Artificial destratification of Dipotamos reservoir in Northern Greece by low energy air injection
Manassis Mitrakas, Petros Samaras, Stelios Stylianou, Chris Kakalis and Anastasios Zouboulis

ABSTRACT
The aims of this study were to investigate the Dipotamos water reservoir destratification process that used the low-energy method of air injection, and evaluate its potential to suppress the presence of nuisance compounds to within permitted drinking water limits. The selection of optimum design parameters for the application of air injection through perforated pipes was based on the Davis model. Data collected throughout a 4-year operation period showed that artificial mixing by air injection may be adequate for the complete destratification of the reservoir. The air injection method had a low energy demand, estimated to be around 55 Wh/10^3 m^3/d. Upon air injection start-up, water column homogenization occurred relatively quickly, as deduced by the values of temperature and dissolved oxygen concentration recorded at various sampling points within the reservoir. After the 1st week of operation, the oxygen concentration at 20 m depth increased from nearly zero to 2 mg/L, and doubled 1 week later, reaching equilibrium after 30 d of continuous operation. The air injection method resulted in the improvement of water quality.

Key words | air injection, algae control, artificial destratification, reservoir

INTRODUCTION
Depletion of underground aquifers has led to the increased use of surface water resources and the construction of dams to meet water supply demands. The primary drawback of surface water aquifers is eutrophication which is caused by a combination of factors, i.e. sunlight, nutrient input, silt and organic matter that all contribute to significant algal growth. Algae are vital to marine and fresh-water ecosystems and most algal species are not harmful (Moustaka-Gouni et al. 2000). However, harmful algal...
blooms (HAB) may occur when certain types of microscopic algae grow quickly in water and these may produce either aesthetic problems or toxic metabolites (Bell & Codd 1994). Taste and odor problems in surface water aquifers, i.e. lakes, rivers, canals and reservoirs, are usually associated with the existence of specific phytoplanktonic microorganisms that release a variety of organic substances, with 2-methylisoborneol (MIB) and geosmin (trans-1,10-dimethyl-trans-9-decalol) being the most important. Odor threshold concentrations (OTCs) for geosmin and MIB have been reported to range from 4 to 10 and 9 to 42 ng/L, respectively.

Cyanobacteria are a diverse group of photosynthetic bacteria that have been identified as the source of very potent toxins, such as neurotoxins, hepatotoxins and dermo-toxins, in aquatic ecosystems (Karner et al. 2001; Westrick 2003). Phytoplanktonic organisms are mainly associated with water column stratification, due to the impact of changing water density at varying temperatures. Stratification often occurs in large water bodies during spring and summer, resulting in the formation of three distinct layers each having significantly different water qualities (Bell & Codd 1994; Albanakis et al. 2001).

Several methods have been employed to control toxic cyanobacteria and minimize the risk of severe contamination associated with dissolved toxins. In general, these methods include the implementation of specific water treatment techniques for the removal of cyanobacterial cells and the application of source water management strategies.

The primary aim of a drinking water treatment plant should be the removal of cyanobacteria with minimum cell disruption, since cell lyses will result in the release of toxins into the water. Post-treatment techniques are applied to remove intact cyanobacteria cells through coagulation and filtration, and remove the dissolved toxins by adsorption and/or post-filtration oxidation. With respect to removal or oxidation of the dissolved toxins, ozone oxidation and/or granular activated carbon (GAC) adsorption have relatively high removal efficiency for several cyanotoxins compared with the WHO guideline value of 1.0 μg/L microcystin LR (Drikas et al. 2001; Castaing et al. 2011; Westrick et al. 2010).

Significant efforts are now paid to the development of low-cost and efficient methods to control algal growth, under the framework of source water management strategies. The addition of algaeicides such as copper sulfate (Hanson & Heinz 1984), presents several drawbacks related to significant environmental effects such as direct or indirect increase of the ecotoxicity and disturbance of the local ecosystem. The depression of algal growth rate by practicing ultrasound (Krivograd Klemenčič & Griessler-Bulc 2010), or ultraviolet sterilization has been demonstrated only in relatively small-scale water ponds. Therefore, artificial destratification is the main efficient management tool available to control problems associated with seasonal thermal stratification. Artificial stratification is implemented by applying energy to a water-body, through oxygenation or aeration, thereby establishing circulating currents, and involves complete mixing of the water column to ensure that atmospheric oxygen is available to the entire water column (Schladow 1995; Singleton & Little 2006; Gafsi et al. 2009).

In countries with high levels of sunlight throughout the year, such as Greece, the intense sunshine during spring, summer and autumn, favors thermal stratification of surface water bodies, resulting in significant aesthetic and potentially toxic problems. Stratification problems were identified in the reservoir of Dipotamos when the dam was first operated (2005). These problems resulted in water quality deterioration and revealed the urgent need to implement appropriate management measures. The aim of this study was to determine the best practice of stratification control by implementing a low-cost method, and the objectives included the study of the most suitable aeration process for artificial destratification, identification of the optimum design parameters, and evaluation of process efficiency over a long-term period.

**BACKGROUND STUDY**

**Water body**

The construction of Dipotamos water reservoir was funded by the European Union in order to supply around $20 \times 10^3$ m$^3$/d drinking water to the nearby Municipality of Alexandroupoli (N. Greece). The surface area of the artificial lake at normal operating level is approximately 1 km$^2$, the watershed area is close to 100 km$^2$, volume is estimated at...
13.5 × 10⁶ m², maximum water depth is 30 m, and the point of water abstraction is about 20 m below the overflow level. The typical physicochemical characteristics of the reservoir water are given in Table 1. Concentrations of the trace elements As, Co, Cr, Ni, Pb, Sb and Se were always below 2 μg/L, and those of Cd and Hg below 0.1 and 0.02 μg/L, respectively.

### Stratification effects

Dipotamos reservoir was flooded in early 2005 and the development of stratified layers was observed in early summer of the same year and intensified in autumn (Figure 1). As shown in Figure 1, the water temperature in the epilimnion layer was close to 15 °C and decreased to 8 °C in water depths greater than 12 m (hypolimnion). The exact boundaries between the epilimnion-metalimnion–hypolimnion layers are not distinctly defined; however, a rough identification of the three layers may be based on temperature and dissolved oxygen (DO) profiles. DO concentrations of around 11 mg/L were measured at the water surface. These concentrations decreased in the lower layers of the reservoir and almost reached zero at the lowest points of the hypolimnion. Simultaneously, the formation of MIB and geosmin was observed in the epilimnion layer and Fe²⁺, Mn²⁺ and H₂S were recorded in the hypolimnion. This resulted in serious deterioration of the drinking water quality, since the existing conventional treatment plant, which operates using coagulation–flocculation, sedimentation, sand filtration and disinfection (NaOCl), failed to overcome this complex problem.

A detailed water quality study carried out in 2006 and 2007 revealed the turnover of water during December. Temperature and DO values remained almost constant along the water column until March, with typical values of about 5–6 °C and 10.3–13.5 mg O₂/L, respectively. The temperature increase recorded in April and May resulted in gradual stratification of the water which, however, was not associated with the appearance of any negative effects, as shown in Table 2. The table presents the concentration range of various physicochemical parameters during the wet (December–May) and dry (June–November) periods, at different water depths. Water stratification progressed during the summer and resulted in oxygen depletion at depths greater than 10 m by the end of June.

### Table 1 | Typical physicochemical characteristics of water from Dipotamos reservoir

<table>
<thead>
<tr>
<th>Physicochemical parameters</th>
<th>Cations mg/L</th>
<th>Anions mg/L</th>
</tr>
</thead>
<tbody>
<tr>
<td>pH 7.6</td>
<td>Na⁺ 11</td>
<td>Cl⁻ 15</td>
</tr>
<tr>
<td>Conductivity, µS/cm 335</td>
<td>K⁺ 1.5</td>
<td>HCO₃⁻ 135</td>
</tr>
<tr>
<td>Suspended solids, mg/L 1.1</td>
<td>Ca²⁺ 40</td>
<td>SO₄²⁻ 26</td>
</tr>
<tr>
<td>Total organic carbon, mg/L</td>
<td>Mg²⁺ 9</td>
<td>NO₃⁻ 3</td>
</tr>
<tr>
<td>Boron, mg/L 0.3</td>
<td>Sr²⁺ 0.1</td>
<td>NO₂⁻ &lt;0.01</td>
</tr>
<tr>
<td>SiO₂ 6 1/2</td>
<td>Li⁺ &lt;0.02</td>
<td>PO₄³⁻ 0.02</td>
</tr>
</tbody>
</table>

**Figure 1** | Temperature and dissolved oxygen concentration as a function of depth at the withdrawal sampling points (21–10–2005).
The anoxic state of the hypolimnion favored the dissolution of iron and manganese at the end of the dry season. In addition, formation of sulfide was observed in the hypolimnion in the same period. Surface water reached its maximum annual temperature of around 25.4 °C in July and August, while the water changed color and turned light green, with a simultaneous development of taste and odor compounds (MIB and geosmin), due to the presence of microalgae. Although air temperature decreased gradually during autumn, the stratification side-effects intensified in September and October.

**RESERVOIR MANAGEMENT SOLUTIONS**

Several alternative strategies were examined to manage the algal bloom and the low water quality. Upgrading the existing water treatment plant by installing additional treatment processes was considered, but a cost-benefit analysis revealed that this would impose high construction and operating costs. As a result, other solutions were considered including the application of prevention measures.

One environmentally friendly approach would be the abstraction of water from layers with lower concentrations of nuisance compounds. This selective withdrawal approach was implemented during 2007 and 2008. The organoleptic characteristics of the abstracted water in that period were almost unacceptable, although the concentration of the compounds influenced by stratification (Table 2) marginally met the corresponding drinking water standards (European Council Directive 80/1998). Following this, artificial destratification measures were selected to improve the raw water quality in Dipotamos reservoir. Two artificial destratification techniques are common today: air injection and mechanical mixing. Mechanical circulation systems include surface spray units, impeller-aspirators, and pump-and-cascade systems. Although these systems do set up a circulation pattern, they are not typically designed to destratify a lake, but have higher applicability in non-stratified (shallow) lakes and ponds to increase water oxygen content.

The primary types of air injection systems currently in use include airlift aerators, Speece Cones and bubble plume diffusers, with the latter being the most suitable and widespread process for deep lakes (Schladow 1993; Singleton & Little 2006; Gafsi et al. 2009). In this system, an air compressor is placed safely on land, and supplies air to one or more perforated pipes which are located near the lake, or reservoir, bottom, typically at the deepest part.

A number of design approaches that have appeared in the literature are empirical by nature (Lorenzen & Fast 1977; Davis 1980). A significant improvement was made by a model based on the interaction that occurs between a buoyant bubble plume and the density stratified water column through which it rises (Schladow 1993), as well as by the discrete bubble model which unifies the

### Table 2  Concentration range of the physicochemical parameters influenced by stratification in the water column at the withdrawal sampling points

<table>
<thead>
<tr>
<th>Parameter</th>
<th>Wet season, 01/12/2006–31/05/2007</th>
<th>Dry season, 01/06/2007–30/11/2007</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>0 m</td>
<td>10 m</td>
</tr>
<tr>
<td>DO, mg/L</td>
<td>8.3–13.5</td>
<td>5.5–10.3</td>
</tr>
<tr>
<td>Temperature, °C</td>
<td>6–22.2</td>
<td>5–12.5</td>
</tr>
<tr>
<td>Fe²⁺, mg/L</td>
<td>ND</td>
<td>ND</td>
</tr>
<tr>
<td>Mn²⁺, mg/L</td>
<td>ND</td>
<td>ND</td>
</tr>
<tr>
<td>NH₄⁺, mg/L</td>
<td>ND</td>
<td>ND-0.15</td>
</tr>
<tr>
<td>H₂S, mg/L</td>
<td>ND</td>
<td>ND</td>
</tr>
<tr>
<td>MIB, μg/L</td>
<td>ND</td>
<td>ND</td>
</tr>
<tr>
<td>Geosmin, μg/L</td>
<td>ND</td>
<td>ND</td>
</tr>
</tbody>
</table>

ND: not detected.
hylomnnetic aeration and oxygenation system design (McGinnis & Little 2002; Singleton & Little 2006; Singleton et al. 2007). However, although it is empirical by nature, the Davis model has been calibrated for depths less than 30 m, volumes less than $30 \times 10^3 \text{ m}^3$, and temperature differences less than $4 \, ^\circ\text{C}$, and all these constraints are met at Dipotamos reservoir. Taking also into account that the Davis model is simple, it was selected for the destratification process design in Dipotamos reservoir.

**DESIGN OF THE DESTRATIFICATION PROCESS FOR DIPOTAMOS RESERVOIR**

Compressed air injection through perforated pipes was selected to be installed to destratify the Dipotamos reservoir. The locations of the perforated pipes are shown in Figure 2. The Davis model was implemented for process design and was based on the energy ($E$) required to destratify a certain volume of water. Firstly, the Potential Energy of Stratified system (PES) and the Potential Energy of Mixed system (PEM) were calculated through Equations (1) and (2), by applying the values of critical parameters given in Table 3, as follows:

$$
PES = g \Sigma \rho_i V_i h_i \text{(joules)} \quad (1)$$

$$
PEM = g \Sigma \rho_{im} V_i h_i \text{(joules)} \quad (2)$$

where $\rho_i$ and $\rho_{im}$ are the water densities of each water layer (kg/m$^3$), $V_i$ is the volume of each water layer (m$^3$), $h_i$ is the height of the center of each layer from the bottom of the reservoir (m), and $g$ is the acceleration force due to gravity (9.81 m/s$^2$). The stability ($S$) factor is equal to:

$$
S = PEM - PES \text{(joules)} \quad (3)
$$

The energy required for destratification ($E$) is calculated by Equation (4):

$$
E = S + R - W \quad (4)
$$

where $R$ is the solar heat, approximated as 5 J/m$^2$ per day required for destratification, and $W$ is the wind energy which is neglected in a conservative approach, since wind velocity during the artificial mixing period (summer–early autumn) is usually extremely low.

The required air flow rate ($Q$) in L/s is then calculated by Equation (5):

$$
Q = \frac{0.196 E}{T \ln \left(1 + \frac{D}{10.4}\right)} \quad (5)
$$

where $T$ is the time to achieve destratification (s), and $D$ is the perforated pipe's depth (m). Finally, from the volume of water to be destratified ($V$, m$^3$) and the air flow rate

![Figure 2](https://iwaponline.com/ws/article-pdf/13/4/1046/414999/1046.pdf)
(L/s), the length (m) of perforated pipe was calculated by Equation (6):

\[
L = \frac{V^3(1 + \frac{D}{10.4})}{T^3Q\left(\ln\left(1 + \frac{D}{10.4}\right)\right)^3}^{1/2}
\]  

(6)

Since the monthly water volume abstraction barely exceeded 0.6 × 10^6 m^3, and considering the meandric flow type of the reservoir (Figure 2), the artificial destratification process was applied to a water volume equal to 6 months’ consumption, located in an area ca 250 m from the withdrawal point. Based on the assumption that the destratification process should be applied around the end of spring, the required theoretical energy for destratification, the corresponding air flow and the length and size of the perforated pipes were calculated for a temperature range of 18–22 °C (for the epilimnion) and 5–20 °C (for the hypolimnion).

The results arising from the Davis model showed that the higher the difference between epilimnion and hypolimnion temperatures, the greater the energy required for destratification and the higher the associated air flow rate (Figure 3).

As shown in Figure 3, to ensure a temperature difference of 4 °C in the water column, a specific air flow rate value of 9.9 m^3/km^2/min is necessary (Table 1s, Supporting information, available online at http://www.iwaponline.com/ws/013/082.pdf). It must be noted that this value is close to the criterion set by Lorenzen & Fast (1977) corresponding to 9.2 m^3/km^2/min. Therefore, the required air flow rate for mixing the selected area of the reservoir, with a temperature difference of 4 °C between the epilimnion and hypolimnion layers, was calculated to be:

\[
Q = 9.9 \text{ m}^3/\text{km}^2/\text{min} \times 0.2 \text{ km}^2 = 1.98 \text{ m}^3/\text{min} - 33 \text{ L/s}
\]

(7)

An Atlas Copco GA15VSD15 air compressor system (controlled by inverter), with a flow rate of between 20 and 50 L/s and power of 15 kW, was installed on land, resulting in a specific air flow rate range of between 6 and 15 m^3/km^2/min. The compressed air was cooled to a temperature of below 40 °C as a safety precaution to protect the polyethylene perforated pipes. The Donaldson oil removal system includes a pre-filter to retain oil particles larger than 0.01 mm and an activated carbon filter for complete oil removal. The 1 m^3 air tank helps to normalize the compressor’s function and three flow meters (0–50 L/s) provide accurate air distribution (Figures 1s, 2s, 3s and 4s, Supporting information, available online at http://www.iwaponline.com/ws/013/082.pdf).

A thorough survey on pipe sizing-up showed that the demand for a headloss of less than 2 m of water column (mwc) for every 100 m pipe length can be safely achieved by a Ø 40 and Ø 50 mm pipe for air flows of 10 and 20 L/s, respectively (Table 2s, Supporting information, available online at http://www.iwaponline.com/ws/013/082.pdf). Polyethylene pipes were selected due to their low cost and their high resistance to corrosion. A high pressure resistant (10 atm) Ø50 mm pipe was installed in lines A and B (Figure 2), providing Q_{max} = 20 L/s, whereas a pipe with Ø40 mm was selected for line C, giving a flow rate of Q_{max} = 10 L/s. In-situ measurements showed that the headloss in all pipelines, rating at Q_{max} was kept well below 2 mwc,
thus validating the accuracy of the theoretical calculations. To counterbalance air diffusion uplift, the installed perforated pipes were connected with Ø 8 mm stainless steel wire, which was attached to cement blocks.

A significant drawback of the Davis model is the calculation of the required length (7–22 m) for the air diffusion pipes (Table 1s, Supporting information). Installation of such a small pipeline to mix a 0.2 km² surface area of meander shape more than 500 m in length appears quite ineffective from an engineering point of view. To cope with this issue, the required pipe length was estimated using the Davis model criterion of holes spacing at around 0.1 × depth (m) of injecting air. Using the latter pre-assumptions, three pipelines of 800 m total length were installed (Figure 2). Pipes were perforated with 1 mm diameter holes, spaced at 1 m intervals, since oxygen transfer within the hypolimnion was found to be inversely proportional to the initial bubble radius, for radii greater than approximately 1 mm (Singleton et al. 2007).

**Start-up of the destratification process**

Air injection commenced on October 7th 2008, when the values of the water’s critical quality parameters were close to those in Table 2. At this time the water also emitted a strong earthy odor due to the presence of geosmin and MIB. The air injection process, rating at a specific air flux of 15 m³/km²/min, reached equilibrium in about 1 h from compressor start-up and mild surface turbulence appeared, caused by injecting air through the pipelines. As the air injection process continued, the geosmin odor gradually decreased, and about 1 week later no organoleptic characteristics were detectable in the water. The corresponding results of DO concentration and temperature as a function of water column depth, at various time periods are shown in Figure 4 for samples collected from sampling point 1. The observed change in DO concentrations was quite interesting, since at the water abstraction point (20 m depth) the DO concentration measured almost zero at air injection start-up and then increased gradually to 1.2, 1.8 and 4.8 mg/L at the end of the 1st, 2nd and 4th week, respectively. In contrast, DO concentrations at the surface layer decreased from 8.3 ± 0.2 to 7.7 ± 0.2 mg/L. A better illustration of DO variation is shown in terms of saturation level. The oxygen saturation level at the surface layer during that 4-week period decreased from 89 ± 3% (8.3 ± 0.2 mg/L at 18 °C) to 76 ± 3% (7.7 ± 0.2 mg/L at 14 °C). It should be clarified that at a given altitude above sea level the DO saturation level is reversely related with temperature. Consequently, the oxygen diffusion rate from the atmosphere to the surface water, along with the oxygen supported by the diffused air from the perforated pipes, was lower than the oxygen consumption rate, since the latter supported chemical and biochemical activity for the oxidation of Fe²⁺, Mn²⁺, NH₄⁺, H₂S and organic matter throughout the reservoir. Similar to DO, a homogenization of temperature profile was observed during the air

![Figure 4](https://iwaponline.com/ws/article-pdf/13/4/1052/414999/1054.pdf)
injection process. The temperature difference between the epilimnion and hypolimnion layers prior to air injection, which exceeded 10°C, became more uniform after applying the destratification measures.

At the end of the 2nd week the temperature difference between the epilimnion and hypolimnion was around 4°C, while at the end of week 4, the water temperature almost reached equilibrium without any stratification (see Figure 4). Similar results were also observed for all the other sampling points, even for sampling point 5, which is located 300 m away from the end of perforated pipe A. From these results it can be estimated that the total volume of water mixed was around $5 \times 10^6$ m$^3$, which is well over the assumed value of $3.2 \times 10^6$ m$^3$ used in the Davis model as a design parameter. Conclusively, by practicing compressed air injection with an energy consumption as low as 55 Wh/10$^3$ m$^3$/d, which results in a daily energy cost of around 30 €, good quality tap water is ensured for the $80 \times 10^3$ habitants of Alexandroupolis city. In addition, it must be highlighted that the capital cost of this project was close to $100 \times 10^3$ €, which rendered functional an infrastructure (dam) that cost around $20 \times 10^6$ €.

RESULTS OF LONG-TERM OPERATION

Following the preliminary results that indicated the promising destratification potential of the applied method, a long-term operation was established. During the winter, temperature and DO presented almost uniform profiles through the water column, with the concentration of oxygen being close to saturation values. These conditions remained constant until the middle of spring (April) of every year. During this time no nuisance compounds were detected, indicating a healthy reservoir. The variation of temperature and DO values in the water column at sampling point 1, on specific days in 2009 is presented in Figure 5, while similar results were observed during 2010, 2011 and 2012.

Air injection commenced in the first days of May (spring) each year, when temperature differences between the epilimnion and hypolimnion layers were greater than 4°C, and ceased by the end of October (autumn). The temperature difference dropped to below 4°C within 2 weeks of start-up, and remained almost constant throughout summer and autumn. Consequently, an almost uniform temperature profile was ensured all year round due to the air injection method. The reservoir’s temperature ranged from between 5°C in January and 25°C in summer (July and August). In addition, a side-effect of destratification was observed. The homogenization of the water temperature during the summer months created a buffer of heat energy which prolonged the temperature decrease time in October (as shown in Figures 4 and 5).

The DO concentration in the water column was close to the corresponding saturation value from January to June,
and from September to the end of the year (Figure 5). The abrupt temperature increase in July led to gradual oxygen depletion at the bottom of the reservoir, resulting in a DO concentration close to zero. This annual DO distribution profile in the water column, which was considered marginally acceptable, was also observed in 2010, 2011 and 2012. The lowest DO concentration of 0.5–1 mg/L was observed around mid-August. This can be attributed to several factors acting synergistically. Firstly, the higher water temperature increases microorganism activity resulting in higher oxygen consumption. In addition, the high air temperature decreases the transfer rate of oxygen to water, due to both lower oxygen solubility and a decrease in compressor performance. This significant drawback indicates that more intense mixing during the high temperature period from mid-July to August should be practiced. However, DO almost reached the saturation level in the water column at the beginning of September, shortly after the decrease in air temperature observed at the end of August.

Following air injection implementation, water quality parameters were stabilized throughout the year with values close to those of Table 1, while nuisance compounds related to stratification met with drinking water Maximum Contaminant Limits (MCL). More specifically, the dissolved concentrations of Fe²⁺, Mn²⁺, NH₄⁺ and H₂S⁻ in abstracted water were always found to be below the detection limit of the method, which were estimated to be 0.05, 0.02, 0.05 and 0.05 mg/L, respectively, while the organoleptic characteristics of the water were at acceptable levels. Positive concentrations of Fe²⁺, Mn²⁺ and NH₄⁺, close to their detection limits, were detected during the middle of August near the bottom of the deepest part of the reservoir at sampling point 3. However, they did not influence the quality of the abstracted water. Conclusively, the well oxygenated reservoir, as reflected by the homogeneous oxygen profile in the entire water body, led to a significant increase in fish populations which are considered a biological indicator of a healthy reservoir.

It should be underlined that the reservoir water also met with MCL for pesticides, polynuclear aromatic hydrocarbons, polychlorinated biphenyls and herbicides, attributed to the environmentally safe watershed. The microbiological quality of abstracted water was good and stable and at 37 °C contained less than 3 total coliforms per 100 mL and less than 10 total cells per mL. Additionally, counts of *Escherichia coli* and streptococcus per 100 mL were not normally detected, although counts of 1 to 3 were measured occasionally. In contrast, heavy precipitation in the watershed significantly reduced the microbiological quality of the abstracted water for several days each year. In these cases, as many as 120 total cells per mL were measured at 37 °C, while counts of total coliforms, *Escherichia coli* and streptococcus did not exceed 20/mL. However, the conventional water treatment plant effectively eliminated these drawbacks and produced safe water to EU drinking water standards.

**CONCLUSIONS**

The application of a hypolymnetic air injection process with an energy consumption as low as 55 Wh/10³ m³/d was studied for the destratification of Dipotamos reservoir. The process design was based on the Davis model. Air injection commenced in the first days of May each year and ceased by the end of October for four consecutive years. In general, the variation of temperature in the water column was lower than 4 °C and the reservoir’s surface temperature ranged between 5 °C in January and 25 °C in July and August. DO concentration in the water column was close to the corresponding saturation value from January to June, and from September to the end of the year. The physicochemical and microbiological quality of the water was excellent and stable with minor exceptions for a few days every year during heavy precipitation.

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