model, each additional substance added to the model mixture will increase its toxicity.

Another approach to addressing mixtures is to include the self-purification capacity of the watercourse. For example, the GWF study on urban wastewater [21] identified ammonium nitrogen ($N\text{-}NH_4^+$) as the substance most often determining GWF. In rivers, ammonium nitrogen is rapidly oxidized to other forms of nitrogen, however, the *Water Footprint Assessment Manual* GWF model does not account for this fact. Therefore, some researchers include the self-purification process directly into GWF models [22, 23].

A yet different approach to addressing GWF of mixtures can be found in the L'Oréal product eco-design article [24]. Their methodology is based on the use of techniques used in LCA, i.e., on the principle of additivity of the effects of each component in proportion to its concentration in the formula.

The above-mentioned uncertainties of the solution, as well as the different approaches to GWF by different authors, highlight the need for further research on GWF. In our view, this research should focus on three areas:

- The first area is the identification of substances that may determine GWF. Our studies of malting barley GWF (this paper) and micropollutants in treated urban wastewater [14] have shown that commonly monitored pollutants may not be (and often are not) the most critical ones for GWF determination. Thus, the selection of non-representative pollutants leads to a systematic underestimation of GWF values. A number of research studies in different water-related fields are needed to find relevant pollutants for different sectors and water uses.
- The second area deals with mixtures in GWF models. On the one hand, the "independence" of the water footprint values from external influences must be maintained. The water footprint is one of the environmental indicators that describes the behaviour of the assessed system. An indicator whose value would change without changing the assessed system itself is not well set. On the other hand, issues related to new, so-called emergent pollutants, which are often bioactive substances and behave differently in different mixtures, need to be adequately addressed.
- The third area where we consider the current state of knowledge to be incomplete is in assessing the GWF sustainability. We do not consider approaches that introduce a self-purification process into GWF models to be appropriate practice. The self-purification capacity of the aquatic environment is independent of the product systems assessed by GWF. Therefore, the water self-purification capacity should not be included in a GWF model. A modification of the sustainability assessment seems to be a more appropriate solution. The current system, described in the *Water Footprint Assessment Manual* [8], compares GWF values with available sources to dilute pollution using actual runoff from the catchment. Thus, this approach compares the runoff in a particular catchment with the dilution water needs in different parts of the assessed catchment. This can lead to an overestimation of the discharged pollution impact due to the neglect of the self-purification capacity in the aquatic environment.

CONCLUSION

This study confirmed that GWF is an important indicator for assessing the environmental impacts of agriculture, and that all applied substances, i.e. not only fertilisers but also pesticides, should be included. In malting barley production, the insecticide deltamethrin had the greatest impact on water resources. Due to the high ecotoxicity of pesticides and their long-term persistence in aquatic ecosystems, it is important that future studies include a detailed analysis. Local conditions such as climatic factors, soil types, and water availability must be considered in GWF assessment. The implementation of measures to reduce GWF, such as optimising the use of agrochemicals and innovative technologies in agriculture, can contribute significantly to a more sustainable use of water resources and environmental protection.

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Declaration of conflict of interest

The representatives of Radegast Brewery and the related companies had no influence on the results of the study. The second author is part of the TGM WRI management, which publishes the VTEI journal, and chairman of the VTEI journal editorial board. However, these facts had no influence on the support provided for the writing of the article or on the acceptance of the article for publication in the VTEI journal.

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Interview with Ivan Tučník, Head of Group Sustainability Asahi Europe & International

How does Radegast Brewery work with water when brewing beer, what are the brewery's aims regarding sustainability, to what extent does the brewery use the latest technological trends in its production, and why do we like bitter beer in the Czech Republic? The offer to be interviewed for our VTEI journal was accepted by Ing. Mgr. et Mgr. Ivan Tučník from Asahi Europe & International; in the Czech Republic, the company owns Radegast, Plzeňský Prazdroj, Velkopopovický Kozel, and many other breweries across Europe.

Mr. Tučník, Radegast has long been one of the world leaders in water conservation in beer production. How far can you go in water conservation?

A lot depends on how far you set your limits. There are breweries in the world that can go somewhere between 1.6 and 1.7 litres per litre of beer produced. The question then becomes how much each tenth of a litre saved is worth. You get to the stage where, to save water even more, you need to use, for example, reverse osmosis technology, which is not only an energy-intensive process, but also generates hazardous waste. For us, we have set this limit at a level that we have mapped out, but above all at a level that we are able to apply in production. In addition, there is not a single brewery in the Czech Republic that would be at risk of water shortage, so there is no need to lower this limit even more. Nevertheless, we are trying to find ways of maximising the efficiency of water use without having to treat it in such a complex way. In fact, we have already reached our limits. This is evidenced by the fact that we have made no progress in reducing consumption in the last two years. Rather, our aim is to maintain this threshold.

How are the other breweries in the Asahi Group doing with water conservation? To what extent is Radegast a model for other breweries?

We are fortunate that there are people at Radegast who are interested in water conservation during the brewing process and who place great emphasis on this topic. What is important, however, and where Radegast has an undeniable advantage over, say, the Pilsner brewery, is the complexity of the operation and beer production itself. The more types of beer you brew within one brewery, the higher the water consumption. Every time you start brewing a different kind of beer, it means a complete sanitization of the entire system, which is reflected in the aforementioned water consumption. Radegast Brewery has the advantage that there are not so many types of beer brewed here compared to other breweries in the group. It is definitely an inspiration for others, but with the small footnote that not everything that is possible at Radegast is possible elsewhere. Radegast Brewery is the absolute leader in terms of water consumption, not only in comparison to the entire group of breweries belonging to Asahi, but also globally.

To what extent can you use this know-how with other breweries?

A lot. We always try to share our experience with each other, and if something works, we try to apply it further. However, we have to take into account the local context every time. The extent to which solutions can be replicated is sometimes limited. We operate four breweries in the Czech Republic and Slovakia, with a total average consumption of 2.8 litres of water per beer. Our ambition is to get to 2.75 without using energy-intensive technologies.

Surely, such low water consumption also works as a good promotion point…

Of course, this is perceived very positively by our consumers. This approach did not start as a marketing concept coming from an idea of a marketing team or a PR manager. The whole current approach has its foundations from the bottom, and other things have built on top of that over time. There are other activities that we do at Radegast.

One of the proofs is not only water conservation in operation, but also management of rainwater. What is Radegast's approach in this respect?

We have a rather unique way of treating rainwater directly in the brewery. We call it the brewery ponds, which is a biotope that we built here about twenty years ago. It is a system of several ponds through which we treat rainwater from the brewery before we discharge it into the Morávka river. In addition, we have a grant programme through which we support community projects around the breweries, such as Beskydy landscape management.

The brewery's cooperation with Forests of the Czech Republic is well known…

Yes, cooperation with Forests of the Czech Republic is basically a way to extend our approach to landscape protection to the whole country. We are currently preparing an evaluation of the effect that these activities have had so far on the total volume of water retained.

How financially demanding are these activities?

These activities come at a cost, but they are very important to us. And they help the brand. Our initial communication of this initiative was very cautious, mainly because the use of sustainability in communication is not so widespread in the Czech Republic. We put a lot of emphasis on being able to prove all our activities and back them up with valid research and robust methodology. Therefore, we collaborate with universities and research institutions.

You want to be water neutral by 2030.

Through projects that build pools and restore wetlands, we plan to retain the same volume of water in the landscape as we use in operations. By 2030, we aim to have enough similar projects in place to ultimately retain a volume of water equivalent to around 570 million litres, which is our annual consumption at Radegast Brewery at current operating levels.

Where are your activities expanding or heading next?

Cooperation with Forests of the Czech Republic in building pools and restoring wetlands is probably what is most visible now. However, we are also focusing a lot of effort on cooperation with farmers. We are aware that the Czech Republic is beginning to struggle with drought, which we see, for example, in our barley and hops suppliers. We feel that working with our suppliers from a value chain perspective is the most natural for us. Let us take the example of our most famous hop variety – Žatec semi-arid red. If this variety were to disappear or if its production were to be dramatically reduced, it would have a major impact on the Czech brewing industry and on the quality of Czech beer. I like to compare hops to spices in food; you do not need a lot of it to make beer, but it is absolutely essential to the taste and quality of beer. Without Žatec red, our Pilsner lager would not be what we are used to.

Please describe to our readers how such cooperation with farmers works.

To give you an example, in our three-year research project "For the Hops" we were trying to understand how hops themselves manage water and how they react to external stimuli. At six sites, we installed devices to collect and assess meteorological data, including data on soil processes. We measured soil moisture and temperature at twelve different depth horizons. At the same time, we monitored the development of the hop garden using time-lapse cameras. We went so far as to use sensors on selected plants to monitor sap flow, stem shrinkage, and assess the stress level of the plant in response to water and temperature. The result is the first software solution aimed at efficient irrigation of hops. I was surprised myself how little we actually know about hops.

How will you use the results of this research?

Last year we tested this solution on twenty-eight hop farms, roughly one tenth of Czech hop growers. We are now in the process of building a network of weather stations across all hop-growing regions, and we will offer our solution to other hop growers in the Czech Republic. We are collaborating with three technology start-ups, including a Czech software company, and professionally with our hop-growing institute in Žatec. It is quite a complex collaboration of about forty people living on three continents and in about fourteen different cities, which is sometimes a bit difficult to coordinate.

That sounds very interesting. Can you tell us how much interest there is in this product?

There is definitely interest, and what I would like to point out is that we do not ask anything from the growers in return. We realise that a farmer's decision to grow hops is not a year-to-year decision, like other crops. It is a decision for several years, often decades. Our offer is essentially a service to a small community of hop growers, which is an important raw material for us.

In addition to the technology initiatives mentioned above, we have a project where we are testing regenerative hop growing, which basically means that we are focusing on growing crops in the inter-row, which is normally ploughed and nothing grows there. We are in our third year of cooperation with the Czech University of Life Sciences on about twenty-five hectares. We are looking at the effects of intercropping on yield, quality and soil, but also on the amount of organic matter in the soil, water-holding capacity, cooling of the soil during hot days, and many other things. We have observed that we can cool the soil by two to five degrees with the appropriate choice of intercropping.

How does Radegast Brewery manage wastewater? Do you use any higher levels of water purification or recycling?

We do not want to have this tunnel vision where we are only dealing with water and we do not care that we will create problems elsewhere. The associated high energy consumption is not just about cost, but also about carbon footprint. We are trying to balance the different parts of the process that we are focusing on, and our possibilities are therefore limited. So, we do not really see this as a path for breweries in the Czech Republic that we want to apply intensively and on a large scale because in our conditions it does not really make sense to us at the moment.

Could the energy needs be met by solar power?

In breweries where it was possible, we have solar panels on the roofs. In Nošovice we have them on the automated warehouse, which makes it basically energy neutral, but in terms of the total consumption of the brewery it is about three per cent. Therefore, for us, it is more of a supplement, not a final solution. However, we were able to find that in Slovakia. While we are still looking for a partner in the Czech Republic, in the east of Slovakia, about fifty kilometres from our brewery, we have just launched the largest greenfield solar park as part of the so-called VPPA project, which supplies electricity to the grid which is then used in the brewery.

Regarding the quality of water at the inlets, do you use raw water from your own sources or water from the supply system?

We have water treatment plants in all our breweries to ensure the parameters we need. In Nošovice we have three wells of our own, but in most breweries it is a combination. In Pilsen, for example, we have our own 100-metre-deep wells that we use for the beer itself, but we take surface water from the system for all the technical processes around it, because in that case, using groundwater would be wasteful. By the way, this is also why Pilsner lager is not brewed anywhere else but Pilsen; whether you have it in Tokyo or Washington, it always comes from the same brewery and the same brewhouse. Otherwise, we have a multi-stage system of controlling the quality of water and its parameters. We are not using carbon filtration yet; our water treatment is more parametric in terms of mineral content, de-ironing, etc. And in the next control stage, for example in Želivka, its quality is monitored by live trout (*laughs*). The quality of residues of agro-preparations and pesticides, whether in hops or barley, is monitored both by the agronomist and by us when we receive the goods, and if the analyses do not come out well, we reject the batch and these substances do not get into the beer.

How about buying these products directly from certified organic farms?

If we wanted to convert everything to organic at our volumes, it would not be realistic. For example, there are only maybe three or four organic hop fields in the Czech Republic, which might be enough for a microbrewery, but not for us. We see the future mainly in improving the soil for growing hops and barley, both in terms of quality and carbon content and the water-holding capacity. This is one of the things we have committed to at Radegast. We now have two research projects on this and then, based on the results, we will look at ways to scale this across our suppliers.

Is climate change having any effect on the quality of our hops?

Primarily, it manifests itself in an increase in yield fluctuations and content of bitter substances in the hops. For example, about three years ago, we had the best harvest in the last century; the year after, we had the worst harvest since the 1960s. So it is about reducing your predictability, which has implications for medium-term commitments with our partners. By having our recipes standardised for alpha bitters, essential oils and other things, we are able to compensate for that so you do not taste anything in the actual beer. However, it may lead to the fact that you need twice as many hops as the year before to achieve the same quality of beer, because the concentration of the substances in question is lower in that particular harvest. And if you combine the variation in quality and quantity, the year-to-year variation is very noticeable.

The worst-case scenario is a poor harvest with low alpha acid content. However, it cannot be said that a warmer and sunnier year – as in the case of winemakers – means better hop quality; there are many more factors at work. We generally have problems with a higher number of tropical days in a row in a longer rainfall-free period. To some extent, this can be compensated for by lowering the soil temperature in regenerative cultivation and the plants in the undergrowth. The results of our research will give more clues.

What about the other important raw material, barley?

Climate change does not have that much impact on yields, but it does have an impact on the malt quality; we need it to have a certain ratio of nitrogen, protein, and other substances to be malleable. From the grower's point of view, it is actually a bit of a lottery because you do not know until the harvest whether you are going to get malting parameters in barley or whether you will have to sell it as feed. And the difference in the purchase price is often double. This risk is leading to a lot of growers moving away from malting barley – the malting barley area has halved in the last 25 years. Within our research, we are looking for ways to stabilise quality for growers. We are looking at changes in sowing practices and improving soil quality, and we believe that this could be the way forward – somewhere between organic and conventional production.

Is it possible to calculate how much barley is used to make beer compared to making bread? Beer is called liquid bread...

We need about one hundred and fifty thousand tonnes of barley per year, which is about thirty thousand hectares. If we consider that the arable land in the Czech Republic is two million hectares, this is not an entirely insignificant amount. Otherwise, about one million tonnes of malting barley are produced in the Czech Republic every year, of which we account for about fifteen per cent. It is also a very important export commodity. We export both barley and malt to many European countries. As the Czech Republic, we are fully self-sufficient in its production.

And what about the declining trend of "going to the pub" in the Czech Republic?

We sell about thirty-five per cent of our beer to pubs and the rest is domestic consumption. We are doing what we can to maintain this ratio. We cooperate with pubs a lot and invest about four hundred million a year in them to help them remain an attractive place. We invest in repairing their facades as well as their interiors and toilets, so it is not just about providing them with taps and glasses. We are particularly mindful of the quality of beer, so we invest in training so that the pub staff know how to treat the beer well. Having a beer in a pub is about the experience, there has to be some added value – a properly chilled glass, a well-adjusted tap. As we say: the brewer brews the beer and the innkeeper makes it. The quality of the beer and the quality of the tapping is about half and half.

And what about the phenomenon of Czechs not drinking classic 10- and 12-degree beers so much and turning to modern beers?

I am going to surprise you. If I take bottom-fermented beers in the sense of lagers and draught beers, their production is definitely above ninety per cent. In reality, Czechs still want bottom-fermented beer and the general trend is more lager than 10-degree beer. The Czechs are very conservative in this respect and their consumption is built on bottom-fermented beers, especially Pilsner-type beers. In cities, however, there is more experimentation with other types of beer.

Czech breweries are competing to see who can come up with the most bitter beer. Why do you think Czechs like bitter beer so much, whereas Western Europe, for example, tends to prefer sweet, malt, and sour beers?

When we look at our most popular brands and the overall character, beer in the Czech Republic is generally more hoppy than in Western Europe; it is a Czech specificity. In Slovakia, we also see a leaning towards more bitter and more hoppy beers. It is a historical development and a long-term local habit. Basically, since the emergence of Pilsner Urquell as the benchmark for bitter beer, which has a character built on local Žatec hops, it has made its way into Czech beer culture. In recent years, however, we have also seen sweeter and more sour beers that have their famous predecessors elsewhere in the world. It is also about what one likes, whether the beer is well brewed, treated, and properly tapped. It is clear that the younger adult generation tends to prefer less bitter beers, and we are meeting that with our range, led by Proud beer.

Are preferences also changing in bottle sizes and packaging?

There is much more demand for smaller packages, which is also related to modern trends in reducing alcohol consumption. That is why we now offer Radegast in one-third-litre returnable bottles. In terms of packaging type, cans have been growing in popularity for a long time. We are happy to stick with returnable bottles because they are great from an environmental point of view; we fill each bottle on average twenty-six times and we have a ninety-eight per cent return. The life of a bottle is approximately seven to eight years. If a bottle is rotated so many times in the system, it is the most environmentally friendly way to package beer ever. Of course, the older bottles may be a bit worn, but we check thirty parameters of the bottles for quality before filling them, and if one of them does not fit, the bottle is discarded and goes for recycling. Conversely, the worst option for beer in terms of carbon footprint and all the other things is if you throw away a newly produced bottle of beer after drinking it. This means a non-returnable bottle because it is heavy from a distribution point of view and it is energy intensive to produce. For us, it costs three to four times more than a returnable bottle. Four years ago, we also reduced our ecological footprint by replacing the aluminium and plastic part of the Pilsner beer label with paper. Even such a detail has a significant impact on the ecology of the operation and only underlines our long-term path and vision.

Mr. Tučník, thank you for taking the time to talk to us.

Ing. Josef Nistler
RNDr. Tomáš Hrdinka, Ph.D.

Ing. Mgr. et Mgr. Ivan Tučník

Ing. Mgr. et Mgr. Ivan Tučník, born on 26th February1986 in Považská Bystrica. He graduated from Masaryk University in Brno with a Master's degree in International Relations, Business and Management, and European Studies. Prior to joining Asahi Europe & International and Plzeňský Prazdroj (since September 2017), he worked as a consultant and communications manager at MAKRO Cash & Carry, Bison & Rose, and AMI Communications.

Fig. 1. In the last two years, the Radegast brewery has built 60 pools in 19 locations across the Czech Republic

Radegast beer gave its word: it will return water to the landscape

Radegast brewery, part of Plzeňský Prazdroj, has started a campaign for water. By 2030, it will retain more water in the Czech landscape than it consumes.

Radegast is one of the breweries with the lowest water consumption in the world. With 2.3 hectolitres of water per hectolitre of beer, it ranks among the absolute world leaders. Water consumption includes water for the complete production of beer, sanitation of equipment, cleaning in the brewery, water in the toilets, etc. The Nošovice brewery has achieved this success through technological innovations that have reduced the brewery's water consumption by 44 % over the last 15 years. With such low consumption, however, finding further savings is not easy.

For this reason, Radegast focuses on projects that promote water retention in the landscape. As part of these, since 2015, Radegast has invested more than CZK 8 million in the construction and restoration of water features such as pools and wetlands, including the brewery's ponds in close proximity to the brewery, which help to drain rainwater from its premises. All projects are carried out by the brewery in collaboration with experts in the field.

New pools around the country

Cooperation with Forests of the Czech Republic and support for the development of new pools are the key steps Radegast is taking to achieve its goal. In 2023, the brewery and Forests of the Czech Republic built 29 retention pools and added 30 more across the country in 2024. The common goal is to improve the water regime in the landscape and promote biodiversity. The pools will not only retain water in the landscape but also create a habitat for aquatic plants and animals.

Radegast Brewery's cooperation with Forests of the Czech Republic was established in 2023. Radegast has long been committed to minimising water consumption at the brewery, while supporting a number of projects contributing to water retention in the landscape. Forests of the Czech Republic, which has its own "Giving Water Back to the Forest" programme, welcomes any project supporting the adaptation of forests to climate change. Thus, they view cooperation with Radegast positively as it has developed naturally and is based on the same values.

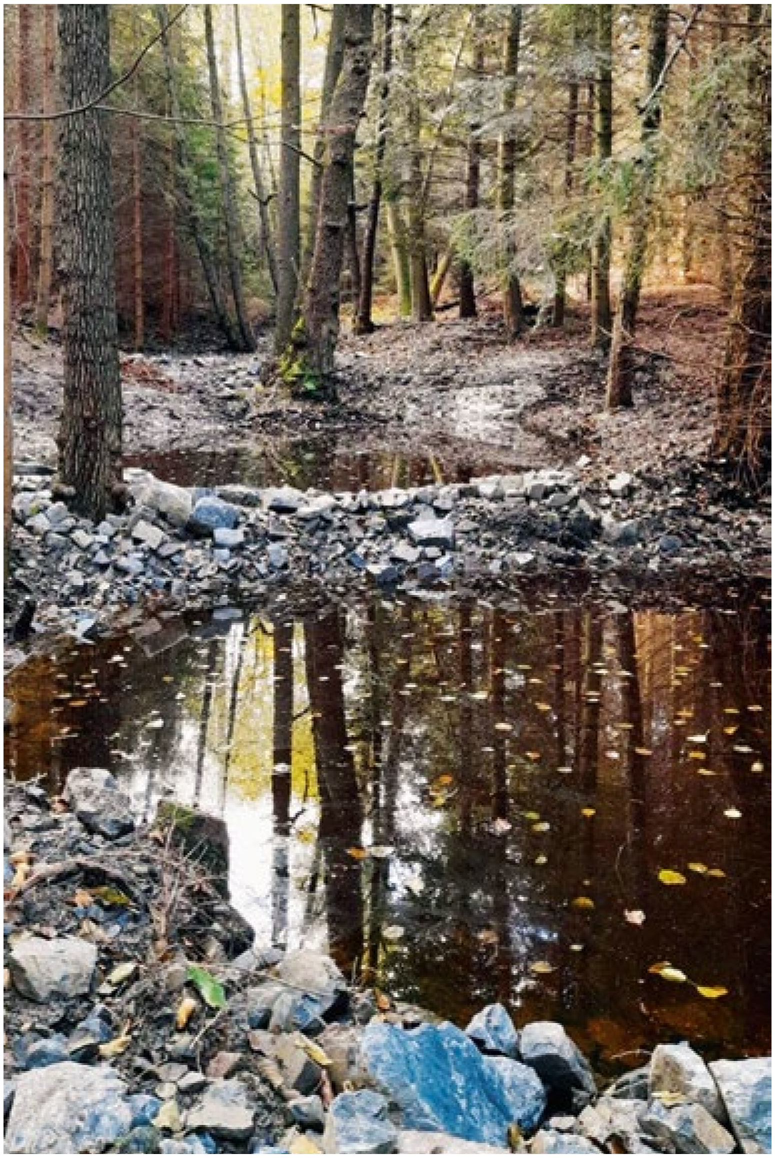

Fig. 2. Pools help retain rainwater and also contribute to flood protection

Fig. 3. By retaining water, pools contribute to supporting biodiversity

Fig. 4. Within about a year of construction, a natural habitat for plants and animals will be created around the pool

Wetland management in the Beskydy mountains

Long-term cooperation with the ČSOP Salamandr organization, focused on the protection and restoration of the Beskydy wetlands, is another example of Radegast supporting biodiversity and water retention in the landscape. Wetlands mitigate the effects of drought and torrential rains and are a refuge for a diverse range of plant and animal species.

Campaign for water 2030

Radegast brewery needs around 570 million litres of water annually for beer production and its overall operations, and it is therefore supporting the development of new pools and river meanders to meet this commitment by 2030. It is also focusing on another key pillar of its commitment – supporting regenerative agriculture, which helps to improve the soil's ability to retain water and strengthen the overall resilience of the landscape to drought. The savings from individual water retention measures will be calculated by the T. G. Masaryk Water Research Institute as the project's guarantor, which will assess the impact of these projects according to its methodology and gradually quantify the volume of water retained in the landscape.

Radegast brewery will thus be the first brewery in the Czech Republic and one of the first in the world to return water to the landscape – where it belongs.

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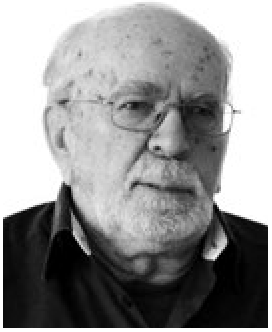
An informative article that is not subject to peer review.

Ladislav Kašpárek left us forever



It is with great sorrow that we announce that our colleague and friend Ladislav Kašpárek has passed away. It happened unexpectedly – this loss has affected all of us even more, as we saw him in good spirits and health in the middle of March, as he enthusiastically presented his new, and ultimately final book at a two-hour signing session at T. G. Masaryk Water Research Institute.

He was born on 27th June 1943, in Čáslav – on the name day of Ladislav, which also determined his first name. However, his friends and colleagues always called him Slávek. He spent his youth in Čáslav and the surrounding area, in the region of the Iron Mountains and the rivers Doubrava and Klejnárka. His father worked as a water management specialist, so he was introduced to this field literally from childhood – water management plans, literature, and direct contact with experts were a part of the family's daily life. In the 1950s, he spent summers with his family at the Pařížov reservoir, where his lifelong connection to canoeing was born.

Despite the educational reforms of the 1950s, he graduated from secondary school at the age of 17 and enrolled at the Czech Technical University (CTU) in the field of hydraulic engineering. After his military service, he began working at the Directorate of Water Management Development (now Water Management Development and Construction, VRV), from where he moved to the Hydrometeorological Institute (HMI) in 1968. At the time, both institutions were located almost next to each other and complemented each other professionally.

In 1969, he got married, and shortly afterward, his sons Jan and Pavel were born. At that time, he was starting out in the field of hydrological forecasting. He transformed part of his family house in Prague-Suchdol into a small department of what was by then the Czech Hydrometeorological Institute (CHMI) – the so-called 'hydrology laboratory' – making him one of the first pioneers of working from home.

In 1979, he became head of the Department of Regime Information. When the CHMI relocated to Komořany in the second half of the 1980s, Ladislav Kašpárek decided to move to the Water Research Institute in 1987 (later T. G. Masaryk Water Research Institute – TGM WRI). At TGM WRI, he served as a senior research scientist and Vice-Chairman of the Institute's Council, and held the positions of head of the Department of Hydrology and head of division. He spent the remainder of his professional career at TGM WRI. At the same time, he continued to provide methodological oversight in the field of regime hydrology at CHMI.

He earned the scientific degree of CSc. in the field of hydrology and water management in 1987 at the Czech Technical University.

Ladislav Kašpárek had vast professional expertise – from stochastic approaches in hydrology to water balance, evaporation, drought, groundwater, and the impacts of climate change. Among his significant recent work was leading the hydrological component of the project "*Reassessment of Groundwater Resources*". He was not only skilled at designing solutions but also at explaining them clearly and often with a smile, even to non-experts. He was a popular consultant for students and colleagues at Charles University, the Czech University of Life Sciences in Prague, and other institutions.

A special focus in his work was placed on extreme hydrological events, particularly floods. When mapping the flood of 1981, he became interested in history, especially the forgotten flood of 1872, which he later researched in detail. This interest culminated in his final monograph *Historical Floods on Rakovnický Stream*, which concluded his long-term research on extreme events, especially in the Rakovník region. He always emphasized the importance of thorough documentation so that these events could be accurately captured in statistical analyses and models.

His last public appearance – the presentation of the aforementioned book on 18th March 2025 – was the culmination of many years of work and personal dedication. Although in recent years he faced great personal trials, including the sudden loss of his firstborn son Jan, he remained a humble, kind, and wise person who was able to offer advice, encouragement, and lift others' spirits.

With the passing of Ladislav Kašpárek, we lose not only an outstanding expert but, above all, an extraordinary person who left a deep mark on both hydrology and in our hearts.

Goodbye, Slávek. And thank you.

Friends and colleagues from TGM WRI and CHMI

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DLOUHÉ STRÁNĚ

Dlouhé Stráně pumped-storage hydroelectricity power station (DS PSH) sparked much debate during its construction, which was completed in 1996. And it continues to do so to this day. In any case, it represents a unique water and energy structure with no equivalent in the Czech Republic or even in Europe. Let's set aside the technical specifications of this hydroelectric facility and focus on a few interesting facts. The inundation area of the lower reservoir of DS PSH (*see photo*) begins at the confluence of the Divoká Desná and Česnekový streams, with both closing profiles equipped with limnological stations. Last year, the historic weir on the Česnekový stream underwent reconstruction; such weirs are characteristic of the upper part of the basin. These massive structures can also be found on the Sviní stream, Velký and Malý Dědův streams, Zámecký stream, and other sites, including Hučivá Desná above the lower reservoir. They highlight the technical skill and perseverance of the people of the Jeseníky region at that time. Similar structures can also be found, for example, in the upper parts of the Moravice river basin. An interesting site is Zámčisko, where the aforementioned streams converge, and the grandeur of the valley is enhanced by rocky outcrops. Above this confluence point, the Hučivá Desná flows through Medvědí důl, which is a typical mountain stream valley with rugged terrain and an increasing presence of mountain and boreal fauna and flora. Interesting species include, for example, boreal owl (*Aegolius funereus*) and Siberian hawkweed (*Crepis sibirica*). The very rare spotted gentian (*Gentiana punctata*) can also be found here.
DS PSH also played a positive role in mitigating peak flows of the Desná river during the September flood of last year. DS PSH strives to further optimize the operation of the hydroelectric facility from the hydrological and water management point of view, collaborating, among others, with the Czech Hydrometeorological Institute.

Text and photo by doc. RNDr. Jan Unucka, Ph.D.

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