MEMBA Su Bilimleri Dergisi

https://dergipark.org.tr/tr/pub/memba

Nutrients in Sediment and Overlying Water in the Homa Lagoon of İzmir Bay (Aegean Sea, Turkey)

Emine Erdem Cingöz[1*], Hasan Baha Büyükışık[2]

Abstract: Benthic fluxes of biogenic compounds play a critical role in the biogeochemistry of shallow aquatic ecosystems. Measuring these fluxes at the lagoon scale is challenging due to heterogeneous sediment structures and the combination of diffusion and advective transport processes at the sediment-water interface. In this study, monthly nutrient flux experiments were conducted in the sediment of İzmir Bay Homa Lagoon (Aegean Sea, Turkey). The flux of reactive silicate (RSi) ranged from 14.6 to 255.24 µgatSi/m²/hour, while the ammonium (NH_4^+) flux varied between 3.59 and 95.8 µgatN/m²/hour. The nitrite (NO_2^-) flux ranged from 0.93 to 13.99 µgatN/m²/hour, and the nitrate (NO_3^-) flux varied from 27.76 to 300 µgatN/m²/hour. The flux of reactive phosphorus (RP) ranged from 0.74 to 5.80 µgatP/m²/hour. The research indicated that the RSi flux peaked during the summer months, while the NO_3^- flux occurred in both winter and summer, transferring nutrients from sediment to water and vice versa. It was determined that the NO_2^- flux significantly transferred to the sediment, except during the summer months. The RP flux flowed from sediment to water during the summer, while it was bound to the sediment during mid-winter and autumn. The NH_4^+ flux showed transitions from sediment to water in winter and from water to sediment in summer. In conclusion, the fluxes of NO_3^-, NH_4^+, and RP indicate that these components are sediment-derived.

Keywords: İzmir Bay, Homa Lagoon (Aegean Sea, Türkiye), Nutrient flux, Pore water, Sediment

İzmir Körfezi Homa Lagünü'nde Sediment ve Sediment Üstü Suda Nutrientler (Ege Denizi, Türkiye)

Özet: Biyojenik bileşiklerin bentik akışları, sığ su ekosistemlerinin biyojeokimyasında kritik bir rol oynar. Bentik akışların lagün ölçeğinde ölçülmesi, heterojen sediment yapıları ve sediment su arayüzündeki difüzyon ile advektif taşıma süreçlerinin birleşimi nedeniyle zordur. Bu çalışmada, İzmir Körfezi Homa Dalyan'ı (Ege Denizi, Türkiye) sedimentinde aylık nutrient akış denemeleri gerçekleştirilmiştir. Reaktif silikat (RSi) akışı 14.6 ile 255.24 µgatSi/m²/saat arasında, amonyum (NH_4^+) akışı 3.59 ile 95.8 µgatN/m²/saat arasında, nitrit (NO_2^-) akışı 0.93 ile 13.99 µgatN/m²/saat arasında, nitrat (NO_3^-) akışı 27.76 ile 300 µgatN/m²/saat arasında, reaktif fosfat (RP) akışı 0.74 ile 5.80 µgatP/m²/saat arasında değişim göstermiştir. Araştırma, RSi akışının yaz aylarında maksimum seviyeye ulaştığını, NO_3^- akışının kış ve yaz dönemlerinde hem sedimentten suya hem de sudan sedimente gerçekleştiğini ortaya koymuştur. NO_2^- akışının yaz dışında önemli ölçüde sedimente geçtiği belirlenmiştir. RP, yaz aylarında sedimentten suya geçerken, kış ortası ve sonbaharda sedimente bağlanmıştır. NH_4^+, kış aylarında hem sedimentten suya hem de sudan sedimente akış yaparken yaz aylarında sudan sedimente akış göstermektedir. Sonuç olarak NO_3^-, NH_4^+ ve RP'nin sedimentten su kolonuna doğru akışı, bu bileşenlerin sediment kökenli olduğunu göstermektedir.

Anahtar Kelimeler: İzmir Körfezi, Homa Dalyan'ı (Ege Denizi, Türkiye), Nutrient akışı, Pore suyu, Sediment

Article Info (Research)

1 Manisa Celal Bayar University, Faculty of Engineering and Natural Sciences, Biology Department, Manisa/Türkiye
2 Ege University, Faculty of Fisheries, Hydrobiology Department, İzmir/Türkiye
*** Corresponding author:**
ehyurur@gmail.com

Citation: Cingöz, E.E., Büyükışık, H. B., (2025), Nutrients in Sediment and Overlying Water in the Homa Lagoon of İzmir Bay (Aegean Sea, Turkey), Memba Water Science Journal, 11, (1) 10–23, https://doi.org/10.58626/memba.1667313

Submission Date: 12 January 2025
Acceptance Date: 12 March 2025
Publishing Date: 28 March 2025

1. Introduction

Over recent decades, the interplay of economic growth and rising population numbers, coupled with the rapid expansion of agriculture and industry in neighboring countries has resulted in increased discharges of fertilizers and wastewater into rivers and coastal waters. A significant portion of these nutrients ultimately deposits in the sediment layers (Grenz et al., 2010; Zhang et al., 2013; Jin et al., 2009; Lin et al., 2005). At the same time, inorganic nutrients (inorganic nitrogen, PO_4^{-3} and SiO_3 present in sediments can migrate into the overlying water, re-entering the euphotic zone, which is vital for sustaining nutrient balance and facilitating primary production in the water column. Increased nutrient loads can stimulate higher rates of algal growth and the accumulation of labile organic matter in surface sediments. Under these conditions, elevated mineralization rates can occur, modifying biogeochemical processes and promoting reactions that lead to the release of significant amounts of inorganic nutrients into the water column (Berelson et al., 1998). Therefore, sediments function not only as a major reservoir but also as a crucial source of nutrients, fulfilling an important environmental role (Zhang et al., 2013; Chelsky et al., 2016; Kim et al., 2016). Low concentrations or no oxygen at all in pore water and near-bottom water can result from currents not supplying sediment with enough dissolved oxygen to support degradation processes. This condition may result in the release of phosphates and ammonia stored in the sediment (Ingall & Jahnke, 1997; Duce et al., 2008). The release of phosphorus linked to reducible iron oxide phases contributes to increased benthic phosphorus fluxes (Ingall & Jahnke, 1997). Anoxic conditions obstruct denitrification which in turn diminishes the removal of nitrogen from aquatic systems in the form of dinitrogen gas (Jäntti & Hietanen, 2012). In oxygen-depleted environments, nitrate can undergo reduction to ammonium through the process of dissimilatory nitrate reduction (Jørgensen, 1989). This phenomenon exacerbates the 'vicious cycle' of eutrophication as nitrogen and phosphorus are recycled into bioavailable forms (Aller & Benninger, 1981; Conley et al., 2002; Diaz & Rosenberg, 2008). The flux of biogenic compounds from benthic environments is essential to understanding the biogeochemical processes occurring in shallow aquatic systems. Measuring these fluxes at the scale of a lagoon is complex particularly due to the heterogeneity of sediments and the interplay of diffusive and advective transport mechanisms at the sediment water interface. Nutrient fluxes from the benthic zone emerge from concentration gradients that exist between the pore waters in surface sediments and the water column. Increased export production and sediment degradation processes cause NH_4^+, PO_4^{-3}, and Si to build because of the discrepancy between the greater solute concentration in the top most sediment layer and the concentrations in the bottom water. Diffusive fluxes result from this (Aller & Benninger, 1981). Water advection can also be produced by benthic creature activity (Aller, 1982). The crucial role of species identity and their ecological characteristics in the cycling of biogeochemical elements and the exchange of solutes at the sediment-water interface which has important implications for the ecological dynamics of overexploited regions (Alvarez et al., 2018; Kendzierska et al., 2020;). The Homa Lagoon serves as a crucial ecological and economic resource and stands as the only active lagoon within the İzmir Bay, situated in the Eastern Aegean Sea. It is one of nine lagoons found in the Mediterranean region of Western Anatolia, with a total area of around 1.424 hectares and is one of five lagoons located in İzmir Bay, alongside Çakalburnu, Çalıbası, Kırdeniz and Ragıppasa. Among these lagoons, Homa Lagoon is distinguished by its extensive biodiversity and size. This lagoon features both freshwater and saltwater ecosystems, offering habitats for a variety of avian species, plant life, and aquatic organisms (Somay & Filiz, 2003). Its importance is amplified by its role in providing optimal nutrition, shelter and nesting sites for numerous species with differing habitat preferences. Furthermore, it regularly accommodates about 207 species of seabirds (Sıkı, 2002). The Homa Lagoon is impacted by the Gediz River, which suffers from significant pollution resulting from agricultural runoff, industrial effluents and domestic sewage originating from the surrounding region (Uluturhan et al., 2011). This lagoon does not receive direct inputs from wastewater treatment facilities; however, the agricultural practices along the Gediz River have a notable effect on the lagoon's ecosystem. Fertilizers utilized in farming are transported to the lagoon, especially during rainy periods, via the Gediz River's mouth, the sea and various non-point sources (Minareci et al., 2009). The combination of sedimentation from the Gediz River and inadequate freshwater inflow has led to severe shallowing of the Homa Lagoon. In 2014, the İzmir Metropolitan Municipality initiated projects aimed at deepening the lagoon and enhancing water circulation (Uluturhan et al., 2011). The aim of this study is to assess the contribution of sediment derived nutrients within the Homa Lagoon.

2. Materials and Methods

2.1. Description of Study Area

The research area under consideration is Homa Lagoon, located at coordinates 38° 33' 10"N and 26°49'50"E, approximately 25 kilometers to the northwest of the Gulf of İzmir, within the administrative boundaries of Menemen (Fig.1). This lagoon covers an area of 1.800 hectares and features a beach that stretches about 11 kilometers in length. The input of freshwater to the lagoon is significantly limited and there is a drainage canal that is occasionally opened to facilitate water flow into the lagoon. The lagoon's average depth is recorded at 0.75 meters during the wet season when water levels are elevated and 0.5 meters during the dry season when water levels recede. The lagoon is experiencing a rapid decrease in depth, attributed to seasonal fluctuations in the hydrological cycle and sediment runoff from adjacent agricultural areas (Sisman Aydın & Simsek, 2015; Tosunoglu et al., 2015).

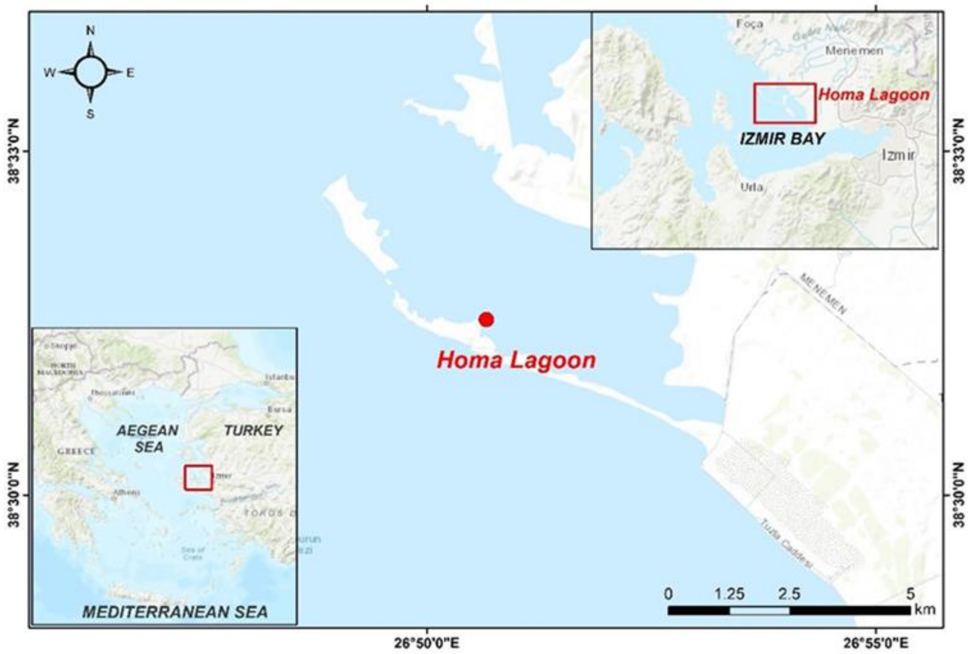


Figure 1. Topographic map of the Homa Lagoon and the location of water and sediment sampling

The lagoon is recognized as one of the ten most productive lagoons in the Eastern Aegean Sea and serves as the only active fish trap in İzmir Bay. It is located in close proximity to Turkey's most significant salt pan and the İzmir Bird Paradise. Both Homa Lagoon and Gediz Delta have been designated as protected wetlands under the RAMSAR and Bern Conventions, highlighting their importance in terms of biodiversity and conservation efforts (Ermert, 2003; Parlak et al., 2006). The Lagoon area is interconnected with İzmir Bay at various points through channels that facilitate water exchange. It consists of two sections: the Homa Lagoon (the main channel) and the Kırdeniz Lagoon (the smaller channel). The smaller channel has become shallower due to the sediment carried by the Gediz River, resulting in a loss of its channel characteristics. In the larger channel section selected for study, the maximum depth is 80 cm, while the average depth ranges from 40 to 45 cm. The Homa Lagoon is adversely affected by the organic and inorganic pollution loads from the Gediz River, which is one of the significant rivers in the Aegean Region, as well as from İzmir Bay.

2.2. Sampling and Analysis

Sediment samples were systematically collected on a monthly basis from a designated station at the lagoon's sea connected point, which constitutes the focal area of this study from January 2006 to December 2006. Notably, sampling was not feasible in March due to unfavorable weather conditions. The sediment was extracted using a gravity corer, featuring a core barrel with an inner diameter of 4.7 cm. The apparatus was secured at both ends, ensuring that the samples remained undisturbed during transportation to the laboratory, where they were kept upright in a plastic container at a temperature of 0°C. To make it easier to recover sediment layers, the core samples were divided into 4 cm slices in the lab using a tool that had a plastic bordered plate and a piston that was in line with the core pipe. A 10 ml injector, modified by cutting off its tip, was utilized to collect sediment samples of uniform volume by inserting it directly into the sediment matrix. The assessment of chlorophyll degradation products within the sediment was conducted in accordance with Lorenzen (1971).

2.2.1. Sediment Pore Water Parameters

A pore water extraction device was applied to the first 10 cm of sediment that was taken from a different core sample taken at the same spot. A two-layer GF/C filter paper was used to filter the sediment's pore water under pressure, producing a clean 50 ml liquid. The methods developed by Strickland & Parsons (1972) were used to assess the concentrations of nutrients (NH_4^+, NO_3^-, NO_2^-, PO_4^{-3} and SiO_2).

2.2.2. Water Column Parameters

At the sample station, measurements were made of temperature, salinity, dissolved oxygen, and the percentage of dissolved oxygen saturation. Salinity was calculated using Harvey's argentometric technique (Martin, 1972). The concentration of dissolved oxygen was measured using the Winkler method, while the saturation level of oxygen was calculated using the equations established by Benson & Krause (1984). This approach ensures accurate assessment of oxygen levels in the study area. Monthly water samples were collected vertically from the surface to the bottom at this station. Nutrient analyses were conducted in accordance with the procedures outlined by Strickland & Parsons (1972), utilizing the Hach Model DR-4000 spectrophotometer for accurate measurements.

2.2.3. Benthic Flux Incubations

In order to maintain consistent thermal conditions, the core samples were incubated at in situ bottom water temperatures. A hose attached to the core was used to introduce bottom water, which increased the volume of the sediment to 200 milliliters. The trials were conducted over a 24 hour period in order to minimize any potential variations in flux levels brought on by the oxygenation of the water covering the sand. Water samples taken from the surface using a siphon at the start and end of a day served as the basis for the measurements. To prevent upsetting the top layer of silt, the water above it was carefully blended. GF/C filter paper was used to filter about 100 milliliters of the resultant sample. A Milipore HA 0.45 µm filter paper was used for silicate analysis. Following the procedures outlined by Strickland & Parsons (1972), the amounts of ammonium, nitrite, nitrate, silicate and phosphate in the filtered samples which had been diluted with distilled water were examined. Aller & Benninger's (1981) equations served as the foundation for the computations.

$$J = V_t\,(C_t\; - \;C_0)\,/\,t.A$$

where J=nutrient flux rate (µmol/m^{2}day), A=core surface area, V_t=the water volume on the surface of the core at the time of t, C_t=the solute concentration at the time of t, C_0=the solute concentration at the onset of the experiment, t=sampling day (onset of the experiment is considered as 0).

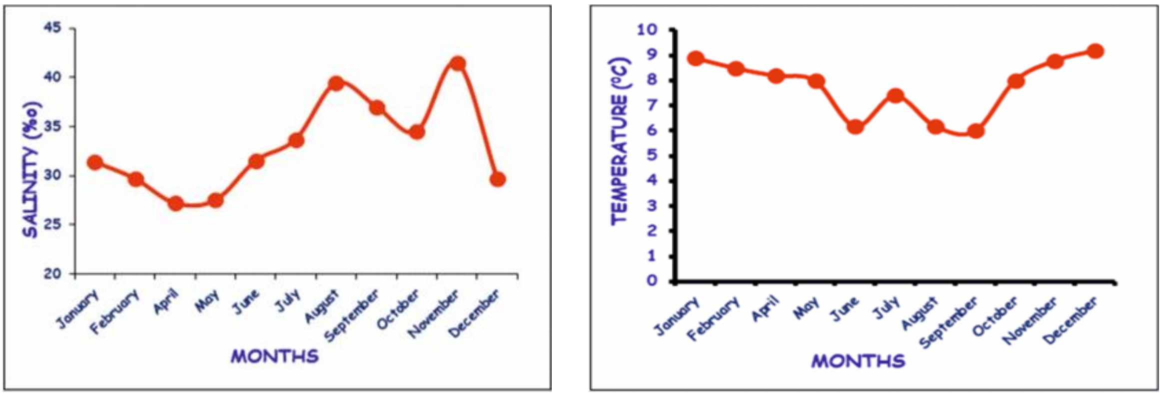
3. Results and Discussion

3.1. Sediment Solid Phase Parameters

The composition of sediments in cores collected at the depths of 4, 8, 12,16 and 20 cm of the sampling station. Sediment composition indicating that a single mode was identified at certain depths, but not at others. The minimum and maximum levels of the dry and wet mud densities were found to be 1.80–2.90 and 1.18–1.62 g/ml, respectively. The estimated porosity of the sediment is 74%. Since the pore water was obtained from the 10 cm core sample the diffusive fluxes were not able to be calculated however its direction was determined. The observation that the vertical distribution of organic carbon did not show variability in the core samples collected from different months. The monthly variations of organic carbon in the dynamic surface sediment ranged from 1.07-3.39 %.

3.2. The Spatio-Temporal Distribution of the Water Column Parameters

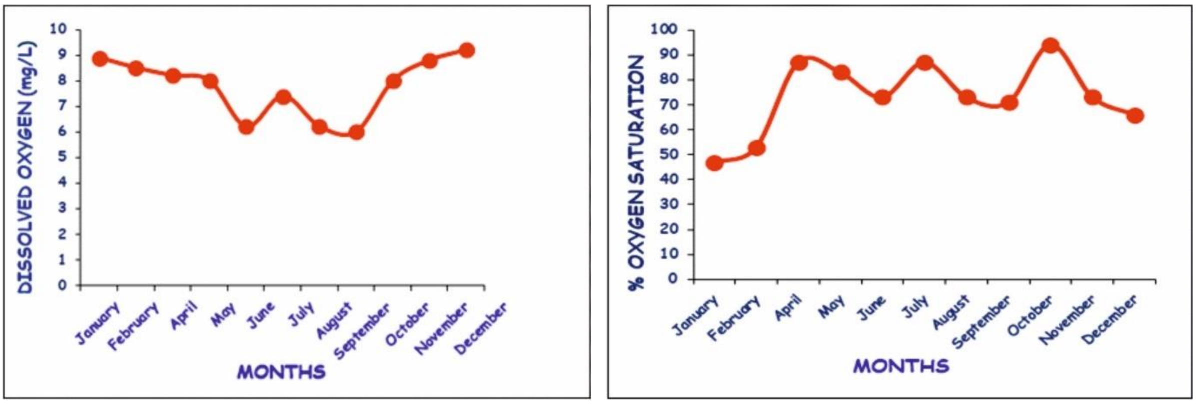
It is deduced from the water column's spatiotemporal salinity variations shows a significant increase in August and November (40‰) (Fig. 2a). This increase is attributed to evaporation resulting from the lack of water extraction from Tuzla's inlet. In April and May, rainfall causes the salinity to drop below 30‰. Additionally, there is a marked increase in temperature from June to October (Fig. 2b).



a b

Figure 2. The spatio-temporal distribution of salinity (a) and temperature (b) at the station at Homa Lagoon surface water

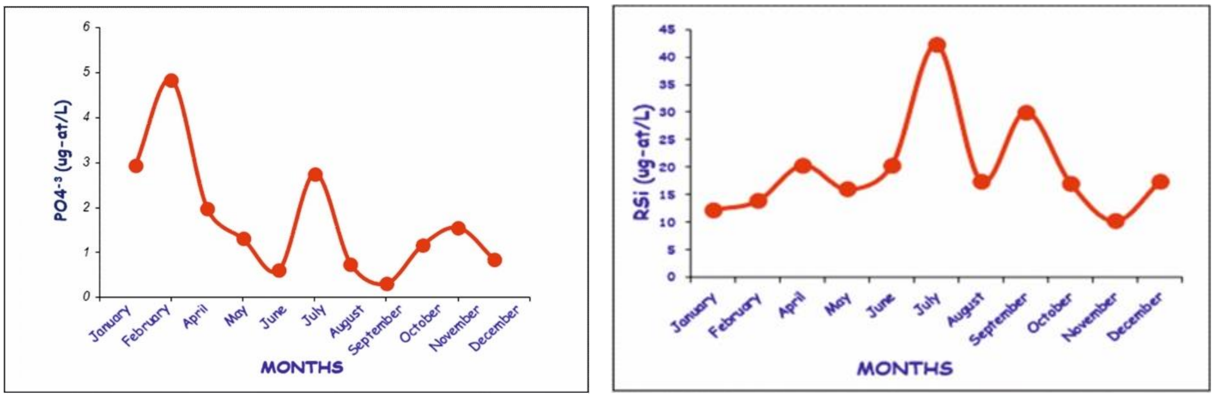
Salinity was correlated with the distribution of dissolved oxygen (DO) levels and dissolved oxygen saturation percentage. A significant increase in temperature which began in June, led to noticeable decreases in dissolved oxygen concentrations in August and September (Fig. 3 a,b). This situation indicates the presence of heterotrophic activity in the environment. Although saturation concentrations decreased with the rise in temperature, the reduction in saturation percentage must be related to the respiration of aquatic animals. The increase in temperature in July correlates with the decrease in saturation. In August and September, the maximum temperatures reached in surface waters also resulted in reduced oxygen solubility. Therefore, the saturation percentage should have approached 100%. However, the drop in saturation to levels between 60-70% may be attributed to the respiration of the developing fish population and other aquatic organisms within the environment (Kristensen & Kostka, 2005; Timmermann et al., 2006).



a b

Figure 3. The spatio-temporal distribution of dissolved oxygen concentration (a) and percent oxygen saturation (b) at Homa Lagoon surface water

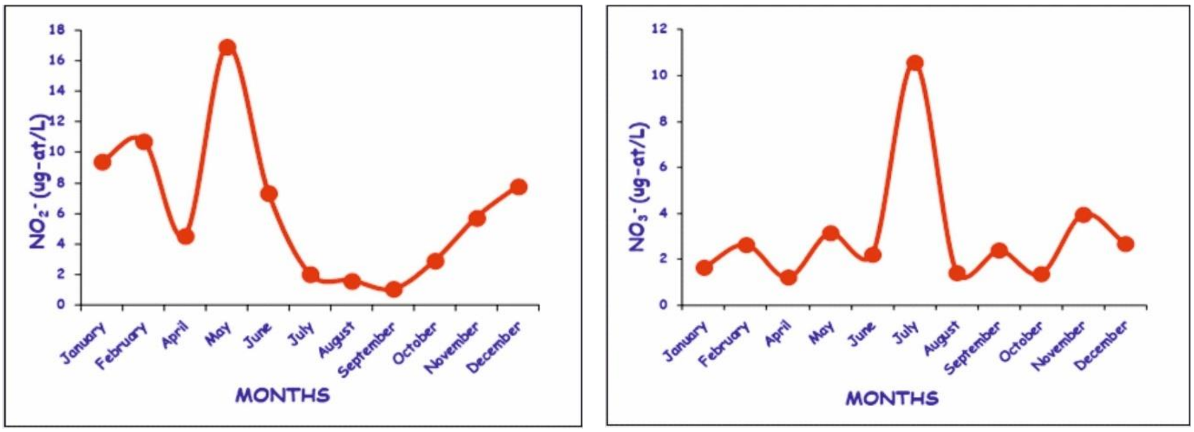
There is a maximum increase in reactive phosphorus (RP) in February and July (Fig. 4a). When examining the temporal variations of reactive silica (RSi), a maximum increase is observed in July. Additionally, there is a second maximum in silica in September (Fig. 4b).



a b

Figure 4. The spatio-temporal distribution of the phosphorus (a) and reactive silicate (b) concentrations at Homa Lagoon surface water

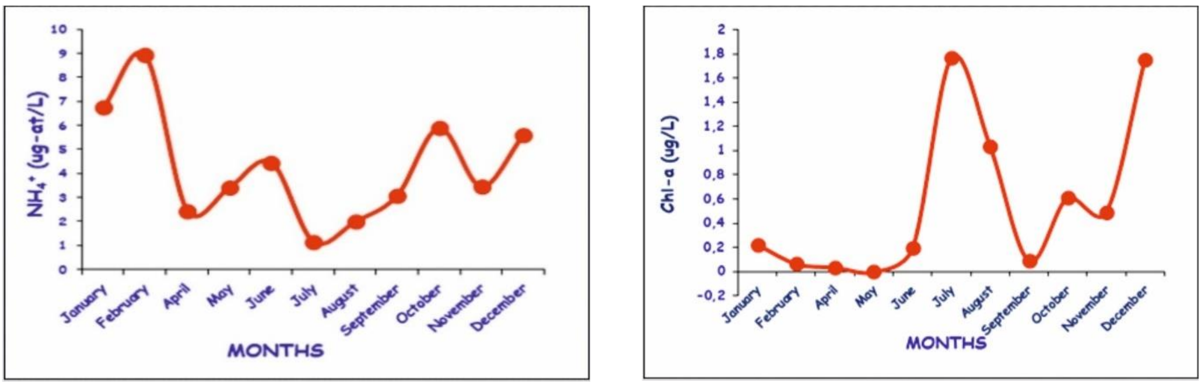
When examining the temporal changes of NH_4^+, NO_2^-, and NO_3^-, it is observed that during January, February and April, they exhibit parallel distributions at low temperatures. In May, a maximum of NO_2^- is observed, followed by a maximum of NO_3^- in July (Fig. 5a,b). In the autumn months, the increase in NH_4^+ is followed by increases in NO_2^- and NO_3^- (Fig. 6a). This indicates that nitrification continues in the shallow lagoon water as well.



a b

Figure 5. The spatio-temporal distribution of nitrite (a) and nitrate (b) concentrations at Homa Lagoon surface water

The maximum increases in Chlorophyll-a observed in July and December correspond with high concentrations of PO_4^{-3}, RSi and NO_3^- in July, reflecting the impact of nutrients on primary production (Fig. 6b).



a b

Figure 6. The spatio-temporal distribution of ammonium (a) concentrations and Chlorophyll-a concentrations (b) at Homa Lagoon surface water

In February, a decrease of 0.2 in pH was observed and in November, a significant decrease of 0.9 in pH was noted indicating respiratory activities (Fig. 7). Indeed, the reduced Chlorophyll-a concentrations in November also suggest heterotrophic activity.

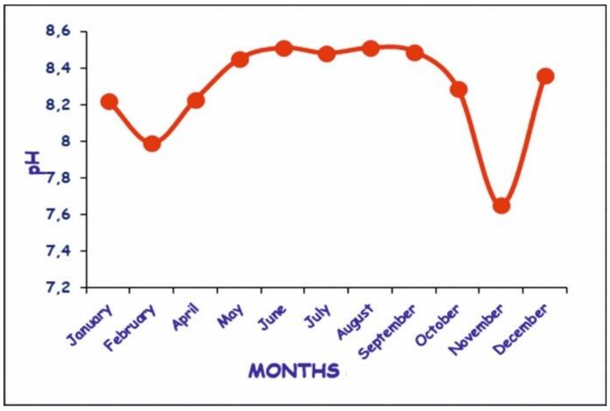
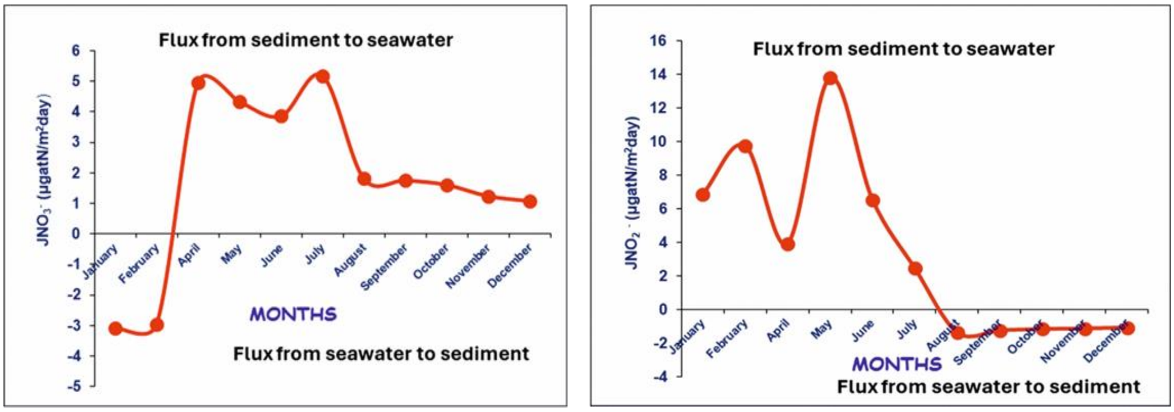


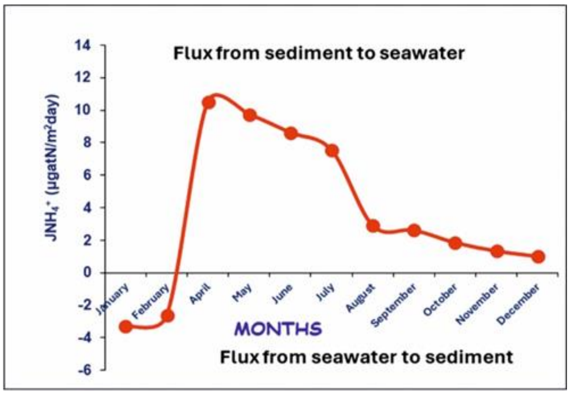
Figure 7. The spatio-temporal distribution of pH at Homa Lagoon surface water

3.3. Nutrient Fluxes in Sediment

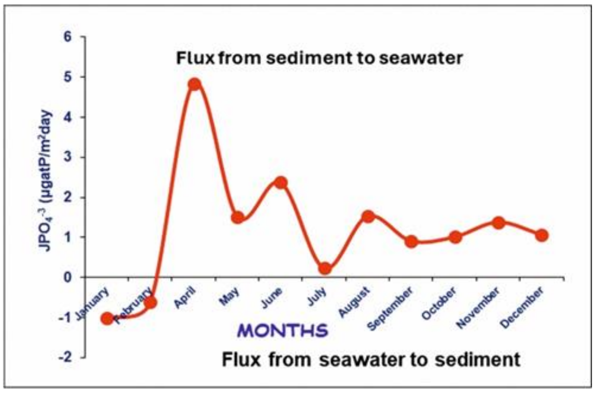
Temporal changes in nutrient flux are illustrated in Fig. 8. Nitrate fluxes from water to sediment were recorded at 3.07 $\mu gNm^{-2}day^{-1}$ in January and 2.96 $\mu gNm^{-2}day^{-1}$ in February, respectively (Fig. 8a). Nitrate flux from sediment to water persisted from April to December, exhibiting minimum and maximum values of 1.08 $\mu gNm^{-2}day^{-1}$ in December and 5.19 $\mu gNm^{-2}day^{-1}$ in July, respectively. Nitrite flux from sediment to water was observed from January to July. The maximum and minimum values were recorded in July and May, with measurements of 13.8 and 2.44 $\mu gNm^{-2}day^{-1}$, respectively (Fig. 8b). Nitrite fluxes from water to sediment were measured at 1.38 $\mu gNm^{-2}day^{-1}$ in August and 1.08 $\mu gNm^{-2}day^{-1}$ in December. Ammonium flux from water to sediment was recorded at 3.27 $\mu gNm^{-2}day^{-1}$ in January and 2.62 $\mu gNm^{-2}day^{-1}$ in February (Fig. 8c). From April to December, a flux of ammonium from sediment to water was observed. The flux from sediment to water showed maximum and minimum values of 10.53 $\mu gNm^{-2}day^{-1}$ in April and 1.02 $\mu gNm^{-2}day^{-1}$ in December, respectively. In January, the RP flux from water to sediment was measured at 1.01 $\mu gatPm^{-2}day^{-1}$ while in February, it was 0.62 $\mu gatPm^{-2}day^{-1}$ (Fig. 8d). The flux from sediment to water ranged from 4.84 $\mu gatPm^{-2}day^{-1}$ in April to 0.23 $\mu gatPm^{-2}day^{-1}$ in July. The silicate flux was observed in January and February, showing values of 0.8 $\mu gatSim^{-2}day^{-1}$ and 1.82 $\mu gatSim^{-2}day^{-1}$, respectively for the water to sediment direction (Fig. 8e). The silicate flux from sediment to water continued from April through December with maximum and minimum values recorded in April (10.4 $\mu gatSim^{-2}day^{-1}$) and July (3.7 $\mu gatSim^{-2}day^{-1}$), respectively.



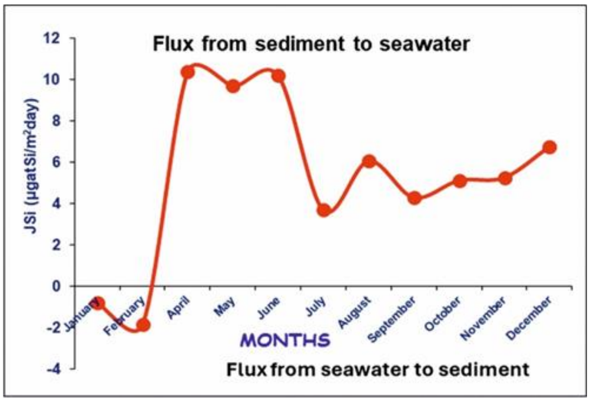
a b



c



d

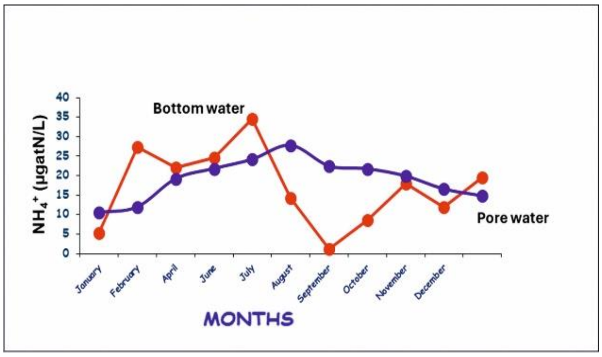


e

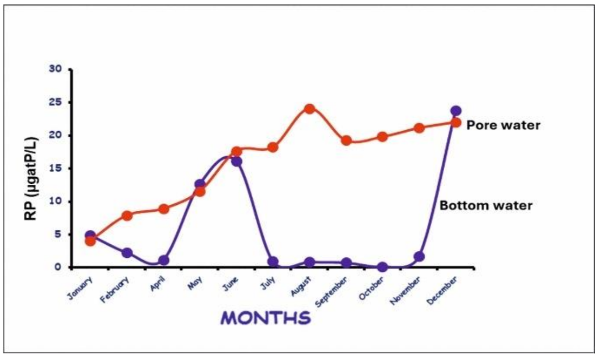
Figure 8. The monthly variation of NO_3^- (a), NO_2^-(b), NH_4^+(c), PO_4^{-3}(d) and Si (e) obtained from core incubations at Homa Lagoon surface sediment

3.4. Solute-Temporal Variations in Bottom and Pore Waters

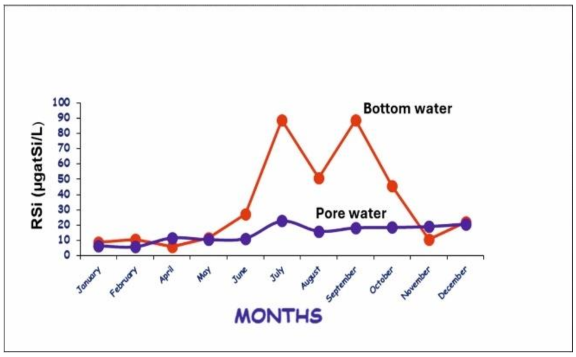
Between January and July, the concentrations of ammonium in pore water varied, reaching a minimum of 10.52 µgatN/l and a maximum of 27.74 µgatN/l. The ammonium concentrations in bottom water varied more than those in pore water, ranging from 19.2 to 48.7 µgatN/l between January and July, respectively (Fig. 9a). Bottom water nitrate concentrations fluctuated from 6.27 µgatN/l in January to 12.05 µgatN/l in December. Notably, the minimum concentration was recorded at 4.78 µgatN/l in December, while the maximum reached 13.21 µgatN/l in June (Fig. 9d). Pore water nitrite concentrations ranged from 5.75 µgatN/l in January to 32.93 µgatN/l in December. In contrast, bottom water nitrite concentrations varied from 7.32 µgatN/l in January to 31.87 µg atN/l in December (Fig. 9e). Bottom water reactive phosphate (RP) concentrations ranged from 4.94 µgatP/l in January to 23.78 µgatP/l in December. Meanwhile, pore water RP concentrations increased from a minimum of 4.02 µgatP/l in January to a maximum of 24.13 µgatP/l in August. Notably, there was a consistent rise in RP concentrations from January to August (Fig. 9b). Pore water silicate concentrations varied from a minimum of 5.92 µgatSi/l in February to a maximum of 22.83 µgatSi/l in July. In comparison, bottom water silicate values ranged from 6.12 µgatSi/l in February to 22.16 µgatSi/l in December (Fig. 9c). Between January and December, silicate concentrations exhibited a general upward trend in both bottom water and pore water.

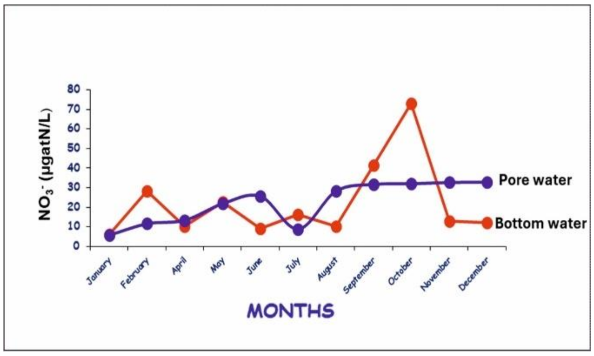


a

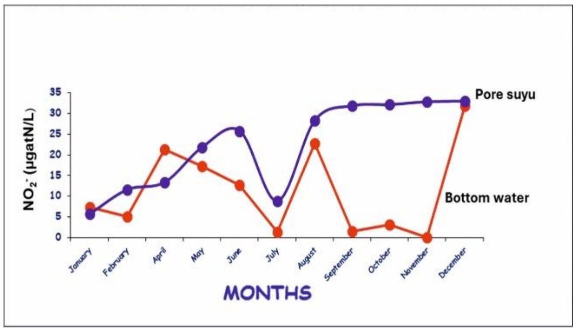


b





c d



e

Figure 9. The monthly variations of bottom water NH_4^+ (a), RP (b), RSi (c), NO_3^- (d) and NO_2^-(e) concentrations at Homa Lagoon surface sediment

The concentrations of bottom water NH_4^+ show high variability from late winter until June, rapidly decreasing during the summer months and remaining below pore water levels until December. The significant decline in bottom water NH_4^+ concentrations in July and August can be explained by the relative increase in phytoplankton uptake, as evidenced by elevated Chlorophyll-a concentrations. In other words, the formation of bottom water NH_4^+ concentrations below pore water levels can be attributed to microalgal uptake. The flow of NH_4^+ from water to sediment in January is contrary to diffusive flow. This flow can be explained by benthic microalgal uptake (BMA) and/or the adsorption to clay minerals. The sediment to water flow in February, contrary to diffusive flow can be explained by the degradation of a past algal bloom. In June, both the diffusive flow and incubation flow are in the same direction, indicating both diffusion and absorption by clay minerals. The incubation flow obtained in September, consistent with diffusion represents diffusive flow. The concentrations of bottom water and pore water in October, which are nearly identical, indicate the absence of diffusive flow; however, it can be explained by the benthic microalgae uptake (BMA) and the involvement of clay minerals. In November, pore waters contain higher NH_4^+ concentrations than bottom waters, and the flow obtained from core incubation is identical to the diffusive flow. In addition to diffusive flow. Degradation of the sediment surface layer may potentially contribute NH_4^+ to the water column. The concentrations of NH_4^+ in surface water are noteworthy, exceeding 80 µgatN/l levels in May and June. Salinity is at its lowest values indicating that the outflow of water is associated with NH_4^+ from non point sources during rainfall. This concentration rapidly decreases from May reaching zero by September. This decline is influenced by the influx of water into the bay in June and the contributions of chlorophyll maximum and nitrification (the nitrification of NH_4^+ to NO_2^-) in July. The turbidity caused by frequent mixing of the water column due to winds should also have an effect on the continuation of nitrification. This process proves to be particularly effective in shallow water environments (Fagherazzi et al., 2007, 2013; Ganju et al., 2005, 2013; Carniello et al., 2005). The ammonium (NH_4^+) fluxes, which can reach a maximum of 7000 µgN/m²/day, align with the low flux values documented for salt marshes (Barbanti et al., 1995; Essonni, 1998; Hyacinthe et al., 2001). Furthermore, these fluxes are lower than those reported by Ozkan et al. (2008) for İzmir Bay and by Bonometto et al. (2019) for salt marshes, yet they are comparable to the values presented by Mna et al. (2022). The presence of negative values in NH_4^+ flux suggests that the nitrification of ammonium in the upper sediment layer is a critical process in wetland ecosystems. In terms of nitrate (NO_3^-) concentrations, both bottom water and pore water exhibit no diffusive fluxes in January, while incubation fluxes are significantly high and directed from sediment to water. This phenomenon can be attributed to the nitrification process converting nitrite (NO_2^-) to nitrate (NO_3^-). Notably, the influx of NO_2^- into the sediment is also considerably elevated. From April to June, diffusive fluxes occur from sediment to water. Conversely, incubation fluxes are directed from water to sediment and tend to be higher during the summer months. It can be concluded that the nitrification-denitrification process functions optimally in June. The concentrations of nitrate (NO_3^-) in the bottom water showed small increases in February, May and July, followed by a significant increase in September and October. The pore water NO_3^- values exhibited an upward trend throughout the year, except for July. In September and October, the diffusive flow is directed from water to sediment. In September, the incubation flow is reversed, indicating the dominance of nitrification. In October, the flow is aligned with the diffusive direction, suggesting both diffusion and nitrification-denitrification processes. Indeed, the high concentrations of nitrite (NO_2^-) in pore water from late summer to December indicate that nitrification is progressing. The surface water concentrations of NO_3^- exhibit two notable maxima: one during the months of April, May and June and another in December. The high values observed in April and May are attributed to terrestrial inputs while the peak in June is influenced by nitrification. These concentrations decline in July coinciding with an increase in Chlorophyll-a. The increase in December can be explained by inputs from both sediment and terrestrial sources due to rainfall. The estimated fluxes indicate that nitrates are produced below the sediment-water interface. This finding aligns with the processes of NO_3^- production or consumption occurring in sediments, which globally results in NO_3^- production in oxic sediments and consumption in less oxygenated sediments (Belias et al., 2007; Rigaud et al., 2013). In January, the fluxes of NO_3^- can reach a maximum value of 40.000 µgN/m²/day which is higher than the value reported by Mna et al. (2022). At the sampling station, which has a sediment structure similar to that of a saline marsh channel (%27 coarse material), the NO_3^- flux in September was close to the value reported by Murray et al. (2006). However, the maximum value of 40.000 µgN/m²/day in January is ten times higher, indicating that nitrification is significantly more effective in the Ancao basin. Pore water NO_2^- concentrations are generally high in bottom waters, except in April. Diffusive flows should be directed from the sediment to the water. However, all incubation flows are directed from the water to the sediment, suggesting that this nutrient is significantly converted to NO_3^- through nitrification in the sediment. There has been a significant decrease in bottom water values in July, which is also reflected in pore water. During this month, characterized by high temperatures, it may indicate nitrification along with denitrification and/or increased Chlorophyll-a concentrations. While pore water values continue to rise from August onwards, bottom water values approach zero from September to November. In December, there is again an increase in bottom water concentrations, approaching those of pore water. The surface water concentrations of NO_2^- remain below 10 µgatN/l throughout the year with the maximum value detected in May indicating the presence of NO_2^- from terrestrial regions due to rainfall. Bottom water RP concentrations show an increase in May, June and December while they are low in other months. Pore water concentrations rise from 4.02 µgatRP/l in January to 24.13 µgatRP/l in August. The sediment of Homa Lagoon has been found to be an effective phosphate trap. The values in the water continuously contribute to pore water. Incubation experiments indicate that, despite the RP flows not being purely diffusive there is evidence of benthic microalgae uptake (BMA) and potential incorporation into the sediment through binding with FeOOH (ferrihydrite) from January onwards (Blomqvist et al., 2004). From February to June, the incubation flows are consistent with diffusive flows, suggesting that the diffusive flow can explain the incubation flow. Additionally, the decomposition of benthic detritus may also contribute to the sediment. In September, the diffusive flow is directed from the sediment to the water. However, the incubation flow is directed from the water to the sediment and can be explained by benthic microalgae uptake (BMA) or binding to FeOOH. Ferric oxyhydroxides found in sediments interact with phosphate (PO_4^{-3}). Under anaerobic conditions, these compounds release dissolved iron (Fe^{+2}), causing phosphate to enter the water. When Fe^{+2} comes into contact with oxygenated water, it precipitates again as ferric oxyhydroxide (FeOOH) and adheres to particulate matter, dispersing between sediment and overlying water. This process plays an important role in nutrient cycles and sediment chemistry (Aller & Benninger, 1981). In October, the reverse direction of the incubation flow with diffusive flow may also be related to benthic microalgae uptake (BMA) and binding to FeOOH, similar to September. The concentrations of RP in surface water remain low throughout the year (< 2.8 µgatRP/l), with increases in February attributed to flow from the sediment. It can also be stated that the increases observed in December, May and June originate from the sediment. The concentrations of RP in surface water are transported to the sediment through adsorption onto FeOOH and clay minerals. The consistently low levels detected throughout the year support this. The low flow rates of the RP are consistent with the sediment particle structure provided by Moraes et al. (2023), which contains 27-30% clay and the measured values are three times lower for saline wetlands. In this process, the coastal structure of the particulate material may also be significant. The concentrations of reactive silica (RSi) in the bottom water are nearly identical to the pore water concentrations from late autumn to early spring (including winter months). Starting from June and continuing until late autumn, two significant increases in bottom water RSi concentrations are evident (in July and September). In contrast, pore waters have shown relatively smaller variations, ranging from 5.92 µgatSi/l to 22.83 µgatSi/l. Since the RSi in pore water is influenced solely by factors affecting the solubility of particulate silica, it is noteworthy that the frequent decrease in pore waters obtained from sediments at the end of incubation experiments indicates that the very high silica concentrations in bottom water (for example, in September) suggest that RSi flow has significantly completed, and therefore, the experimental flow is quite low. The relatively lower bottom water value in August compared to July and September indicates that the RSi flow has not yet been completed. Indeed, RSi flow reaches its highest value. In silicate flows, the core incubation flow in January is the same as the diffusive flow. Except for November, the diffusive flow is directed from water to sediment during all months, while core incubations are directed from sediment to water from February to November. The disintegration of diatom skeletons in the sediments top layer is what causes the contrast with diffusive flow. However, the diffusive flow is redirected from sediment to water in November. Conversely, the incubation flow is directed from water to sediment, reflecting benthic microalgae uptake. In surface waters, RP and RSi concentrations are higher and are consistent with the increases observed in bottom waters during June, July and October. The relatively high RSi values in surface waters may be attributed to water entering from the İzmir Bay, the RSi flow from sediment and terrestrial sources. At the same time, nutrients released from the oxidation of organic matter at the sediment-water interface can provide a substantial fraction (5-22%) of the nitrogen and phosphate requirements essential for primary production in lagoons. This underscores the critical role of sedimentary nutrient dynamics in supporting the productivity of these aquatic ecosystems. (Lourey et al., 2001; Denis and Grenz, 2003, Kim et al., 2020). During core incubations, phosphate and silicate fluxes were oriented towards the sediment. This could be related to the photosynthetic processes of the microphytobenthos at the sediment water interface. The values given for Si fluxes are consistent with those reported by Ozkan et al. (2008) and Charbonnier et al. (2023). However, the value of 10.000 µgatSi/m²/day reported by Aller and Benninger (1981) for 20 °C is smaller than the measured value of 35.000 µgatSi/m²/day at 28 °C in August. Recent research advocates for the adoption of ecosystem based strategies aimed at minimizing both external and internal nutrient inflows, thereby fostering sustainable biogeochemical processes. Such strategies may encompass aeration, modifications to hydrological patterns (enhanced circulation), the removal of degraded sediments (through dredging) and focused restoration of habitats and shorelines (Harris et al., 2015; Fox & Trefry, 2018; Fox & Trefry, 2023; Ma et al., 2022a,b). These globally employed techniques also present themselves as a promising approach for the Homa Lagoon (İzmir Bay), which is characterized by elevated nutrient levels.

4. Conclusion

Nutrient flux across the sediment-water interface has been assessed for Homa Lagoon with the following findings:

1. The nutrients evaluated were ammonium (NH_4^+), nitrate (NO_3^-) + nitrite (NO_2^-), reactive phosphate (RP) and reactive silicate (RSi). The flux of these nutrients exhibited significant variability across different months.
2. The flux values recorded were NH_4^+ ranging from 3.59 to 95.8 µgatN/m²/hour, NO_2^- from 27.76 to 300 µgatN/m²/hour, RP from 0.74 to 5.80 µgatP/m²/hour, and RSi from 14.6 to 255.24 µgatSi/m²/hour. These values reflect the general observation that NH_4^+ was released from sediments while NO_2^- was taken up by the water, although the net flux of dissolved inorganic nitrogen was out of the sediments. In contrast, RP tended towards a steady-state condition. RSi flux generally occurred from sediment to water. Overall, the flux determined in this study was at the high end of the range when compared to other studies. Additionally, it was identified that nitrification and denitrification processes are significant for Homa Lagoon.
3. Ecosystem-based approaches necessitate the reduction of both external and internal nutrient loads to promote biogeochemical processes in Homa Lagoon.

5. Compliance with Ethical Standard

a) Author Contributions

1. E.E.Y.: Conceptualization, process, software, verification, formal analysis, research, materials, composing the first draft, composing the review, and editing,
2. H.B.B.: Conceptualization, process, software, verification, formal analysis, inquiry, materials, data curation, authoring the first draft, reviewing and revising it, visualization, and oversight. The published version of the manuscript has been read and approved by both authors.

b) Conflict of Interests

There is no conflict of interest, according to the authors.

c) Statement on the Welfare of Animals

Not relevant

d) Statement of Human Rights

There are no human subjects in this study.

e) Funding

This study was supported by Manisa Celal Bayar University Scientific Research Project Coordination (BAP) (Project No. FEF-064). This study contains a part of PhD dissertation prepared by Emine Erdem Yürür in Manisa Celal Bayar University.

6. References

Aller, R. C. (1982). The effects of macrobenthos on chemical properties of marine sediment and overlying water. In: McCall, P.L., Tevesz, M.J.S. (Eds.), Animal-Sediment Relations. Plenum. New York, pp. 53–102.

Aller, R. C., & Benninger, L. K. (1981). Spatial and temporal patterns of dissolved ammonium, manganese and silica fluxes from bottom sediments of Long Island Sound, USA. Journal of Marine Research, 39(2): 295–314.

Alvarez, M. F., Bazterrica, M. C., Fanjul, E., Addino, M. S., Valiñas, M. S., Iribarne, O. O. & Botto, F. (2018). Effects of two estuarine intertidal polychaetes on infaunal assemblages and organic matter under contrasting crab bioturbation activity. Journal of Sea Research, 139: 33–40. https://doi.org/10.1016/j.seares.2018.06.008

Barbanti, A., Bergamini, M. C., Frascari, F., Miserocchi, S., Ratta, M. & Rosso, G. (1995). Diagenetic processes and nutrient fluxes at the sediment-water interface, Northern Adriatic Sea. Italy. Marine and Freshwater Research, 46: 55–67 https://doi.org/10.1071/MF9950055

Belias, C., Dassenakis, M. & Scoullos, M. (2007). Study of the N, P and Si fluxes between fish farm sediment and seawater. Results of simulation experiments employing a benthic chamber under various redox conditions. Marine Chemistry, 103 (3-4): 266-275 https://doi.org/10.1016/j.marchem.2006.09.005

Benson, B. B. & Krause Jr, D. (1984). The concentration and isotopic fractionation of oxygen dissolved in fresh water and sea water in equilibrium with the atmosphere. Limnology and Oceanography, 29 (3): 620–632 https://doi.org/10.4319/lo.1984.29.3.0620

Berelson, W. M., Heggie, D., Longmore, A., Kilgore, T., Nicholson, G. & Skyring, G. (1998). Benthic Nutrient Recycling in Port Phillip Bay, Australia. Estuarine, Coastal and Shelf Science, 46: 917-934 https://doi.org/10.1006/ecss.1998.0328

Blomqvist, S., Gunnars, A. & Elmgren, R. (2004). Why the limiting nutrient differs between temperate coastal seas and freshwater lakes: A matter of salt. Limnology and Oceanography, 49 (6): 2236–2241 https://doi.org/10.4319/lo.2004.49.6.2236

Bonometto, A., Feola, A., Rampazzo, F., Gion, C., Berto, D., Ponis, E. & Boscolo Brusà, R. (2019). Factors controlling sediment and nutrient fluxes in a small microtidal salt marsh within the Venice Lagoon. Science of the Total Environment, 650: 1832–1845. https://doi.org/10.1016/j.scitotenv.2018.09.142

Carniello, L. A. Defina, Fagherazzi, S. & D'Alpaos, L. (2005). A combined Wind Wave-Tidal Model for the Venice lagoon, Italy. Journal of Geophysical Research., 110, F04007. doi:10.1029/2008JF001157

Charbonnier, C., Anschutz, P., Tamborski, J. & van Beek, P. (2023). Benthic fluxes and mineralization processes at the scale of a coastal lagoon: Permeable versus fine-grained sediment contribution. Marine Chemistry, 254: 1-14. https://doi.org/10.1016/j.marchem.2023.104274

Chelsky, A., Pitt, K. A., Ferguson, A. J. P., Bennett, W. W., Teasdale, P. R. & Welsh, D. T. (2016). Decomposition of jellyfish carrion in situ: short-term impacts on infauna, benthic nutrient fluxes and sediment redox conditions. Science of The Total Environment, 566 (567): 929–937. https://doi.org/10.1016/j.scitotenv.2016.05.011

Conley, D. J., Humborg, C., Rahm, L., Savchuk, O. P. & Wulff, F. (2002). Hypoxia in the Baltic Sea and basin-scale changes in phosphorus biogeochemistry. Environmental Science & Technology, 36: 5315–5320. https://doi.org/10.1021/es025763w.

Denis, L. & Grenz, C. (2003). Spatial variability in oxygen and nutrient fluxes at the sediment-water interface on the continental shelf in the Gulf of lions (NW Mediterranean). Acta Oceanologica, 26, 373–389. https://doi.org/10.1016/S0399-1784(03)00017-3

Diaz, R. J. & Rosenberg, R. (2008). Spreading dead zones and consequences for marine ecosystems. Science, 321: 926–929. https://doi.org/10.1126/science.1156401

Duce, R. A., LaRoche, J., Altier, K., Arrigo, K. R., Baker, A. R., Capone, D. G., Cornell, S., Dentener, F., Galloway, J., Ganeshram, R. S., Geider, R. J., Jickells, T., Kuypers, M. M., Langlois, R., Liss, P. S., Liu, S. M., Middelburg, J. J., Moore, C. M., Nickovic, S., Oschlies, A., Pedersen, T., Prospero, J., Schlitzer, R., Seitzinger, S., Sorensen, L. L., Uematsu, M., Ulloa, O., Voss, M., Ward B. & Zamora, L. (2008). Impacts of atmospheric anthropogenic nitrogen on the open ocean. Science, 320: 893–897. https://doi.org/ 10.1126/science.1150369

Ermert, A. (2003). Turkey goes to court to protect Gediz Delta. http:/ /www. panda. org/ about_ wwf/ what_we _do/freshwater /news /news. cfm? uNews ID=5322 9.Jan.2003

Essoni, N. (1998). Etude de la dynamique des sels nutritifs et des métaux lourds en Relation avec la sédimentologie et l'hydrodynamique dans le large du golfe de Tunis. Ph.D. Thesis. Université De Tunis II, Tunus.

Fagherazzi, S., Palermo, C., Rulli, M. C., Carniello, L. & Defina, A. (2007). Wind waves in shallow microtidal basins and the dynamic equilibrium of tidal flats. Journal of Geophysical Research, 112: 1-12. doi:10.1029/2006JF000572

Fagherazzi, S., Wiberg, P. L., Temmerman, S., Struyf, E., Zhao, Y. & Raymond, P. A. (2013). Fluxes of water, sediments, and biogeochemical compounds in salt marshes. Ecological Processes, 2(3): 1-16. doi:10.1186/2192-1709-2-3.

Fox, A. L, & Trefry, J. H. (2018). Environmental dredging to remove fine-grained, organic-rich sediments and reduce inputs of nitrogen and phosphorus to a subtropical estuary. Marine Technology Society Journal, 52: 42–57. doi: 10.4031/mtsj.52.4.3.

Fox, A. L. & Trefry, J. H. (2023). Nutrient fluxes from recent deposits of fine-grained, organic-rich sediments in Florida estuary. Frontiers in Marine Science, 10: 1-21. https://doi.org/10.3389/fmars.2023.1305990

Ganju, N. K., Schoellhamer, D. H. & Bergamaschi, B. A. (2005). Suspended sediment fluxes in a tidal wetland: measurement, controlling factors, anderror analysis. Estuaries, 28: 812–822. doi:10.1007/BF02696011

Ganju, N. K., Nidzieko, N. J. & Kirwan M. L. (2013). Inferring tidal wetland stability from channel sediment fluxes: Observations and a conceptual model. Journal of Geophysical Research: Earth Surface, 118: 2045–2058. https://doi.org/10.1002/jgrf.20143

Grenz, C., Denis, L., Pringault, O. & Fichez, R. (2010). Spatial and seasonal variability of sediment oxygen consumption and nutrient fluxes at the sediment water interface in a sub-tropical lagoon (New Caledonia). Marine Pollution Bulletin, 61: 399–412. https://doi.org/10.1016/j.marpolbul.2010.06.014

Harris, L. A., Hodgkins, C. L. S., Day, M. C., Austin, D., Testa, J. M., Boynton, W., Van Der Tak, L. & Chen, N. W. (2015). Optimizing recovery of eutrophic estuaries: Impact of destratification and re-aeration on nutrient and dissolved oxygen dynamics. Ecological Engineering, 75: 470–483. doi: 10.1016/j.ecoleng.2014.028

Hyacinthe, C., Anschutz, P., Carbonel, P., Jauanneau, J. M. & Jorrisen, F. J. (2001). Early diagenetic processes in the muddy sediments of the bay of Biscay. Marine Geology, 177:111-128. https://doi.org/10.1016/S0025-3227(01)00127-X

Ingall, E. & Jahnke, R. (1997). Influence of water-column anoxia on the elemental fractionation of carbon and phosphorus during sediment diagenesis. Marine Geology. 139 (1–4): 219–229. https://doi.org/10.1016/S0025-3227(96)00112-0.

Jin, H. Y., Chen, J. F., Weng, H. X., Li, H. L., Zhang, W. Y., Xu, J., Bai, Y. C. & Wang, K. (2009). Variations of paleoproductivity in the past decades and the environmental implications in the Changjiang Estuary in China. Acta Oceanologica Sinica, 31 (2): 113–119 (in Chinese with English abstract). DOI10.1007/s13131-010-0035-x

Jäntti, H. & Hietanen, S. (2012). The effects of hypoxia on sediment nitrogen cycling in the Baltic Sea. Ambio, 41 (2): 161–169. https://doi.org/10.1007/s13280-011-0233-6.

Jørgensen, K. S. (1989). Annual pattern of denitrification and nitrate ammonification in estuarine sediment. Applied and Environmental Microbiology, 55:1841–1847. https://doi.org/10.1128/aem.55.7.1841-1847.1989

Kendzierska, H., Lukawska-Matuszewska, K., Burska, D. & Janas, U. (2020). Benthic fluxes of oxygen and nutrients under the influence of macrobenthic fauna on the periphery of the intermittently hypoxic zone in the Baltic Sea. Journal of Experimental Marine Biology and Ecology, 530–531. https://doi.org/10.1016/j.jembe.2020.151439

Kim, S., Choi, A., Yang, E., Lee, S., & Hyun, J. (2016). Low benthic respiration and nutrient flux at the highly productive Amundsen Sea Polynya, Antarctica. Deep Sea Research, Part II 123: 92–101. https://doi.org/10.1016/j.dsr2.2015.10.004

Kim, S.H., Lee, J.S., Kim, K.T., Kim, S.L., Yu, O.H., Lim, D & Kim, S.H. (2020). Low benthic mineralization and nutrient flux in the continental shelf sedimenr of the northern East China Sea .Journal of Sea Research, 164, 101934. https://doi.org/10.1016/j.seares.2020.101934

Kristensen, E. & Kostka, J.E. (2005). Macrofaunal burrows and irrigation in marine sediment:

Microbiological and biogeochemical interactions. In: Kristensen, E., Haese, R.R. & Kostka, J.E. (Eds.), Interaction between Marcro- and Microorganisms in Marine Sediments. American Geophysical Union, Washington, DC, pp. 125–157. https://doi.org/10.1029/CE060p0125

Lin, C., Ning, X., Su, J., Lin, Y. & Xu, B. (2005). Environmental changes and the responses of the ecosystems of the Yellow Sea during 1976–2000. Journal of Marine Systems, 55: 223–234. https://doi.org/10.1016/j.jmarsys.2004.08.001

Lorenzen, C. J. (1971). Chlorophyll-Degradation products in sediments of Black Sea. WoodsHole Oceanographic Institution Contribution No.2828, p.426–428.

Lourey, M.J., Alongi, D.M., Ryan, D.A.J. & Devlin, M.J. (2001). Variability of nutrient regeneration rates and nutrient concentrations in surface sediments of the northern Great Barrier Reef shelf. Continental Shelf Research. 21, 145–155. https://doi.org/10.1016/S0278-4343(00)00084-4

Ma, S., Yu, Q., Chen, G., Su, H., Tang, W., Sun, Y., Zhou, Z., Jiang, L., Zhu, J., Chen, L., Zhu, B. & Fan, J. (2022a). Above ground net primary productivity mediates the responses of soil respiration to nutrient additions in two tropical montane rainforests. Agricultural and Forest Meteorology, 327, 109200 https://doi.org/10.1016/j.agrformet.2022.109200

Ma, S., Chen, X., Su, H., Xing, A., Chen, G., Zhu, J., Zhu, B. & Fang, J. (2022b). Phosphorus addition decreases soil fungal richness and alters fungal guilds in two tropical forests. Soil Biology and Biochemistry, 175, 108836 https://doi.org/10.1016/j. soilbio.2022.108836.

Martin, D. F. (1972). Marine chemistry, vol. 1 Analytical methods (p. 389). NY: Marcel Dekker Inc.

Mna, H. B., Alsubih, M., Oueslati, W., Helali, M. A., Amri, S., Added, A. & Aleya, L. (2022). Diagenetic processes and nutrients diffusive fluxes at the sediment-water interface in the Bizerte Lagoon (North Tunisia). Journal of African Earth Science, 196(3):1-15. https://doi.org/10.1016/j.jafrearsci.2022.104671

Minareci, O., Öztürk, M., Egemen, Ö. & Minareci, E. (2009). Detergent and Phosphate Pollution in Gediz River, Turkey. African Journal of Biotechnology, 8: 3568-3575. DOI: 10.5897/AJB09.167

Moraes, P. C., Sutti, B. O., Chiozzini, V. G. & Braga, E. S. (2023). Benthic aerobic respiration and nutrient fluxes in Cananéia-Iguape Estuarine-Lagoon complex along a salinity gradient. Ocean and Coastal Research, 71(1): 1-20. https://doi.org/10.1590/2675-2824071.22135pcm

Murray, L. G., Mudge, S. M., Newton, A. & Icely, J. D. (2006). The effect of benthic sediments on dissolved nutrient concentrations and fluxes. Biogeochemistry, 81:159–178. DOI 10.1007/s10533-006-9034-6

Ozkan, E. Y., Kocatas, A. & Buyukisik, B. (2008). Nutrient dynamics between sediment and overlying water in the inner part of Izmir Bay, Eastern Aegean. Environmental Monitoring and Assessment, 143:313–325. DOI 10.1007/s10661-007-9984-8

Parlak, H., Çakır A., Boyacıoğlu, M. & Çakal Arslan, Ö. (2006). Heavy Metal Deposition in Sediments from the Delta of the Gediz River (Western Turkey): A Preliminary Study. E.U. Journal of Fisheries & Aquatic Sciences, 23 (3-4): 445–448.

Rigaud, S., Radakovitch, O., Couture, R. M., Bruno Deflandre, B., Cossa, D., Garniere, C. & Garnier, J. M. (2013). Mobility and fluxes of trace elements and nutrients at the sediment–water interface of a lagoon under contrasting water column oxygenation conditions. Applied Geochemistry, 31: 35–51. https://doi.org/10.1016/j.apgeochem.2012.12.003

Sıkı, M. (2002). The Birds of Gediz Delta (Izmir Bird Paradise). Ekoloji, 11: 11-16.

Sisman Aydın, G. & Simsek, K. (2015). Homa Dalyanı (Izmir) Derinlestirme Çalısmalarının Nutrient Degisimleri Üzerine Etkilerinin Izlenmesi. 18. Ulusal Su Ürünleri Sempozyumu, İzmir.

Somay, A. M., & Filiz, S. (2003). Hidrology, hydrogeology and hydrochemistry of wetlands: a case study in Izmir Bird Paradise, Turkey. Environmental Geology, 43: 825–835. DOI 10.1007/s00254-002-0697-6

Strickland, J. D. H., & Parsons, T. R. (1972). A practical handbook of seawater analysis, Bull. No. 167 (p. 310). Ottawa: Fisheries Research Board of Canada.

Timmermann, K., Banta, S.T. & Glud, R.N. (2006). Linking Arenicola marina irrigation behavior to oxygen transport and dynamics in sandy sediments. Journal of Marine Research. 64, 915–938. DOI: 10.1357/002224006779698378

Tosunoglu, Z., Ünal, V., Kaykaç, M. H., Mermer, A. & Onem, R. (2015). Ege Dalyanlarının Güncel Durumu. Ege Üniversitesi Bilimsel Arastırma Projesi. No. SUF/006. Izmir, 332p.

Uluturhan, E., Kontas, A. & Can, E. (2011). Sediment concentrations of heavy metals in the Homa Lagoon (Eastern Aegean Sea): Assessment of contamination and ecological risks. Marine Pollution Bulletin, 1989-1997. https://doi.org/10.1016/j.marpolbul.2011.06.019

Zhang, L., Wang, L., Yin, K., Lü, Y., Yang, Y. & Huang, X. (2013). Spatial and seasonal variations of nutrients in sediment profiles and their sediment-water fluxes in the Pearl River Estuary, Southern China. Journal Earth Science, 25: 197–206 https://doi.org/10.1007/s12583-014-0413-y