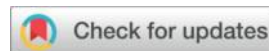




Eco-sustainable potential of *Pistia stratiotes* for cost-effective nitrogen removal from domestic wastewater in southern Algeria



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Abstract

This 12-month study (2023) evaluated the efficiency of *Pistia stratiotes* (water lettuce) in treating domestic wastewater in Brezina (El Bayadh), focusing on the removal of NH_4^+ , NO_3^- , and NO_2^- . The plant achieved high removal efficiencies of 97.50% (NH_4^+), 74.78% (NO_3^-), and 60.61% (NO_2^-), with an average annual nitrogen purification rate of 77.43%. Environmental conditions-temperatures ranging from 19.40–33.40 °C and an average pH of 7.64 - enhanced bacterial nitrification–denitrification processes, highlighting *Pistia stratiotes* as a sustainable, eco-friendly solution for reducing nitrogen pollution in arid and semiarid regions.

Keywords: *Pistia stratiotes*, domestic, wastewater, treatment, ammonium nitrogen, nitrates, nitrites.

1. Introduction

Nitrogen is one of the most essential nutrients for living organisms, especially plants, and is considered among the key elements involved in increasing agricultural productivity. From a reproductive perspective, its role is crucial. However, the presence of nitrogen at high concentrations in aquatic environments is undesirable, making it a negative factor. This duality presents a

contradiction: while nitrogen is vital for the nutrition of living organisms, it becomes harmful when its concentration in water increases, considering that water is a fundamental element for life. High nitrogen concentrations in water pose a serious threat to both animals and plants. This contradiction can be attributed to various factors, particularly changes in the physicochemical properties of water, such as pH, temperature, electrical conductivity, and dissolved oxygen levels, alongside increased turbidity and suspended solids. These alterations can intensify transpiration in green plants. A second negative impact of high nitrogen concentrations in water is biological stress. This can lead to reproductive stress in aquatic organisms, potentially causing their extinction. For these reasons, Algerian State has attracted particular interest in the scientific study of nitrogen pollution in water by identifying its causes, exploring methods to eliminate or reduce it, and increasing the degradation requirements for removing this pollutant from wastewater. This includes the implementation of wastewater treatment plants and the promotion of tertiary treatment to obtain clean effluents that comply with discharge standards. With this approach, Algeria has established 960 conventional mechanical wastewater treatment plants, such as activated sludge systems in Tizi Ouzou and Touggourt, artificially aerated lagoon systems in Ouargla, and naturally aerated lagoons such as the treatment plant in Ghardaïa, among others. However, these plants have shown limited efficiency in removing nitrogen pollution. As a result, nitrogen remains a persistent pollutant in treated domestic wastewater. To address this issue, we proposed the creation of pilot treatment systems adjacent to these conventional plants via phytoremediation techniques. Plants are living organisms that absorb nitrogen and its compounds through metabolic processes, incorporating them into biological structures such as chlorophyll. Notably, the use of constructed wetlands for wastewater treatment is not a recent practice. It dates back to 1950 in the United States. Studies have shown that the purification efficiency in treatment basins is influenced by biogeochemical factors, hydraulic conditions, and climatic factors, which cause variations in purification rates in constructed wetlands, even under similar climate conditions [1]. These variations are due mainly to weak hydraulic profiles that were not adequately considered during design. Key parameters such as hydraulic retention time, short-circuiting, thermal stratification, and dead zones influence water distribution and contact time between microbial communities (bacteria, protozoa, macroinvertebrates) and pollutants. These hydrodynamic dynamics directly affect the ability of plants to purify domestic wastewater [2]. Climatic conditions also play a crucial role in the hydraulic performance of basins, especially considering the volume of water to be treated and water losses due to plant transpiration, which must be accounted for when calculating treatment yield. To demonstrate the effectiveness of aquatic plants in treating domestic wastewater, we conducted a year- round (2023) study in the town of Brezina, El Bayad'h Province. This study assessed the nitrogen removal efficiency of *Pistia stratiotes* (water

lettuce) in domestic wastewater. To overcome the limitations mentioned above, we built a pilot wastewater treatment station employing a hybrid phytoremediation system (vertical and horizontal subsurface flow) located next to the conventional treatment plant in Brezina. The system utilized pretreated domestic water. In this research, this study aimed primarily to identify the core issue related to nitrogen, with the objective of optimizing its dual role both as an essential nutrient and as a potentially harmful contaminant across different components of the environment. This work also sought to enhance the vital process of reducing nitrogen concentrations in treated wastewater or, at the very least, minimizing them as much as possible.

2. Materials and Methods:

2.1. Presentation of the study area

The territory of the municipality of Brezina is located in southeastern El Bayadh Province, 72 km south of El Bayadh. It is an oasis located on the southern foothills of the Saharan Atlas, along the Seggeur River, in a landscape of gorges, remnants of Quaternary deposits. It is located between 33°-05' north latitude and 1°-15' east longitude. Between 849 m and 800 m above sea level. The city of Brizina is one of the main oases of the Algerian Sahara. It is located approximately 700 km from Algiers. It covers an area of 15,702 km. The population is estimated at 12,468 inhabitants on the basis of the growth rates from the official censuses of 2003, as shown in Fig. 1.



Fig. 1. The map of Algeria shows the location of the city of Brizina.

2.2. Description of the plants used in the treatment ponds

Aquatic plants play an active role in purifying wastewater, both domestically and industrially. Among these plants, *Pistia stratiotes*, which is a flower-shaped herb that floats freely on the water surface, was the subject of this study. Its origin is tropical, and its leaves resemble those of the edible lettuce plant, which is why it has earned the name "water lettuce." It develops ideally at a temperature of 18 °C or slightly above. It seems that its genus is monotypic, despite the existence of several species, forming a group of varieties distinguished by their size and leaf shape, with these differences attributed to environmental conditions [3]. The stem of this aquatic plant is characterized by its short length, around which the leaves grow in a dense and soft whorl. The leaves can reach a length of 15 cm and a width of 8 cm in their natural environment. The biometric characteristics of aquatic plants vary greatly from one plant to another, despite the water being rich in nutrients that grow in the artificial garden in Brizina, Al-Bayadh. Researchers have measured the sizes of *Pistia stratiotes* plants and reported that their diameter can reach a maximum of 45 cm [4]. In nutrient-rich polluted plants, the roots are initially short, but they develop and increase in length as the purification yield increases in treated lakes. Some researchers believe that secondary treatment is highly effective when the root length approaches the diameter of the plant [5]. The water conditions in which this plant grows can be stagnant, as the depth of the water body does not affect its growth and development. It reproduces rapidly in the presence of abundant nutrients in the water. Table 1 presents the plant calcification of the *Pistia stratiotes*.

Table 1. Botanical classification of the *Pistia stratiotes*

Taxonomic Rank	Classification
Kingdom	<i>Plantae</i>
Subkingdom	<i>Tracheobionta</i> (Vascular plants)
Division	<i>Magnoliophyta</i> (Angiosperms)
Class	<i>Liliopsida</i> (Monocotyledons)
Subclass	<i>Arecidae</i>
Order	<i>Arales</i>
Family	<i>Araceae</i>
Genus	<i>Pistia</i>
Species	<i>Pistia stratiotes</i> L.

2.3 Description of the experimental station used

In our experiment, we employed macrophyte-based treatment via a hybrid runoff system (horizontal + vertical) composed of a series of horizontal and vertical basins, as illustrated in Figure 2. This configuration is inspired by the model developed by researcher K. Seidel, which was previously applied with a limited number of basins in Germany, the USA, and France [6]. The terminal system consists of basins arranged in two parallel vertical levels interconnected by three horizontally aligned

basins in sequence. We selected this type of setup because of its high efficiency in enhancing the nitrification process, particularly in vertical flow basins. These basins offer a relatively high rate of aerobic ventilation, which promotes the activity of aerobic bacteria responsible for nitrogen removal. In contrast, the nitrogen removal efficiency in horizontal flow basins tends to be lower because of the limited availability of dissolved oxygen, which restricts the activity of nitrifying bacteria [7]. The experimental device used in our research consists of three identical plastic boxes of equal dimensions (50 cm in length, 35 cm in width, and 20 cm in depth), as shown in Fig. 2. These boxes are vertically filled with three stratified layers of substrate materials. The first and second boxes are filled with coarse and fine gravel, respectively, and planted with *Pistia stratiotes*. The third box is filled with sand. The boxes are interconnected by PVC pipes with a diameter of 50 mm, and each connection is equipped with a valve to regulate water flow following the designated hydraulic retention time, as illustrated in Fig. 2. Irrigation begins in the first and second boxes through vertical flow. After a residence period of seven days, the system transitions to horizontal flow, allowing water to reach the third box, where it remains for an additional three days. This setup results in a total hydraulic retention time (HRT) of 10 days. Following this period, water samples are collected from the outlet of the third box for analysis in the laboratory, which specializes in treated wastewater quality.

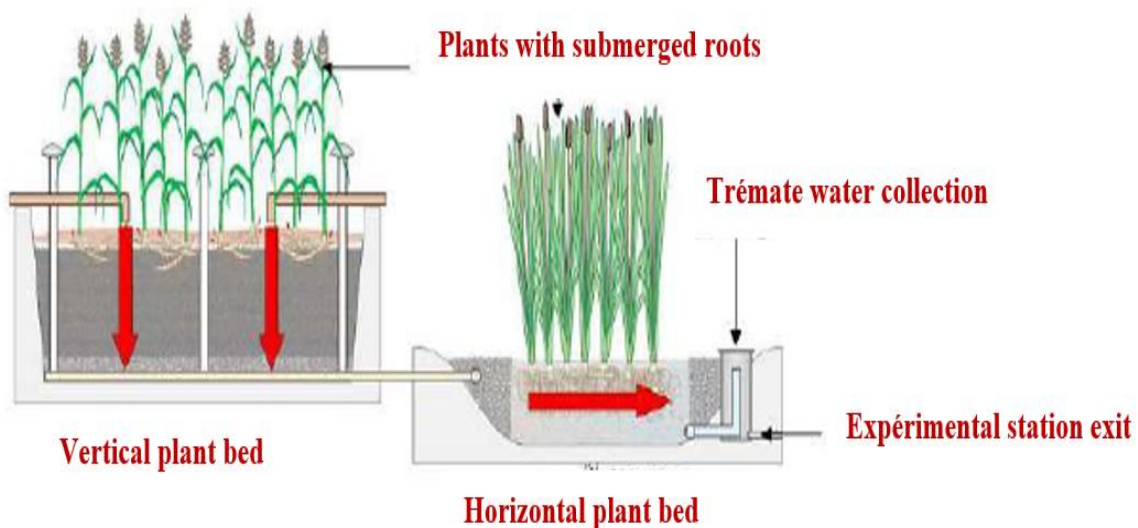


Fig 2. A treatment pond with various flow rates

2.4. Methods of sampling and analysis in the laboratory

The samples of raw domestic wastewater were collected from the tank of the classical station after the initial treatment, three times per month, via glass bottles. These samples were taken on the 1st,

10th, and 20th days of each month, i.e., before the wastewater entered the purification and disinfection basins. For the treated water, samples were also collected three times a month (on the 10th, 20th, and 30th days of the month) from the outlet of the Experimental Station, i.e., after the water had remained in the basins for a retention time of 10 days. All the samples were immediately transported to the laboratory of the classical station in Brezina in a container maintained at 4 °C. In accordance with the general guidelines for the preservation and handling of samples [8-10], the quantity of pollutants was determined according to the standard techniques for collecting and analyzing water samples established by the American Public Health Association [11]. and Algerian standards. To measure the temperature and pH values and the concentrations of nitrogen pollutants, we used the spectrometric devices referred to in Table 2 after each sampling, whether wastewater or treated water. We recorded the results obtained in Table 3.

Table 2. Desired physicochemical parameters.

Indicator	Symbol	Unit	Method of Analysis/Instrument
Temperature	T	°C	Direct reading using multiparameter meter (HANNA H19829)
Acidity	pH		Direct reading using multiparameter meter (HANNA H19829)
Total Suspended Solids	TSS	mg/L	Membrane filtration method
Nitrate	NO ₃ ⁻	mg/L	Test strips (Bandelettes-test)
Nitrite	NO ₂ ⁻	mg/L	Spectrophotometry (WTW Photolab S6)
Ammonium	NH ₄ ⁺	mg/L	Spectrophotometry (WTW Photolab S6)

Table 3. Monthly variation in physicochemical parameters at the station inlet and outlet (2023)

Parameter	Unit	Station	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
T	°C	Inlet	19.40	20.30	22.55	28.00	29.70	32.00	32.01	33.40	31.30	30.10	28.80	20.10
		Outlet	10.20	11.40	15.20	17.20	26.25	31.20	31.00	33.10	27.11	25.10	22.40	12.30
pH	-	Inlet	7.80	7.50	8.10	7.90	7.40	7.96	7.70	7.00	7.88	7.60	7.65	7.30
		Outlet	6.70	6.60	6.80	6.48	6.58	6.60	6.85	6.80	6.90	6.70	6.75	6.65
NO ₂ ⁻	mg/L	Inlet	0.54	0.20	0.34	0.33	0.50	0.70	0.90	0.05	0.104	0.10	0.108	0.168
		Outlet	0.057	0.04	0.031	0.053	0.01	0.02	0.03	0.01	0.031	0.02	0.03	0.023
NO ₃ ⁻	mg/L	Inlet	54.4	34.5	33.00	23.30	18.20	23.50	28.60	22.20	18.50	29.20	29.00	26.30

		Outlet	1.45	7.45	2.25	2.20	4.50	2.50	2.23	2.25	4.50	6.00	5.20	2.22
NH₄⁺	mg/L	Inlet	13.8	16.0	14.3	11.8	13.0	17.5	15.7	14.3	11.8	17.5	12.1	11.8
		Outlet	3.50	5.30	6.60	2.50	2.80	3.00	2.01	2.90	1.30	1.95	1.15	1.10

3. Results and discussion

Before the results are discussed, we explain how to calculate the average values monthly and then annually for one month. Our values are obtained by adding the values obtained each month and then dividing the total by 3, so we obtain the average value for each month. The process is repeated for wastewater at the station entrance or treated water at the station exit, and we record the values obtained in Table 3 above. To calculate the percentage of purification yield via Eq. (1) for each nitrogen pollutant, we applied relation N°1 [12].

$$\text{Deductions (\%)} = \frac{C_E - C_S}{C_E} \times 100 = 100 - \frac{C_S}{C_E} \times 100 \quad \text{Eq. 1}$$

where C_E and C_S are the concentrations in mg/L of the effluent at the inlet and outlet of the wastewater treatment plant, respectively.

3.1. The temporal evolution of physical pollution during the follow-up year 2023.

We represent physical pollution through curves that display the evolution of temperature T and acidic pH values in Figs. 3 and 4, respectively, at the entrance and exit of the station. We present them as follows:

3.1.1. The temporal evolution of the temperature (T) during the monitoring year 2023

The evolution of the water temperature revealed that the incoming (raw) wastewater had an average value of 25.12 ± 1.2 °C, whereas the outgoing (treated) water with a free surface averaged 21.53 ± 0.2 °C (Fig. 3). In the watery lettuce basins, the results represent the means of the three basins described in Fig. 2. These findings are consistent with those reported by Hammadi. B. et al. [13] and Atia, D.J. et al. [14]. Daily temperature fluctuations in December (the coldest month) were more pronounced in both microphyte and macrophyte systems; however, plant cover limited these variations to the surface layers. The temperature difference between raw and treated water ranged from 0.50 to 6.08 °C. Although this variation does not hinder bacterial or microbial activity, the purification efficiency improves with increasing temperature, particularly in warmer months [15]. Elevated ammonia concentrations occur during summer [16], may be toxic to fish, which is why aquaculture ponds are often covered with water to reduce overheating [17]. This plant cover also mitigates ammonia emissions by limiting algal proliferation.

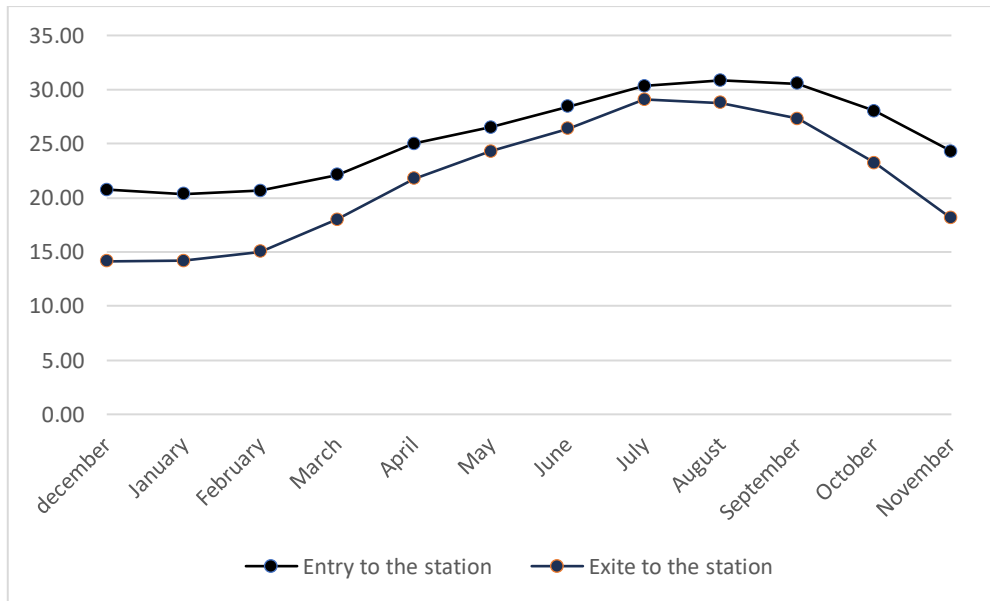


Fig 3. The temporal evolution of temperature (T) during the monitoring year 2023

3.1.2. Temporal evolution of pH acidity

The evolution of pH is illustrated in Fig. 4. The average pH of the incoming (raw) wastewater was 7.85. After a 10-day retention period in the planted basins with macrophytes, the treated water exhibited a pH of 7.65, which falls within the range reported by Cristina et al. [18]. and Bendida A. et al. [19]. Over a period of 10 months, the pH in the planted basins gradually increased by approximately one unit, reaching 8.04, a change attributed to algal photosynthetic activity. Similarly, in the basins covered with water, the pH rose from 7.39 to 8.04, reflecting an almost one-unit increase in alkalinity. These findings are consistent with results from Ghana, where Awuah et al. [20]. observed that the pH decreased by up to two units when the culture age reached four weeks. Furthermore, our investigation confirmed that plant growth remains normal and optimal despite pH fluctuations, provided that values remain within the range of 4–6, in agreement with the conclusions of Khedr A.H.A. et al. [21].

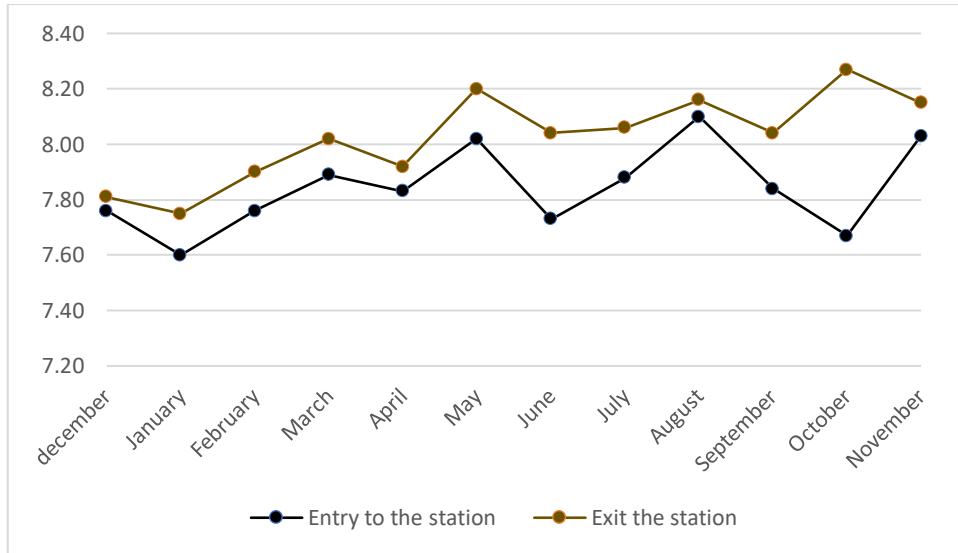
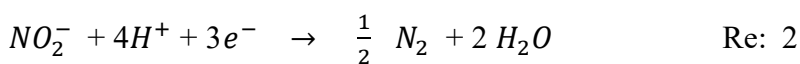
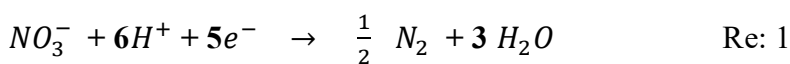


Fig. 4. Temporal evolution of pH acidity during the monitoring year 2023

The observed variations in pH can be attributed to several factors. The activity of nitrate-oxidizing bacteria leads to the release and accumulation of H^+ ions, contributing to acidification of the aquatic environment. In addition, pH changes may result from the abundance of CO_2 generated during photosynthesis and the decomposition of organic matter by heterotrophic bacteria. Plants also absorb cations as part of mineral nutrition, which is compensated by the release of H^+ ions, thereby increasing acidity [22]. Furthermore, plant roots secrete organic acids into the medium [23]. During the anaerobic stage of raw sewage treatment, a slight decrease in pH is observed, which continues progressively throughout the process, leading to a reduction from 7.85 at the inlet to 7.65 at the outlet. This decrease is associated with the activity of nitrate-oxidizing bacteria, which grow within a pH range of 5–8 [24] with an optimal range of 7.5–8.5 [25]. The observed pH evolution is therefore justified by the nitrogen removal process, which can be explained by the following chemical reactions.



During the study period, the pH of the untreated wastewater ranged from 7.60 to 8.10, with an average of 7.85, reflecting the typical characteristics of urban effluents. In the treated wastewater at the outlet of the purification station, the pH ranged from 7.39 to 8.04, with an average of 7.65 in 2023. These values comply with Algerian discharge standards as well as those of the World Health Organization (WHO) and the Food and Agriculture Organization (FAO), thus classifying the treated water as

suitable for agricultural irrigation. Previous studies [26-27] have shown that pH variations in wastewater treatment ponds are driven primarily by biological and biochemical processes, particularly photosynthesis. Furthermore, in ponds covered with floating plants, the pH values remain relatively stable (6.5–7.5), indicating that this parameter has minimal influence on phosphorus precipitation [28].

3.2. Degradation of Nitrogenous Pollution by the Water Lettuce Plant (*Pistia stratiotes*)

Plant growth in rivers, lakes, and ponds is a natural process driven by photosynthesis, which produces organic matter. However, when this growth becomes excessive and unbalanced (eutrophication), it leads to aquatic ecosystem degradation and biodiversity loss. Like terrestrial plants, aquatic plants require essential nutrients, primarily nitrogen (N) and phosphorus (P), which originate from soil leaching or urban wastewater discharges. The nitrogen-to-phosphorus (N:P) ratio varies considerably across different sectors of the hydrographic network. This study investigated the potential of watery lettuce (*Pistia stratiotes*) for the purification of domestic wastewater in the Brezina (El Bayadh) region, with a particular focus on its efficiency in removing nitrogen compounds occurring at significant concentrations. To achieve this goal, we measured the physicochemical parameters of wastewater at both the inlet and outlet of treatment filters and analyzed the temporal dynamics of nitrogenous pollutants during the monitoring year 2023. This approach allows the assessment of *Pistia stratiotes* purification performance and its contribution to nitrogen removal.

3.2.1. Reduction in ammonium nitrogen ($N-NH_4^+$)

Fig. 5 shows the variation in ammonium concentration ($N-NH_4^+$) at the inlet and outlet of the treatment plant throughout 2023. This reduction is attributed to the reciprocal symbiosis between bacteria and plants, where bacteria exploit the oxygen released during photosynthetic enzymatic reactions to degrade nitrogen pollution, thereby lowering ammonia contamination ($N-NH_4^+$).

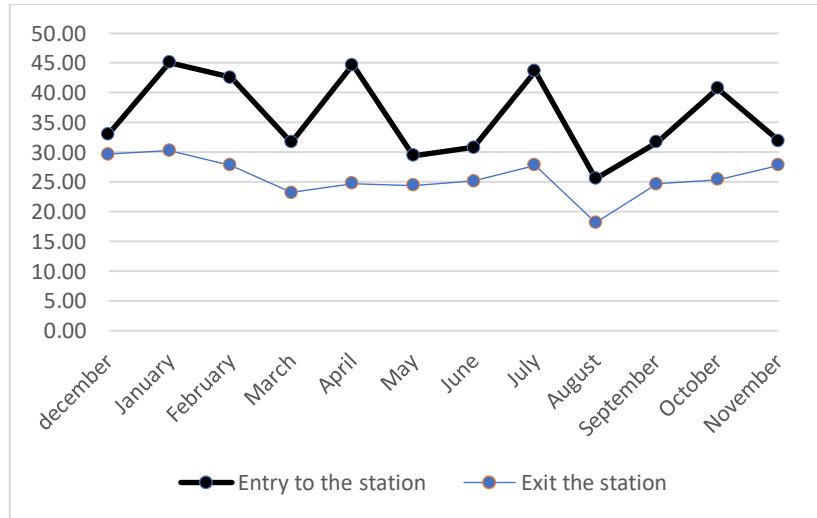
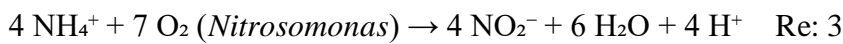
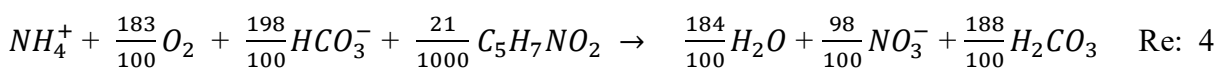


Fig. 5. The temporal evolution of N-NH₄⁺ in mg/L during the monitoring year 2023

The ammonium (N-NH₄⁺) concentrations at the basin inlets ranged between 25.57 and 45.08 mg/L (Fig. 5). These values are consistent with those reported by Tama et al. [29] and remain lower than the maximum recorded by Bendida et al. [19]. The conversion of organic nitrogen into ammonia, known as ammonification, constitutes a major source of ammonium and occurs under both aerobic and anaerobic conditions. At the station outlet, a purification yield of 97.50% was achieved, exceeding the results obtained by Bendida et al. [19] and Tama et al. [29] and far surpassing the 24% and 86.5% removal efficiencies reported by Petemanagnan Ouattara J.M. et al. [30]. This improvement is attributed to the biological oxidation process, whereby part of the ammonium is sequentially transformed into nitrite (NO₂⁻) and nitrate (NO₃⁻) by nitrifying bacteria [31-32]. Aquatic plants, with their extensive internal air spaces, actively transport oxygen to roots and rhizomes, thereby increasing bacterial activity. The remaining fraction of ammonium is directly assimilated by plants for growth [33-34]. Overall, ammonium removal follows two sequential oxidation steps: the conversion of NH₄⁺ into NO₂⁻ and subsequently into NO₃⁻ under aerobic conditions, as described by the following chemical reactions.



Nitrification is an autotrophic bacterial process in which the oxidation reactions can be summarized by the following chemical equation:



Two types of nitrifications exist: heterotrophic and autotrophic, both of which use inorganic substrates as energy sources. Heterotrophic nitrification is performed by various bacteria, fungi, and

algae, and autotrophic bacteria play a key role in ammonia removal. As shown in Fig. 5, nitrogen-fixing aerobic autotrophic bacteria reduced ammonia from an average of 35.94 mg/L in raw wastewater to 1.42 mg/L in treated effluent, achieving a purification efficiency of 96.07%, which is higher than the plant-based purification rates reported by Chan [35] (80.5–87.63%) and Bendida [19] (79.27–81.99%). The process occurs through a first-order oxidation reaction leading to nitrate conversion into N₂, although ammonium reduction here appeared to be independent of the applied load, indicating zero-order kinetics. This may be explained by sufficient ammonium concentrations for maximum *Nitrosomonas* growth (oxygen saturation of 0.03–1.3 mg/L) and stable nitrogen uptake by algae. Nitrification is effective within a broad temperature range, from 5 °C (Niquette et al., 1998) to 40–45 °C, with an optimum temperature range between 25 °C and 36 °C. During this study, the water temperature varied from 25.12 °C at the inlet to 21.53 °C at the outlet (Fig. 3), with representative values presented in Table 4.

Table 4. Some values of the optimum growth temperature for nitrifying bacteria.

Optimal Temperature (°C)	References
25 °C	Bambelle et al.1992[36].
30 – 36 °C	Ford et al.1980[37].
30 °C	Henze et al.1996[38].; Wang et al. 2004[39].

Most authors agree that the van’s Hoff–Arrhenius law describes the effect of temperature on microbial growth, with an optimum range of 5–30 °C, which is consistent with our findings. In our study, the average pH values ranged from 7.60 to 8.04 (Fig. 4), which is in agreement with the optimum pH of 8.5 reported by Henze et al. [38].

3.2.2. The reduction of nitrates ($N-NO_3^-$)

Fig. 6 shows that $N-NO_3^-$ originates from the biological oxidation of different nitrogen forms (organic N, NH_4^+ , NO_2^-), and although it may transform into nitrites with indirect toxicity, no carcinogenic effects have been demonstrated [40]. The optimal activity of nitrifying organisms occurs at pH 7.5–8, which is consistent with our results in Fig. 4 [40].

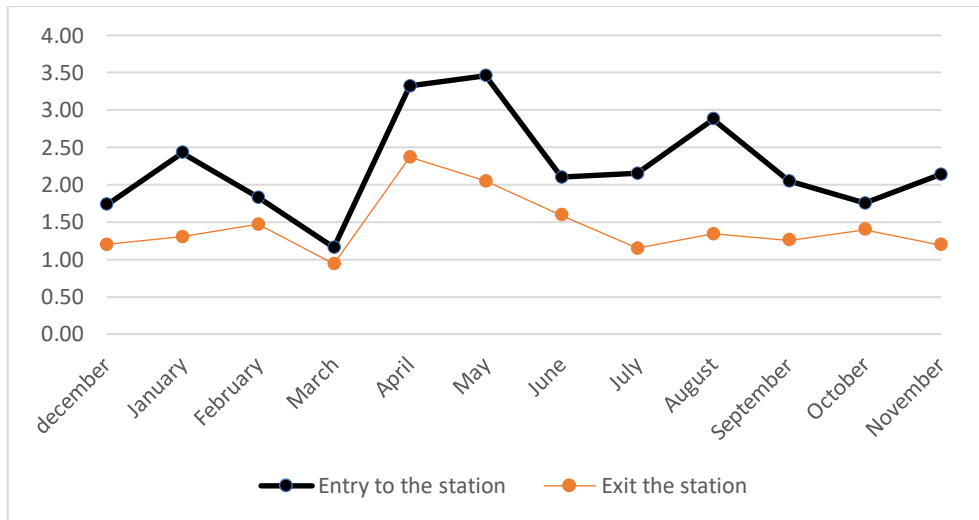
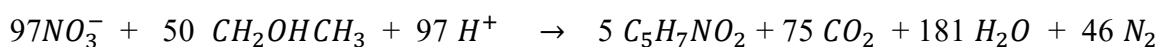
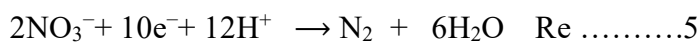


Fig. 6. The temporal evolution of N-NO₃⁻ (mg/L) during the monitoring year 2023

According to the results obtained, nitrate concentrations in treated wastewater (average 0.82 mg/L) remained significantly lower than the international standards for irrigation water set by the WHO and FAO (< 50 mg/L). Compared with raw water (1.16 mg/L), the average removal efficiency was approximately 29.31%, which is relatively low compared with the findings of [41]. At the station inlet, nitrate (N-NO₃⁻) reached 2.25 mg/L but decreased to an average of 2.25 mg/L at the outlet after 10 days of retention in the basin-values still lower than those reported by Hammadi. B. et al. [13]. Temporal variations are illustrated in Fig. 6, which shows that N-NO₃⁻ in raw wastewater ranged from 1.16 mg/L in March to 3.46 mg/L in May, whereas in treated water, it varied from 0.44 mg/L in October to 1.20 mg/L in April. The highest nitrate mass removal occurred in May (2.31 mg/L; 66.77% efficiency), whereas the lowest was recorded in March (0.71 mg/L; 61.21%). On an annual basis, the average removal of N-NO₃⁻ was 0.82 mg/L (29.31%), which is comparable to the reduction efficiency reported by [42] (36.13%). The observed increase in nitrate in planted bed filtrates confirms the occurrence of nitrification, whereas denitrification explains the overall nitrate reduction. This process involves the transformation of nitrates (N-NO₃⁻) into nitrogen gas (N₂) under anoxic conditions by heterotrophic bacteria using organic carbon as an energy source [43] and can be summarized by the following reactions:

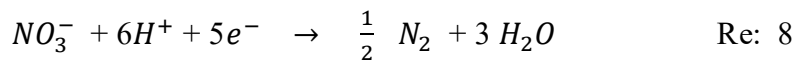
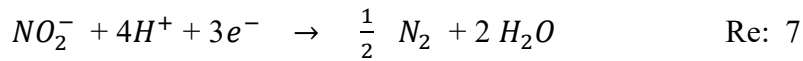


Re: 6

Organics mater

Biomass

The decrease in nitrate pollution is attributed to the biochemical reactions that promote the growth of aerobic bacteria that utilize nitrates formed during nitrification. These bacteria employ $N-NO_3^-$ as an electron acceptor in anaerobic respiration, provided that sufficient BOD_5 is available to sustain heterotrophic organisms in an oxygen-free medium. Thus, nitrate removal is governed by this mechanism and can be further illustrated by the following reactions.



The monitoring results (Fig. 6) reveal a temporal variation in nitrate ion ($N-NO_3^-$) concentrations, with values ranging from 2.25 to 0.44 mg/L, corresponding to a denitrification efficiency of 74.18%. This efficiency is attributed to the activity of heterotrophic bacteria under anoxic stress conditions. During the study period, concentrations varied between 1.16 and 3.46 mg/L, reflecting the rapid reduction of nitrates in the environment. The fluctuations observed in the treatment basins are closely linked to the applied nitrogen loads: under low organic loads, nitrate levels tend to increase, particularly in spring and summer, whereas under relatively high loads, nitrate concentrations either stagnate or decrease.

3.2.3. The parameters of nitrites $N-NO_2^-$

The variation in nitrite ($N-NO_2^-$) concentrations in raw and treated wastewater during 2023 is illustrated in Fig. 7. At the inlet of the treatment plant, the average concentration reached 1.65 mg/L, in contrast to 0.45 mg/L at the outlet, with an intermediate value of 0.95 mg/L in the cultivated basins, reflecting microbial oxidation of NH_4^+ and yielding an efficiency of 56.81%. Seasonal fluctuations were observed: in raw water, concentrations ranged from 1.21 mg/L in spring to 2.20 mg/L in autumn (27.60 °C), whereas in treated water, values varied between 0.13 mg/L in spring and 0.95 mg/L in autumn (21.35 °C). The nitrite removal efficiency also showed temporal variation, with a maximum yield of 90.48% in January (1.90 mg/L removed) and a minimum yield of 53.49% in September (0.93 mg/L removed). On average, the mass of nitrite removed was 1.20 mg/L, corresponding to a treatment efficiency of 60.61%.

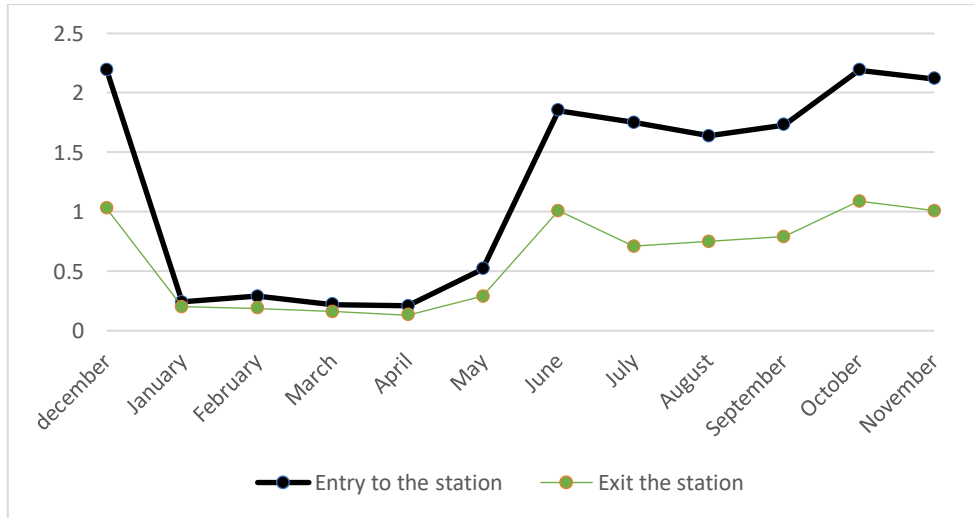
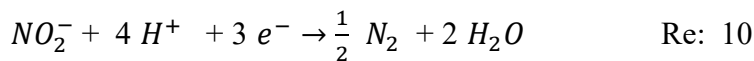


Fig. 7. The temporal evolution of N-NO₂⁻ in mg/L during the monitoring year 2023

The elimination of N-NO₂⁻ occurs primarily through the oxidation of nitrite ions into nitrate, as represented by the reactions below. Alternatively, this process may proceed via denitrification, as described in the following reactions [44-46].



Either by direct reduction to nitrogen gas:



Strong correlations have been established between nitrogen removal efficiency and both initial nitrogen concentrations and plant density [47]. This explains the high purification yield of the studied system for nitrogen reduction, estimated at 87%. Ammonium is recognized as the preferred nitrogen source for aquatic plants [48-50], whereas nitrate absorption occurs through enzymatic processes that modulate ammonium uptake, which is limited to the photosynthetic phase. High ammonium concentrations in domestic wastewater may inhibit nitrate assimilation [51] a phenomenon confirmed in our study on *Pistia stratiotes*. In wastewater treatment systems, ammonium uptake by aquatic plants can reach a maximum efficiency. Plant density, wastewater composition, and the balance between leaf nitrogen content and ambient nitrogen strongly influence ammonium removal. Our results also revealed that maximum plant growth occurred when nitrogen concentrations exceeded 5 mg/L, which is consistent with previous findings [52]. Furthermore, aquatic plants provide surfaces for nitrogen-fixing bacteria, which form biofilms that enhance nitrification and subsequent

denitrification, ultimately converting nitrates into atmospheric nitrogen. This coupled process is particularly efficient under high nitrogen loads, with overall removal yields exceeding 60% in cultivated systems [53]

Conclusion:

Nitrogen removal is generally achieved through biological treatments, particularly nitrification–denitrification, or via ion exchange. During biological processes, nitrogen undergoes successive transformations: from nitrite (NO_2^-) to nitrate (NO_3^-) and ultimately to the gaseous form, with each compound differing in molecular weight. Effective monitoring requires daily measurements based on either nitrogen mass or moles. Aquatic plants play a crucial role in this purification, as they promote nitrification reactions and assimilate nitrogen into their tissues through enzymatic activity. In basins fully covered with vegetation, nitrogen occurs in organic forms (sediments, detritus) and mineral forms (NH_4^+ , NO_3^-). Our results highlight the remarkable efficiency of *Pistia stratiotes*, which achieved removal rates of 97.50% for ammonium (NH_4^+), 74.18% for nitrate (NO_3^-), and 60.61% for nitrite (NO_2^-), with an overall average of 77.43%. These findings confirm the high potential of *Pistia stratiotes* for nitrogen removal and suggest that intensive coverage of treatment basins with this macrophyte can further increase domestic wastewater purification, particularly in arid and semiarid regions.

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