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Environmental impact of an anthropogenic groundwater temperature hotspot

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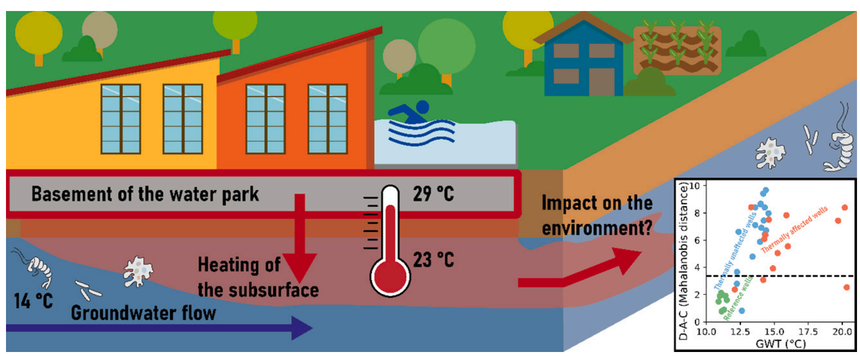
HIGHLIGHTS

- Water parks contribute to subsurface warming
- The "Aquadrom" causes a local thermal anomaly of over 9 K
- Neither hydrochemical nor microbiological parameters show deterioration due to heat
- Groundwater quality at the test site is not affected by subsurface warming
- The aquifer is only inhabited by a small number of fauna due to oxygen depletion

GRAPHICAL ABSTRACT



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ABSTRACT

Heat emitted by buildings and other infrastructure accumulates in the subsurface. This additional heat can cause a pronounced shift in thermal boundary conditions of the important groundwater ecosystem. Shallow groundwater systems in Central Europe are often inhabited by communities of fauna adapted to cold and stable conditions as well as microorganisms, whose activity is dependent on ambient temperatures. At a local groundwater temperature hotspot of up to 23 °C, caused by a water park, we assessed the environmental impact of this thermal alteration on the shallow groundwater system. The results show that the overall groundwater quality at the site is influenced by anthropogenic land use, compared to wells in a nearby water protection zone. However, neither hydrochemical nor ecological characteristics of groundwater from wells in the vicinity of the water park indicate any significant dependence on temperature. Hence, we conclude that in this eutrophic and anoxic aquifer moderate heat stress does not lead to significant alterations in terms of hydrochemistry as well as microbiological properties. Due to the overall low oxygen concentrations (<1 mg/l), stygofauna is present only occasionally and cannot be used as bioindicators. These results have to be verified for other aquifer types and would benefit from a more in-depth analysis of microbial community composition.

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1. Introduction

The shallow subsurface is increasingly utilized by geothermal applications, altering groundwater temperatures (GWT) especially in densely populated areas (Epting et al., 2017; Menberg et al., 2013; Vienken et al., 2019). Beside these wanted interventions, many other unintended heat sources cause increased subsurface temperatures (Noethen et al., 2022). The most prominent and impactful is the anthropogenically induced climate warming as atmospheric temperatures delineate the natural state of shallow GWT (Benz et al., 2024; Hemmerle and Bayer, 2020; Loeb et al., 2021). Typically, shallow subsurface temperatures can be approximated by the annual average air temperature + 1–2 K, depending on hydrogeological factors as well as land cover (Benz et al., 2017a, 2017b; Molnar, 2022; Stauffer et al., 2013). Hence, a rise in air temperatures directly impacts GWT with a temporal delay. Additionally, other heat sources, such as buildings with basements (Benz et al., 2015a; Makasis et al., 2021), tunnels (Epting et al., 2020), and district heating networks (Tissen et al., 2021), contribute unintentionally to subsurface warming by various processes, like conductive heat transport from underground car parks (Noethen et al., 2023), leakage of warm waste water in sewage networks (Benz et al., 2015b), or biochemical heat generation in municipal solid waste landfills (Coccia et al., 2013). This heat can accumulate and ultimately form subsurface heat islands of typically up to 7 K in cities where the underground space is heavily used (Hemmerle et al., 2022; Previati and Crosta, 2021).

Additional heat can affect groundwater chemistry, for example, through enhanced carbonate precipitation (Griffioen and Appelo, 1993), increased mobility of heavy metals (Bonte et al., 2013), or reduced oxygen concentrations (Figura et al., 2013; Riedel, 2019). Furthermore, shallow aquifers host ecosystems that deliver valuable services, such as biological water purification (Griebler and Avramov, 2015). Increasing GWT might affect these ecosystems as groundwater fauna and microbiological communities are adapted to stable conditions. Additionally, urban aquifers are already stressed by various factors, such as salt and heavy metal pollution or oxygen depletion (Becher et al., 2022). A good indicator for adverse changes in groundwater quality is the abundance of prokaryotic cells (Fillinger et al., 2019; Retter et al., 2021), which may increase with rising temperatures (Lienen et al., 2017). While higher water temperatures stimulate microbial metabolism, bacterial growth in clean and oligotrophic groundwater systems is limited due to restricted energy availability (Brielmann et al., 2009). Hartog et al. (2013) found no correlation between bacterial quantities and temperature at a monitored aquifer thermal energy storage (ATES) site (11–35 °C). Microbial biodiversity, on the other hand, is expected to change already at a moderate GWT increase, not only in eutrophic aquifers (Brielmann et al., 2011; Griebler et al., 2016). Additionally, a negative relationship was observed between water temperature and both biodiversity of groundwater fauna (Brielmann et al., 2009; Spengler and Hahn, 2018) and the lethality of individual crustacean species (Brielmann et al., 2011). The impacts of a locally confined heat plume caused by thermal energy discharge on shallow groundwater ecosystems was previously studied by Brielmann et al. (2009). They measured GWT of up to 17 °C and detected no significant correlation between temperature and bacterial and faunal abundances. However, they could associate an increase in bacterial diversity and, at the same time, a decrease of faunal diversity with the thermal anomaly. This could be due to the fact that microbial communities are generally ubiquitous and less adapted taxa can easily be replaced by others as environmental conditions change, whereas stygofauna are mainly cold stenotherms with a small thermal tolerance (Brielmann et al., 2011). Loss of species is hardly compensated by new ones. In most countries, thermal alterations of the subsurface by geothermal applications are not regulated (Hähnlein et al., 2010) while the unintended impacts of anthropogenic structures are not yet considered (Blum et al., 2021b).

In this study, we present a holistic field investigation, aiming to (1) quantify the thermal impact on the groundwater caused by a public water park ("Aquadrom Hockenheim") and its associated basements and (2) investigate the environmental impact of the induced local heat plume on groundwater quality and ecology. By repeatedly sampling nine wells in the immediate vicinity of the water park and three wells in a nearby water protection zone, we are able to determine possible deviations from the natural state and a deterioration between thermally unaffected wells and those affected by the thermal plume of the water park. Evaluation of the ecological state of the groundwater is done by the D-A-C index based on different microbial measures, namely the prokaryotic cell density, activity, and bioavailable carbon (Fillinger et al., 2019). In addition, occurrence of groundwater fauna was evaluated. Insights gained about temperature-induced ecological changes will help uncover possible consequences of unintended thermal impacts of anthropogenic heat sources to groundwater communities and in consequence to groundwater quality. The findings will also support knowledge-based decision making of policy makers and authorities, for example, for the utilization of the subsurface by geothermal applications or low-temperature thermal energy storage systems, such as subsurface thermal energy storage systems (Bott et al., 2024; Fleuchaus et al., 2018).

2. Materials and methods

2.1. Study site

The site of investigation is located in the north-western part of the German state of Baden-Württemberg (Fig. 1A) with the water park called "Aquadrom" at the southern border of the city of Hockenheim. The land use of the surroundings is a mixture of residential and agricultural areas. Three reference wells (RE1–RE3) are embedded in the Reilingen forest, approximately 3–4 km south of the water park (Fig. 1b). Since the wells are located in a water protection zone II, the condition of the groundwater can be considered close to natural and thermally undisturbed. The depth of the three wells is between 30 and 33 m.

The water park itself consists of an indoor and outdoor area. The basement is located mainly underneath the indoor area, but also extends north to the outdoor pools (Fig. 1c). It is about 3 m deep and has an area of 6580 m^2. One extraction well (P9) and three observation wells (GWM1–GWM3) are placed in the south-eastern corner of the property in an upstream location. The extraction well is not in operation and was originally intended for the installation of a geothermal application. One observation well is drilled through the basement slab, allowing the groundwater beneath the building to be studied (P8). Four observation wells are in different downstream locations on the property (P1, P2, P4, P5; Fig. 1c). All wells have a depth of 7–10 m below ground surface, except for P9 which is 15 m deep. In the north-eastern area of the property there is an oil-fueled cogeneration plant, which supplies the water park with heat and power.

The thermal impact of the Aquadrom was investigated earlier in the context of a feasibility study for a thermal energy storage of waste heat in the upper aquifer (Blum et al., 2021a). The results of this study show a limited suitability for aquifer thermal energy storage due to the relatively high groundwater velocity and shallow groundwater depth, which can reduce the recovery rate.

Hockenheim is located on the Lower Terrace of the Upper Rhine Graben which consists of fluviatile Late Pleistocene deposits and extensive fluviatile flood deposits and channel fillings (HGK, 2001). Drillings at the study site show sandy gravels to medium sands as well as peaty layers and clayey-silty lenses, deposited by the small River Kraichbach, which is located about 250 m to the east.

The direction of groundwater flow at the study site is approximately north-west towards the Rhine River (Fig. 1c). A previous pumping test on this site published in Blum et al. (2021a) resulted in a hydraulic conductivity of $2.1 \cdot 10^{-3}$ m/s and a groundwater velocity of about 0.5

m/d. The groundwater has a seasonally fluctuating depth of 3–5 m and is only about 1 m below the base of the Aquadrom.

The water quality in the water protection zone at the Reilingen forest is generally good. Except for the flocculation of iron and manganese, no further treatment is required to achieve drinking water quality in accordance with German law (Stadtwerke Hockenheim, 2023). However, the presence of reduced iron and manganese in the produced water clearly indicates hypoxic to anoxic conditions.

Hockenheim has a transitional climate between maritime and continental influences. The mean air temperature in the period of 1991–2020 at the weather station Waghäusel-Kirrlach in about 7.5 km distance is 11.4 °C (DWD, 2023).

2.2. Temperature monitoring and sample collection

The data for the temperature time series were recorded in the period from May 2022 to May 2023. HOBO data loggers of the type U20L-01 were placed at a depth of 5 m below surface level in all wells shown in Fig. 1c. They monitored the water level as well as the GWT with an accuracy of ±0.37 K and a resolution of 0.1 K. Additionally, four temperature data loggers of the type U22–001 (accuracy: ± 0.21 K, resolution: 0.02 K) were deployed, one of them in P8 at 2.5 m depth below surface level and the others in P9 at 2.5 (attached above groundwater surface), 10 and 15 m depth. Two data loggers of the type MX2201 (accuracy: ± 0.5 K, resolution: 0.04 K) monitored the basement air temperature and one of the same type monitored the outdoor air temperature in a covered and shaded area at 2 m above ground level. All loggers were deployed with a temporal resolution of 1 h.

Sampling was performed every three months, starting in May 2022, and ending in May 2023, resulting in a total of five field campaigns. Before sampling, temperature depth profiles were taken with an RBR*duet*[3] T.D temperature data logger (accuracy: ± 0.002 K, resolution: 0.00005 K). It was attached to a contact gauge and lowered slowly through the well. By measuring water pressure and temperature in 0.5 s time steps, we were able to create quasi-continuous temperature profiles. Subsequently, a multiparameter probe (KLL-Q-2 with MPS-D8, SEBA), which was calibrated prior to application, was used to measure profiles of key physico-chemical parameters (pH, electrical conductivity [EC], redox potential [Eh], concentration of dissolved oxygen, oxygen saturation) in wells wider than two inches diameter. However, this applied only to the following wells: RE1, RE2, RE3 and P9.

The sampling routine involved the collection of well water and freshly pumped groundwater for both chemical and microbial analysis. Well water samples were retrieved using a bailer, while pumping was conducted with a Grundfos MP1 submersible pump. Prior to sampling from the aquifer, the well water underwent pre-purging twice the well volume and until key physico-chemical parameters stabilized. For chemical and adenosine triphosphate (ATP) analysis, samples were collected in sterilized glass bottles. Samples intended for the bacterial abundance (BA) analysis, also referred to as total cell counts, were filled into sterile 15 ml Falcon tubes supplemented with glutardialdehyde fixative (final conc. of 0.5 %). Finally, samples for dissolved organic carbon (DOC) analysis were filtered through 0.45 μm polyvinylidene fluoride (PVDF) syringe filters directly at the sampling site. Before pumping, fauna samples were taken with a modified Cvetkov net according to the procedure described in Hahn and Fuchs (2009).

Due to clogging, pumping was not possible at the wells P2 and P4 and therefore, only well water samples could be obtained here. The clogging was likely caused by iron and manganese deposits in the filter section of the wells. However, further analysis as described in Chapter 2.4 requires aquifer water samples. We have therefore considered P2 and P4 for the temperature analysis, but not for the ecological evaluation.

2.3. Sample analysis

Laboratory analysis of the chemical parameters of both well and aquifer water involved identifying major anions and cations by means of ion chromatography, alongside quantifying DOC with a TOC analyzer.

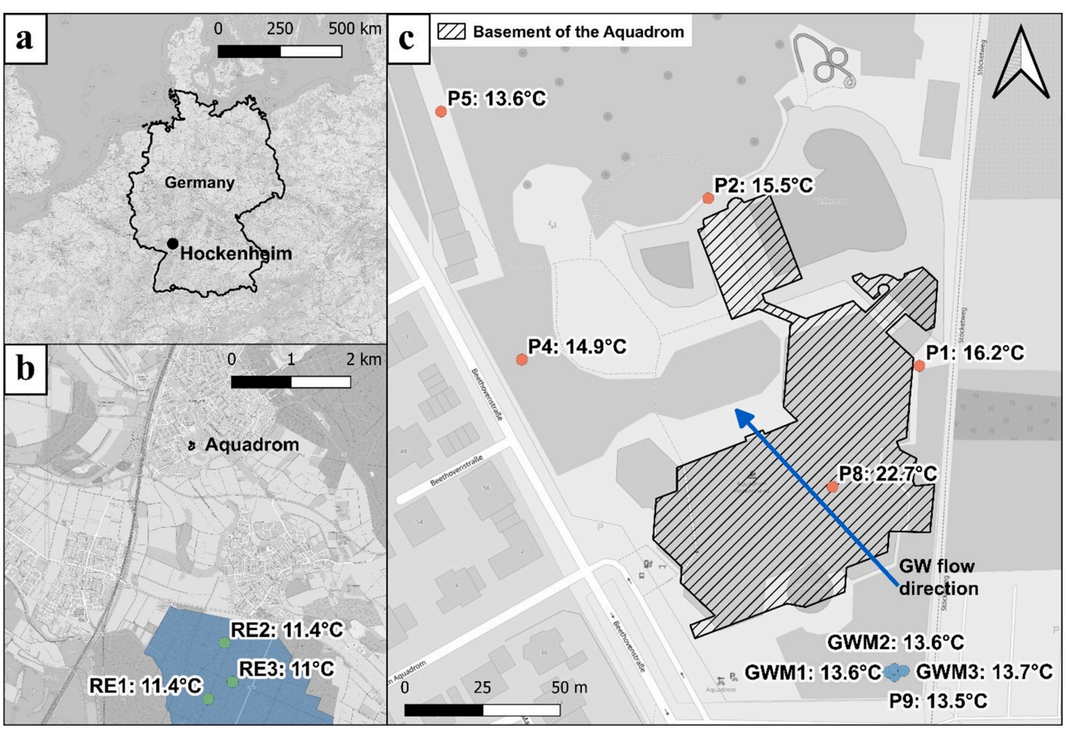

Fig. 1. (a) Location of Hockenheim in Germany. (b) Location of the Aquadrom in Hockenheim and the position of the three reference wells in the Reilingen forest. The water protection zone II is indicated by the blue area (c) The Aquadrom Hockenheim with all observation wells which were used in this study. The temperature labels represent the arithmetic mean over the course of a year in 5 m depth. Colors indicate reference wells (green), thermally unaffected wells (blue), and thermally affected wells (red). Basemap: OpenStreetMap.

Evaluating the microbiological properties of the water comprised determining BA (in cells/ml) and the concentration of cellular ATP (in pM) of prokaryotic cells. Prokaryotic cell counts were obtained utilizing flow cytometry (Cytomics FC500 Flow Cytometer by Beckman Coulter), with a detailed methodology described by Fillinger et al. (2019).

For ATP measurements, the BacTiter-Glo Microbial Viability Assay Kit (Promega) was used, following the method outlined by Hammes and Egli (2010), with adjustments as detailed by Fillinger et al. (2019). Sample preparation and measurements were conducted at room temperature and all measurements were carried out in technical triplicates. Luminescence emitted from the ATP-dependent oxidation of luciferin catalysed by luciferase was detected using the GloMax 20/20 luminometer (Promega). An ATP calibration curve was established using an external ATP standard (100 nM, BioThema) dissolved in ATP-free water (Invitrogen™ UltraPure™, Fisher Scientific). Furthermore, cellular ATP concentrations were derived by subtracting external ATP from total ATP concentrations. Measurements conducted on unfiltered samples reflected the total ATP content (Hammes and Egli, 2010). To isolate extracellular ATP by removing cells, samples aliquotes were filtered (Millex® 33 mm PVDF, 0.1 μm), enabling the assessment of luminescence solely from the extracellular ATP fraction.

2.4. Classification of the ecological state

On behalf of the Federal Environment Agency (Umweltbundesamt) of Germany, Griebler et al. (2014) created an assessment scheme that enables the evaluation of the ecological state of a groundwater body, named GESI. In an ideal case, this assessment is based on the proportion of the crustacea and oligochaeta populations. If not enough fauna is found, when the concentration of dissolved oxygen is regularly below 1 mg/l, or the wells are ochred, the scheme proposes an evaluation based on chemical and microbiological parameters only as living conditions are considered hostile for animals. Overall, at least five criteria have to be assessed from a number of parameters.

However, since there are no suitable reference values provided in the data set, a level 2 assessment has to be conducted (Griebler et al., 2014). This is achieved by firstly defining the natural background values of the local aquifer. The data obtained at the reference wells RE1–3 in the Reilingen forest is suitable to define the natural background since the wells are located in a water protection zone II, where no direct anthropogenic influences are to be expected. The hydrogeological conditions in this area are similar to those at the water park. Furthermore, the groundwater extracted in this forest undergoes no treatment except for the flocculation of iron and manganese, since in its natural state it already meets the chemical criteria defined in the German drinking water guideline (Stadtwerke Hockenheim, 2023). Ideally, these background values are used to evaluate at least two criteria from the following categories: (1) physico-chemical, (2) microbiological and, (3) faunistic criteria. However, faunistic criteria can only be applied to oxic aquifers, where there is a regular presence of stygophile or stygobite fauna. The range of background values is calculated by the minimum and maximum of the mean values of each well. The ecological state of each well at the study site is then calculated by dividing the number of parameters within the reference values (*Ppos*) with the total number of parameters (*Ptot*), thus obtaining a quality grade between 0 and 1:

$$GESI = \frac{Ppos}{Ptot} \tag{1}$$

As an additional assessment scheme, the D-A-C index offers a method to identify environmental disruptions through the assessment of microbiological parameters ATP, BA, and dissolved organic carbon (Fillinger et al., 2019). The D-A-C index is calculated with the Mahalanobis distance, facilitating a multivariate outlier analysis by combining these parameters. This multivariate approach exhibits greater resilience and sensitivity in outlier detection compared to univariate analyses (Retter et al., 2021). In this multivariate space, normally distributed data forms an elliptical cloud defined by the mean values of the variables, while the covariance matrix shapes the ellipse's form and orientation (Retter et al., 2021). The Mahalanobis distance from each sample to the centroid of the ellipse serves as an indicator of disturbances if this distance exceeds distances attributable to random variation. In an unguided scenario, no specific reference groups are designated, and the analysis comprises the entire dataset. Conversely, a guided approach includes the establishment of a reference group, with outliers identified as significantly divergent from these references. The reference group comprises samples collected from wells RE1–RE3, located within the water protection zone of the Reilingen forest. To identify outliers, we defined the critical value at a 99 % confidence level of a chi-squared distribution with three degrees of freedom, following the methodology of Fillinger et al. (2019). Samples with Mahalanobis distances exceeding this threshold are therefore considered as disturbed. Computations were executed using the programming language R, with variables subjected to log10 transformation prior to analysis.

3. Results and discussion

3.1. Thermal impact on groundwater

As a water park with several swimming pools and an extensive basement, the Aquadrom yields an impact on the thermal regime of the subsurface. Extent, intensity, and seasonality of this impact are to be analyzed here. Due to the monitoring not only of groundwater, but also of air temperatures over the course of a year, we are able to identify seasonal variations. The spatial distribution of the GWT at a depth of 5 m is depicted in Fig. 1c. Fig. 2a shows the time series of the GWT in all wells. Additionally, the temperature evolution of surface and basement air is plotted with dashed lines. The outdoor air temperature shows the typical seasonal pattern with an arithmetic mean of 14.1 °C (SD: 7.9 K). The placement in a roofed area close to a building possibly influences the temperature. In the basement, we observed 29.4 °C on average, however the SD is much smaller with 1.7 K, since the basement air is heated by the machines and pools throughout the year. The heat is distributed unevenly. Especially the secluded and only seasonally used northern part is cooler in the winter months (Blum et al., 2021a).

The wells GWM1–GWM3 are thermally unaffected by the site and show the expected patterns at 5 m depth of elevated GWT in fall and lower GWT in spring. Arithmetic means of the GWT are 13.6–13.7 °C, with SD of 0.9–1.0 K. The GWT of well P9, on the other hand, was recorded at a depth of 15 m and shows significantly less seasonal variation (mean: 12.6 ± 0.1 °C). These GWT in the upstream are already higher than the expected range of 11–12 °C for this region, as it can be seen exemplarily at the reference wells which are located in a forest (11.0–11.4 °C). This difference might be caused by the semi-urban setting and the resulting subsurface urban heat island of Hockenheim. Due to the high groundwater velocity (0.5 m/d), it is unlikely that the Aquadrom directly influences the GWT at the upstream wells.

Directly influenced by the heated basement, the well P8 shows high GWT during the whole year (mean: 22.7 ± 0.3 °C), thus overlaying the natural seasonal variations. In Fig. 2b, it can be seen that the heat at P8 comes from above and decreases with depth. The other thermally affected wells show similar seasonal patterns as the unaffected wells. The GWT at these wells is increased by up to 2.6 K. Interestingly, the monitoring well P1 has the highest GWT of the downstream wells with 16.2 °C on average although it lies laterally to the building in groundwater flow direction. This strong impact is due to the small distance of only 1 m to the basement wall. The direct influence of the water park on well P1 is also evident by the temperature-depth-profile in Fig. 2b, which shows the same shape as the one of the well P8. The monitoring well P5, which is located about 100 m downstream from the basement, has a mean GWT of 13.6 °C. Because this value is equal to the GWT at the upstream wells, we assume the thermal plume to be dispersed at this point. Thus, the water park yields a local thermal anomaly in subsurface

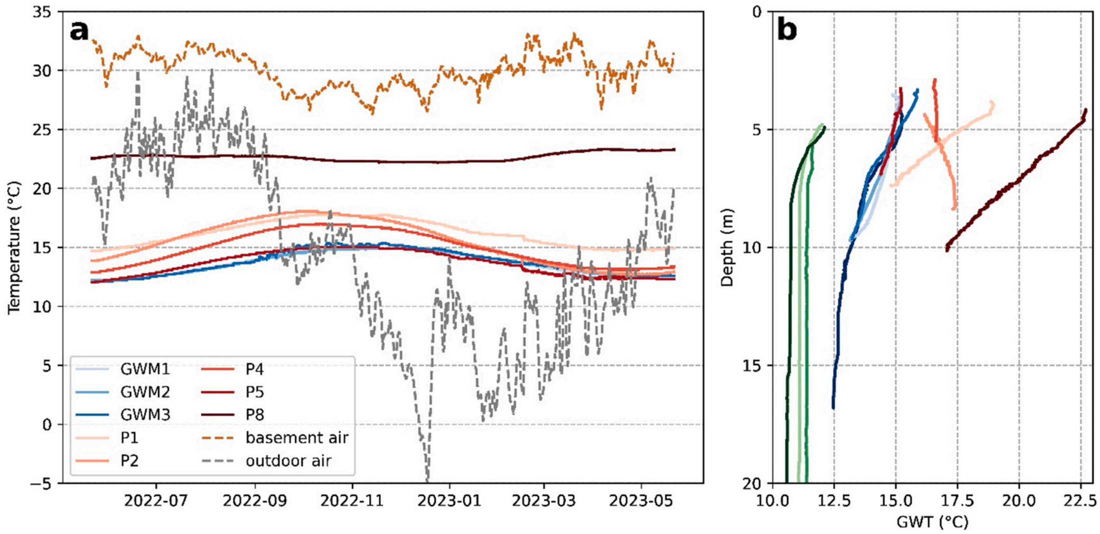

Fig. 2. (a) Time series of monitored groundwater and air temperatures. Groundwater temperatures were recorded in 5 m depth below surface level except at P9, where the logger was installed at a depth of 15 m. All time series were smoothed with a 24 h rolling mean. (b) Temperature depth profiles of all wells, taken in November 2022. The color groups indicate the position relative to the heat source: reference wells (green), thermally unaffected wells (blue), and thermally affected wells (red). Temperature depth profiles of all campaigns are depicted in Fig. A1.

temperatures (see Fig. 1c).

Applying Fourier's law of heat conduction (Fourier, 1888), a simple assessment of the rate of subsurface warming by the water park can be carried out. The equation is applied using the measured data of the thermally unaffected wells, P8 and the basement air temperature. The resulting heat flux into the aquifer is about 44 W/m^2 at the upstream facing building part and 19 W/m^2 around the well P8, where the groundwater is already heated. This suggests that spatially and temporally resolved analysis, for example with numerical modeling, would provide more detailed insights into the distribution of the water park's heat losses into groundwater. However, this first evaluation shows that the emitted energy of the basement slab amounts to 3–7 MWh.

Similar to district heating networks, leakage from pools or pipes can be prominent additional sources of heat input. However, this is difficult to detect and distinguish from the general conductive heat flux as leakages act as point sources. Only few other publications mentioned subsurface thermal anomalies due to swimming pools. Menberg et al. (2013) recorded around 20 °C warm groundwater downstream to a heated public swimming pool in Frankfurt, Germany, while Tissen et al. (2019) detected 16 °C in a well 4 m away from a municipal swimming pool in Germany. In comparison to other regular heat sources, such as residential basements, underground car parks, and tunnels, swimming pools yield higher impacts on the subsurface thermal regime as they are typically filled with heated water instead of air. Geothermal applications, on the other hand, can cause even higher thermal impacts (García-Gil et al., 2020; Mueller et al., 2018). These, however, are regulated by law in many countries (Hähnlein et al., 2010). The heat flux emitted from the water park's basement is on the upper end compared to the fluxes of other heat sources (Noethen et al., 2022). This is probably due to the small distance to the groundwater of about 1 m from the basement's slab and the comparatively high temperature difference between the basement and the groundwater.

3.2. Environmental impact of groundwater heating

3.2.1. Groundwater chemistry

The chemical groundwater data both at the Reilingen forest and the Aquadrom site indicates that the aquifer has reducing conditions since mean concentrations of oxygen (0.71 ± 0.75 mg/l) and nitrate (87 % below detection limit of 0.125 mg/l) are low, while mean concentrations of DOC (10.6 ± 9.4 mg/l) and iron (5.7 ± 3.8 mg/l; Blum et al. (2021a)) are high. Some chemical parameters measured in groundwater at the study site differed from the natural background, for example pH is lower, with a mean of 6.8 for thermally unaffected wells and 7.3 for the reference wells, and EC is higher (mean 1.5 and 0.7 mS/cm, respectively) as shown in Fig. 3. Possible reasons for these deviations are the greater depths of the wells in the Reilingen forest as well as increased releases of nutrients and pollutants into the subsurface due to anthropogenic activities in the urban area of Hockenheim. Especially the high concentrations of sulphate at the test site, with 369 mg/l on average (natural background: 133 mg/l), point at systematic differences. Sulphate concentrations are higher than the threshold defined by the German drinking water regulations of 250 mg/l. High sulphate concentrations may originate from construction waste (e.g. bricks) buried in the shallow subsurface of urban areas. In addition, high sulphate is typical for organic rich aquifers, aquifers containing old saline waters or experience the upwelling of deeper thermal waters. Finally, fertilization of agricultural land upstream of the test site could have contributed to elevated levels (Kaown et al., 2009; Spoelstra et al., 2021). Chloride concentrations are most likely not affected by the chlorination of pool water, since the difference between up- and downstream wells is insignificant (mean 101 and 97 mg/l, respectively). Possible sources for increased chloride contents in comparison to the reference wells are de-icing of streets with salt and fertilization of agricultural land in upstream areas (Kelly et al., 2012; Perera et al., 2013).

However, none of the chemical parameters are correlated with GWT or altered in the thermally affected wells (Fig. 3). In addition, Spearman correlation coefficients (ρ) range between 0.08 and 0.48 (P = 0.01–0.69). Only EC shows a moderate correlation with GWT (ρ = 0.48, P = 0.01). Although the sulphate concentration is higher in the thermally affected wells than in the unaffected wells (412 and 334 mg/l, respectively), the correlation with GWT is weak and not significant (ρ = 0.21, P = 0.23). Mean values of relevant parameters at the reference wells, the thermally unaffected, and affected wells are given in Table A1, alongside thresholds defined by the German drinking water regulations, if applicable. Furthermore, no seasonal temporal trends were observed within the study period.

3.2.2. Groundwater microbiology and fauna

The mean BA of groundwater from wells at the study site ranged from $9.5 \cdot 10^4$ cells/ml at P5 to $1.5 \cdot 10^5$ cells/ml at GWM3. Mean values in groundwater from the reference sites were lower ($1.5 \cdot 10^4$ to $9.0 \cdot 10^4$

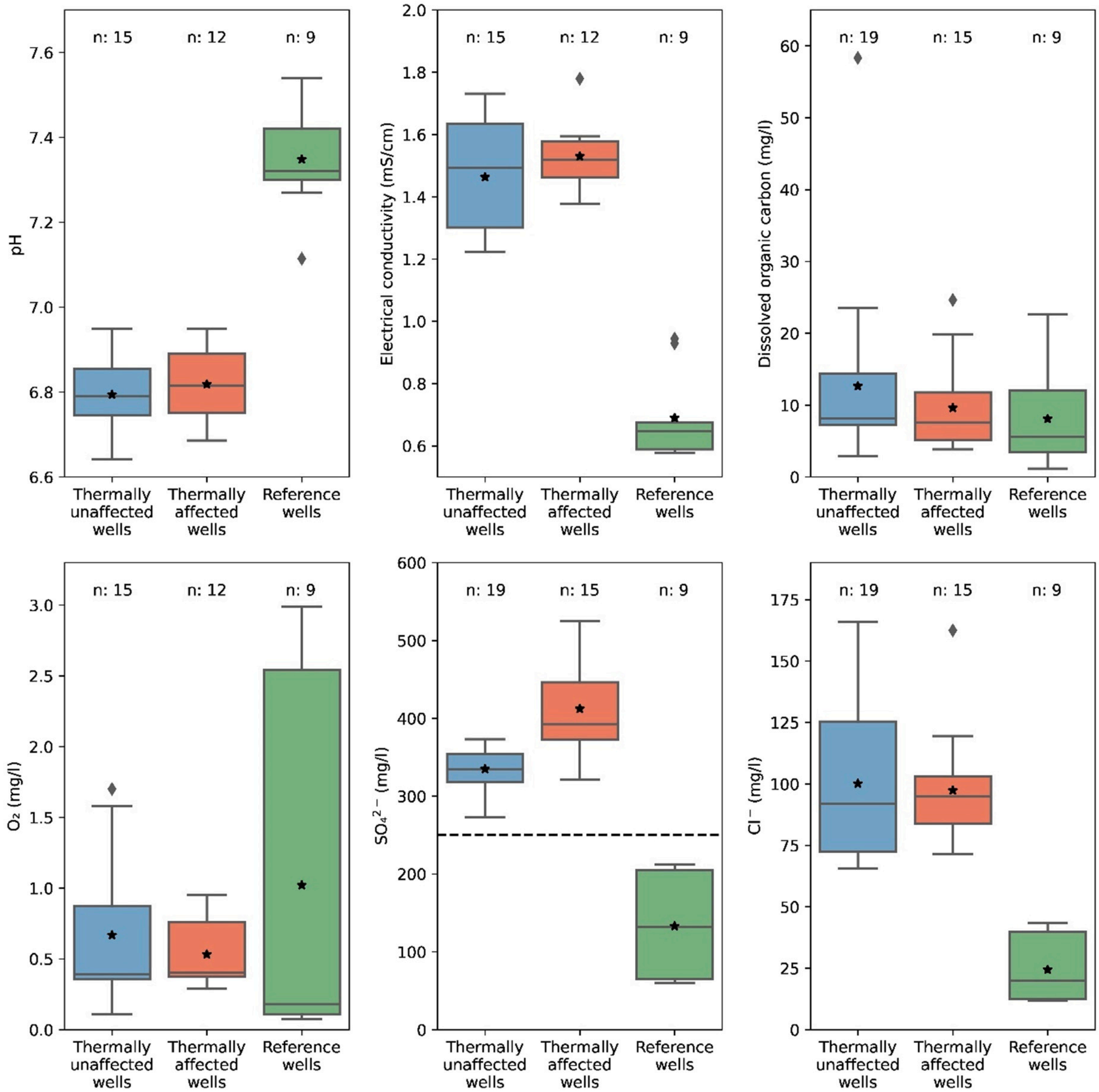

Fig. 3. Selected chemical parameters of groundwater samples taken in the wells at the Aquadrom Hockenheim (thermally unaffected/affected wells) and in the Reilingen forest (reference wells). Mean values are indicated by stars. Sulphate concentration threshold defined by the German drinking water regulations is shown with a dashed line. Other threshold values are outside the data ranges and are given in Table A1.

cells/ml) which can be explained by the natural, non-urban setting and consequently less import of organic compounds and nutrients, as well as by the greater depth of the wells. However, there is no direct correlation between BA and GWT ($\rho = 0.09$, $P = 0.60$). Thermally unaffected wells in the upstream have a slightly lower mean BA value compared to those which are affected in the downstream of the structure ($1.4 \cdot 10^5$ and $1.1 \cdot 10^5$ cells/ml) as shown in Fig. 4. This underlines the sensitivity of BA as an indicator to discriminate between anthropogenically impacted from close to natural groundwater quality, although temperature alone does not seem to be an exclusive driver.

Similar patterns can be observed with the cellular ATP values which differed significantly between the single wells. The highest mean cellular ATP concentration was found in GWM1 (195 pM), the lowest value in P9 (32 pM). Again, the reference wells showed lower values on average (25–88 pM). Statistically, there is no direct significant relation between GWT and cellular ATP ($\rho = 0.11$, $P = 0.53$). Thermally unaffected wells in the upstream had on average similar cellular ATP values as those in the downstream of the Aquadrom (133 and 108 pM, respectively).

Generally, both the BA and cellular ATP in groundwater sampled at the study site are in the upper concentration range when compared to other non-contaminated and close-to-natural shallow aquifers in Germany (Fillinger et al., 2019). This can be explained by the reducing conditions and high DOC concentrations. Anoxic groundwaters generally exhibit higher BA and ATP values due to elevated DOC and phosphate levels (Griebler et al., 2014). Increased inputs of nutrients from agricultural and residential areas are probably additional drivers for microbial abundance and activity. What may be expected, but has not been addressed in this study, is a shift in microbial community composition. Previous studies have shown a community change and an increase in bacterial diversity with groundwater warming (Brielmann et al., 2009; Griebler et al., 2016).

The abundance of fauna in groundwater was generally low both at the study site with 2.0 and in the Reilingen forest with 1.8 individuals per sample on average (Fig. 4). Most samples were either completely devoid of fauna (68 %) or revealed only a small number of 1–3 individuals (25 %), while few outlier samples contained up to 48 individuals. The sample that contained 48 animals was taken in May 2023

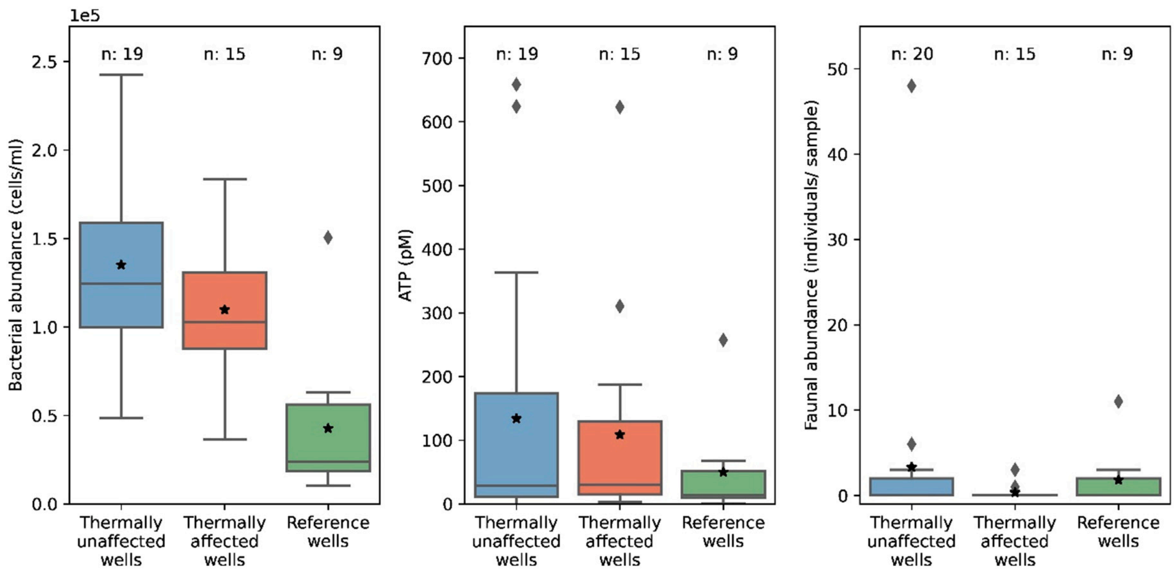

Fig. 4. Biotic parameters of groundwater samples taken in the wells at the Aquadrom Hockenheim (thermally unaffected/affected wells) and in the Reilingen forest (reference wells). Mean values are indicated by stars. ATP = adenosine triphosphate.

in GWM3 and did not show any anomalies in groundwater chemistry (e.g., $O_2 = 0.35$ mg/l). The samples from well P8, which is located beneath the Aquadrom's basement and showed the highest GWT, contained only one animal (Nematoda) throughout all five campaigns.

The low abundance or even absence of fauna in groundwater was expected due to the low amount of oxygen in the shallow aquifer. In consequence, the use of groundwater fauna taxa as indicators for the ecological status of the aquifer at the test site was not applicable. At the prevailing conditions, the absence of fauna in groundwater does not necessarily indicate a poor ecological status (Griebler et al., 2014). An earlier study by Fuchs (2008) on groundwater fauna in Baden-Württemberg has already found that only one of three wells in the city of Hockenheim was colonized.

Bathynellacea (total counts: 6) and Cyclopoida (1), both taxa belonging to the crustaceans, were only found in the reference wells. No crustaceans could be found in wells at the water park. A single individual of the stygophile Cyclopoida genus *Diacyclops languidoides* was found in RE2 in May 2023. Acari (26) were found in most of the wells. Oligochaetes (42) were identified in three samples only, despite the high number of total counts. The majority of them occurred in a single sample in GWM3 in May 2023 and were identified as the stygophile species *Marionina argentea* (32). Other Oligochaetes were identified as *Dorydrillus/Trichodrilus* (7) in the well RE2 in February 2023. Only one Gastropoda was found in P1 and Nematoda (17) were scarce as well, except for GWM3, where 14 individual were found in total in five samples.

3.2.3. Ecological assessment schemes

For the groundwater ecosystem status index (GESI), introduced by Griebler et al. (2014), different abiotic and biotic parameters are assessed integratively against the natural background at reference sites. As already mentioned above, groundwater faunal parameters could not be considered for the index, since living conditions (low or no oxygen) are hostile at both the study site and the reference site. Hence, the Groundwater Fauna Index (Hahn, 2006) could not be applied as well. On a qualitative level, the occurrence of crustaceans at the reference sites and the absence of crustaceans in the Aquadrom area, hints at additional pressures at the test side besides the low oxygen conditions.

With the chosen parameters, the results for the GESI show a heterogenous distribution, ranging from 0.2 to 0.6, as it is shown in Table 1. According to Griebler et al. (2014), the ecological state of all wells at the water park is either considered *severely affected* (GESI ≥0.2–0.4) or *affected* (GESI ≥0.4–0.6), which means a significant deviation from the natural reference values. This is explained by the higher GWT but also due to a greater microbiological abundance. These results were expected since the upper aquifer is influenced by agricultural and residential land use. Furthermore, a decrease of the GESI for the thermally affected wells could not be observed.

In order to detect disturbances in the ecological state of the upper aquifer, we applied another groundwater health assay, the D-A-C index. The results of this multivariate analysis are expressed as the distances of the individual samples in a three-dimensional space (Mahalanobis distance) and plotted against the GWT in Fig. 5.

For this approach, 83 % of the samples taken at the water park exceed the critical value of the chi-squared distribution at a 99 % confidence level with three degrees of freedom. Hence, all samples except for five can be considered as disturbed in comparison to the natural background values. This can mainly be attributed to the previously described increased microbiological features, i.e., the cellular ATP and BA values. Four of the five samples that did not exceed the critical value were taken in May 2022, where the lowest average DOC concentrations were measured (May 2022 mean: 4.6 mg/l, tot. Mean: 11.5 mg/l). There is no significant difference in the D-A-C index between samples from thermally unaffected (6.7) and affected wells (5.7) within the urban area. Considering all samples taken at the Aquadrom, there is no correlation between GWT and the ecological state ($\rho = 0.18$, $P = 0.32$). If we exclusively correlate the samples taken at the thermally unaffected wells, there is a moderate to strong positive correlation with GWT ($\rho = 0.60$, $P = 0.01$). From this, it can be concluded that temperature is one but not the only driving factor of microbial abundance in groundwater. All thermally unaffected wells are in close proximity to each other, while the thermally affected wells are distributed over the whole plot, leading to a broader variation of chemical parameters, possibly overruling the sole impact of groundwater warming. Although temperature is without doubt a parameter influencing groundwater ecology, we could not detect any direct significant deterioration of the ecological state by temperature in this case study.

4. Conclusions

In this study, a water park in south-western Germany was identified as the source of a local groundwater heat plume of up to 23 °C (9 K

Table 1
Parameters included in the Groundwater Ecosystem Status Index (GESI) and results, determined according to Eq. 2. Mean values for each well are displayed. GWT = Groundwater temperature; DOC = dissolved organic carbon; ATP = adenosine triphosphate; BA = bacterial abundance.

Criteria (unit)	Natural background	Thermally unaffected wells				Thermally affected wells		
		GWM1	GWM2	GWM3	P9	P1	P5	P8
GWT (°C)	11.0–11.4	13.6	13.6	13.7	13.5	16.2	13.6	22.7
O_2 (mg/l)	0.1–2.8	0.7	0.6	0.7	0.7	0.5	0.5	0.6
DOC (mg/l)	4.7–12.4	11.9	21.4	9.3	7.2	11.4	9.2	8.2
ATP (pM)	25–88	195	95	194	32	106	148	71
BA (cells/ml)	$1.5{\cdot}10^4$–$9.0{\cdot}10^4$	$1.4{\cdot}10^5$	$1.3{\cdot}10^5$	$1.5{\cdot}10^5$	$1.2{\cdot}10^5$	$1.1{\cdot}10^5$	$9.5{\cdot}10^4$	$1.2{\cdot}10^5$
GESI (–)	–	0.4	0.2	0.4	0.6	0.4	0.4	0.6

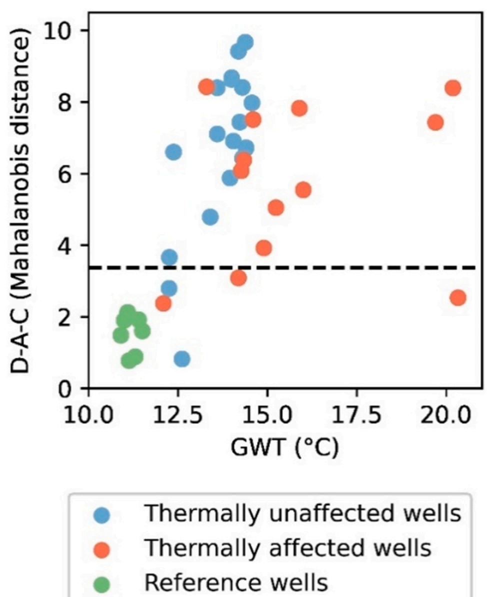

Fig. 5. D-A-C index of all samples, computed following a guided approach and correlated to the groundwater temperature (GWT). The dashed line shows the critical value of the chi-squared distribution at a 99 % confidence level with three degrees of freedom.

increase), induced by the basement with an average air temperature of 29 °C. The main objective, whether this hot spot affects groundwater quality and ecology, was investigated by comparing data from thermally unaffected and affected wells and by applying the ecological assessment schemes GESI and D-A-C index.

Chemical parameters have shown no significant changes due to heat. Increased sulphate and chloride concentrations in comparison to reference wells in a water protection zone imply that groundwater chemistry at the water park is already influenced by anthropogenic impacts due to residential and agricultural land use. Furthermore, analysis of ecological parameters as well as the assessment schemes underlined clear differences between the chosen reference site and the water park. However, GWT might be one driver of the differences but not the only one. This is most likely due to the fact that the influence of GWT is masked by other factors such as the influence of contamination from urban and agricultural land use. Despite the low oxygen concentration with a mean of only 0.71 mg/l, it could be shown that the aquifer still is inhabited by a small number of stygophile fauna which move regularly and actively into the groundwater ecosystem. Impacts from GWT on fauna, however, could not be analyzed due to the overall low oxygen concentrations and the consequently small number of animals. Assessment based on microbial indicators, on the other hand, allowed to clearly discriminate between groundwater from reference sites and samples from the Aquadrom. Within the facility area, the microbial criteria selected were not sensitive enough to reliably separate GWT affected and unaffected from each other. We may speculate that microbial community composition analysis would have delivered a more detailed picture. Also, it is important to note that the insights gained in this case study are transferable to other sites and aquifer types only with caution. Microbial and faunal communities in oxic aquifers, for example, might react differently to heat stress. Hence, we propose to verify our results at other sites and improve our understanding about the impact of heat stress to groundwater ecosystems for other aquifer systems with different microbial and faunal compositions as well.

CRediT authorship contribution statement

Maximilian Noethen: Writing – review & editing, Writing – original draft, Visualization, Resources, Methodology, Conceptualization. **Julia Becher:** Writing – review & editing, Methodology, Conceptualization. **Kathrin Menberg:** Writing – review & editing, Resources. **Philipp Blum:** Writing – review & editing, Resources. **Simon Schüppler:** Writing – review & editing, Resources. **Erhard Metzler:** Writing – review & editing, Resources. **Grit Rasch:** Data curation, Resources. **Christian Griebler:** Methodology, Writing – review & editing. **Peter Bayer:** Writing – review & editing, Supervision, Funding acquisition.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A. Appendix

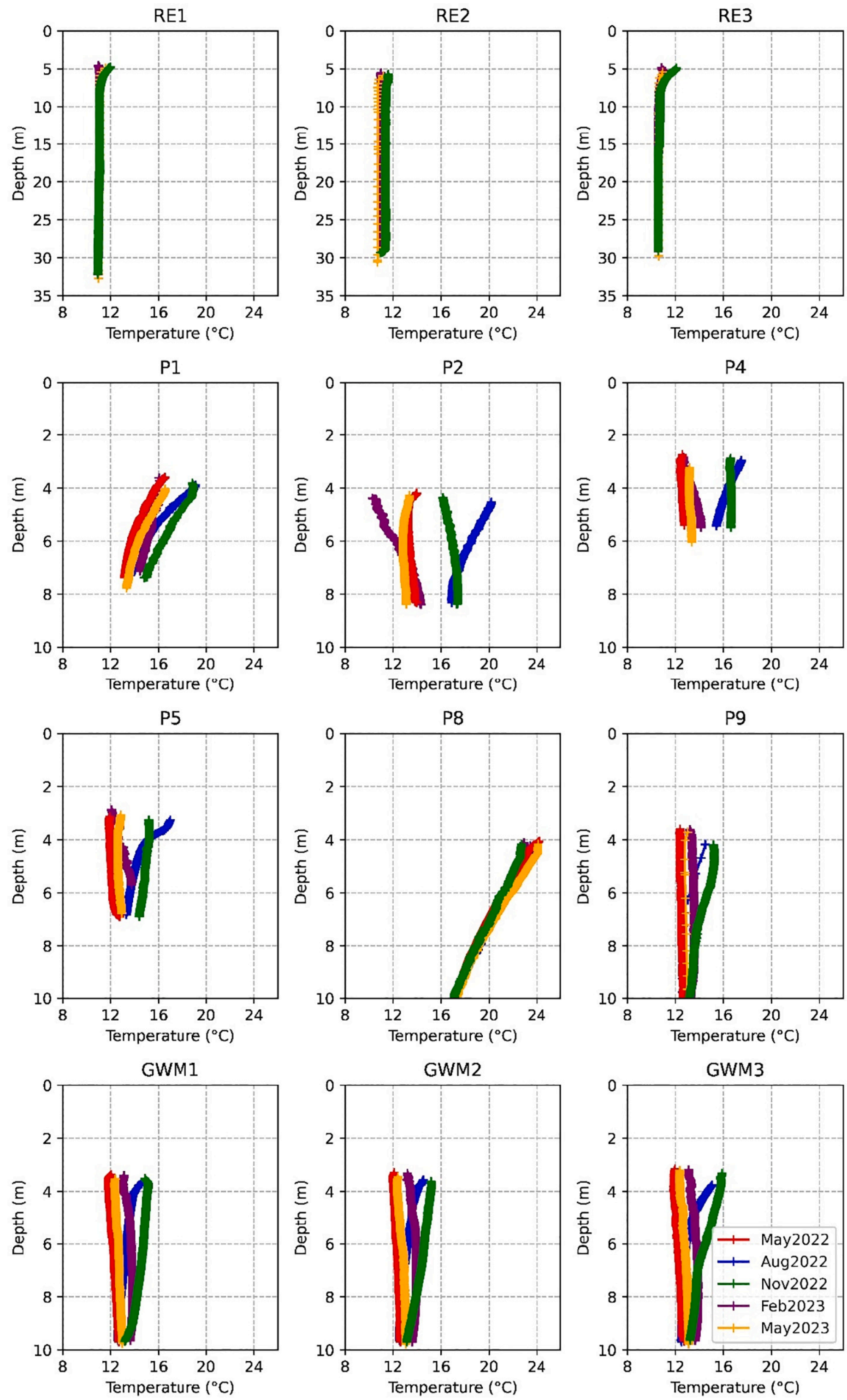

Fig. 1A. Temperature depth profiles of all wells during different seasons.

Table A1
Mean values of measured physical, chemical, microbiological, and faunistic parameters, calculated consecutively for each well and for the classes. Temperature depth profiles were used to calculate the groundwater temperature (GWT) at 5 m depth below surface level. Thermally unaffected wells include GWM1–3 and P9. Thermally affected wells include P1, P5, and P8 as well as P2 and P4 (only GWT). Reference wells are RE1–3. The number of samples (n) is indexed in the footnotes. EC = electrical conductivity; DOC = dissolved organic carbon; ATP = adenosine triphosphate; BA = bacterial abundance, FA = faunal abundance; SD = standard deviation.

Parameter (unit)	Thermally unaffected wells *(SD)*	Thermally affected wells *(SD)*	Reference wells *(SD)*	Limit according to German drinking water regulations
GWT	13.2[a]	16.2[b]	11.3[c]	–
(°C)	*(0.07)*	*(3.69)*	*(0.05)*	
EC	1.5[d]	1.5[e]	0.7[c]	2.79
(mS/cm)	*(0.19)*	*(0.06)*	*(0.12)*	
pH	6.8[d]	6.8[e]	7.3[c]	6.5–9.5
(–)	*(0.06)*	*(0.03)*	*(0.10)*	
O_2	0.7[d]	0.5[e]	1.0[c]	–
(mg/l)	*(0.06)*	*(0.04)*	*(1.52)*	
NO_3^-	85 % < detection limit	87 % < detection limit	89 % < detection limit (0.125)[c]	50
(mg/l)	(0.125)[f]	(0.125)[g]		
NH_4^+	68 % < detection limit	53 % < detection limit	100 % < detection limit	50
(mg/l)	(0.025)[f]	(0.025)[g]	(0.025)[c]	
SO_4^{2-}	355.0[f]	412.1[g]	132.9[c]	250
(mg/l)	*(14.63)*	*(53.83)*	*(73.16)*	
Cl^-	98.7[f]	97.3[g]	24.4[c]	250
(mg/l)	*(33.82)*	*(20.51)*	*(15.29)*	
K^+	3.0[f]	6.2[g]	1.4[c]	–
(mg/l)	*(0.28)*	*(4.98)*	*(0.87)*	
DOC	12.5[f]	9.6[g]	8.1[c]	–
(mg/l)	*(6.2)*	*(1.6)*	*(3.9)*	
ATP	134[f]	108[g]	50[c]	–
(pM)	*(77)*	*(38)*	*(34)*	
BA	$1.3 \cdot 10^5$ [f]	$1.1 \cdot 10^5$ [g]	$4.3 \cdot 10^4$ [c]	–
(cells/ml)	*($1.2 \cdot 10^4$)*	*($1.3 \cdot 10^4$)*	*($4.1 \cdot 10^4$)*	
FA	3.3[a]	0.3[g]	1.8[c]	–
(individuals/ sample)	*(5.7)*	*(0.4)*	*(2.5)*	

[a] $n = 20$
[b] $n = 25$
[c] $n = 9$
[d] $n = 15$
[e] $n = 12$
[f] $n = 19$
[g] $n = 15$

Data availability

Data will be made available on request.

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