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**Comparaison du fonctionnement de deux
écosystèmes aquatiques : Barrage de Cheliff (barrage
de dérivation) et Barrage de Kerrada (barrage de
stockage) dans la région Ouest d'Algérie**

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Comparison of the functioning of two aquatic ecosystems: Cheliff reservoir (diversion reservoir) and Kerrada reservoir (storage reservoir) in the western region of Algeria

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*In loving memory of my grandmother (Mamie Djida)
"May her soul rest in peace."*

*To all the women whose hearts burned with ambition,
yet whose dreams of learning were silenced,
their wings folded before they could fly.*

To Water, my source of positive energy.

To Science, to Passion, and to Humanity.

Lidia Baitiche

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Abstract

Plankton communities play a key role in reservoir functioning and water quality assessment. Phytoplankton, as primary producers, and zooplankton, as primary consumers, together form the foundation of aquatic food webs, reflecting trophic status and environmental changes. This thesis provides an integrated ecological assessment of two hydrologically connected reservoirs in northwestern Algeria, Cheliff (a shallow diversion reservoir) and Kerrada (a deep storage reservoir) which jointly supply over 155 hm³ of water annually to Mostaganem–Arzew–Oran (M.A.O.) axis, providing drinking, agricultural, and industrial water to more than two million people. The study aimed to evaluate how morphometry, hydrodynamics, and nutrient regimes influence water quality, plankton community structure, and trophic status under increasing anthropogenic and climatic pressures, while supporting sustainable freshwater management in Algeria. Two years of monthly monitoring were conducted to assess the composition, abundance, and diversity of phytoplankton and the main zooplankton groups (Rotifera, Copepoda, and Cladocera), along with detailed physicochemical parameters. Results revealed strong spatial and temporal contrasts between the two ecosystems. Cheliff reservoir showed higher temperature, turbidity, salinity, and organic matter, reflecting nutrient enrichment from agricultural and domestic sources. In contrast, Kerrada reservoir displayed greater physicochemical stability but remained sensitive to nitrate inflows from its watershed. Phytoplankton assemblages in Cheliff reservoir were dominated by Cyanophyceae, Chlorophyceae, and Euglenophyceae, while Kerrada reservoir supported Cyanophyceae and Dinophyceae. Cheliff reservoir showed frequent blooms of cyanobacteria and euglenoids, while Kerrada reservoir had milder blooms of green algae and dinoflagellates. Zooplankton composition reflected these environmental differences: rotifers dominated in Cheliff reservoir, whereas Kerrada supported more balanced assemblages of cladocerans and copepods. Redundancy analyses identified depth, temperature, oxygen, and nutrient gradients as key drivers of plankton dynamics. Integration of Carlson's Trophic State Index with plankton-based metrics classified Cheliff reservoir as eutrophic to hypereutrophic and Kerrada as mesotrophic. These results demonstrate that reservoir morphometry and hydrological regimes strongly govern plankton structure and trophic state in semi-arid Mediterranean environments. From a management perspective, eutrophication in Cheliff reservoir may degrade Kerrada's water quality and increase treatment costs. Maintaining good ecological conditions in Cheliff reservoir is therefore critical for the sustainability of the M.A.O. water system. This study provides the first comprehensive plankton-based ecological assessment of Algerian reservoirs and supports the development of adaptive monitoring tools and management strategies for nutrient control and freshwater sustainability in semi-arid regions.

Keywords: Biodiversity, Ecosystem functioning, Trophic state, Water quality, Plankton, Sustainable management, Global environmental changes.

Résumé

Les communautés planctoniques jouent un rôle clé dans le fonctionnement des barrages et l'évaluation de la qualité de l'eau. Le phytoplancton, en tant que producteur primaire, et le zooplancton, en tant que consommateur primaire, forment ensemble la base des réseaux trophiques aquatiques, reflétant l'état trophique et les changements environnementaux. Cette thèse présente une évaluation écologique intégrée de deux barrages hydrologiquement connectés du nord-ouest de l'Algérie : le barrage de Cheliff (peu profond, de dérivation) et celui de Kerrada (profond, de stockage), qui fournissent ensemble plus de 155 hm³ d'eau par an à l'axe Mostaganem–Arzew–Oran (M.A.O.), assurant l'approvisionnement en eau potable, agricole et industrielle de plus de deux millions d'habitants. L'étude visait à évaluer comment la morphométrie, l'hydrodynamisme et les régimes nutritifs influencent la qualité de l'eau, la structure des communautés planctoniques et l'état trophique, dans un contexte de pressions anthropiques et climatiques croissantes, tout en soutenant une gestion durable des ressources en eau douce en Algérie. Deux années de suivi mensuel ont permis d'analyser la composition, l'abondance et la diversité du phytoplancton et des principaux groupes de zooplancton (Rotifères, Copépodes et Cladocères), associés à des paramètres physico-chimiques détaillés. Les résultats ont révélé de fortes différences spatio-temporelles entre les deux écosystèmes. Le barrage de Cheliff présentait des températures, turbidité, salinité et teneurs en matière organique plus élevées, traduisant un enrichissement nutritif d'origine agricole et domestique. En revanche, le barrage de Kerrada montrait une plus grande stabilité physico-chimique, bien qu'il demeure sensible aux apports nitrates de son bassin versant. Les assemblages phytoplanctoniques étaient dominés, à Cheliff, par les Cyanophycées, Chlorophycées et Euglenophycées, tandis que Kerrada était dominé par les Cyanophycées et Dinophycées. Le zooplancton reflétait ces contrastes environnementaux : les rotifères dominaient à Cheliff, alors que Kerrada abritait des communautés plus équilibrées de cladocères et de copépodes. Les analyses de redondance ont identifié la profondeur et les gradients nutritifs comme principaux facteurs déterminant la dynamique planctonique. L'intégration de l'indice trophique de Carlson avec des métriques basées sur le plancton a permis de classer le barrage de Cheliff comme eutrophe à hypereutrophe et le barrage de Kerrada comme mésotrophe. Ces résultats démontrent que la morphométrie et le régime hydrologique des réservoirs contrôlent fortement la structure du plancton et l'état trophique dans les régions méditerranéennes semi-arides. D'un point de vue de gestion, l'eutrophisation du barrage de Cheliff pourrait dégrader la qualité de l'eau de Kerrada et augmenter les coûts de traitement. Le maintien d'un bon état écologique à Cheliff est donc essentiel pour la durabilité du système hydrique M.A.O. Cette étude constitue la première évaluation écologique complète basée sur le plancton dans les barrages algériens et soutient le développement d'outils de suivi adaptatif et de stratégies de gestion pour le contrôle des nutriments et la durabilité de l'eau douce dans les régions semi-arides.

Mots-clés : Biodiversité, Fonctionnement des écosystèmes, État trophique, Qualité de l'eau, Plancton, Gestion durable, Changements environnementaux globaux.

ملخص

تلعب مجتمعات العوالق دورًا أساسيًا في عمل السدود المائية وتقييم جودة المياه. فالعوالق النباتية، باعتبارها المنتج الأولي، والعوالق الحيوانية، بوصفها المستهلك الأولي، تُشكّلان معًا الأساس للشبكات الغذائية المائية وتعكسان الحالة التغذوية والتغيرات البيئية. تقدّم هذه الأطروحة تقييمًا بيئيًا متكاملًا لسدين مرتبطين هيدرولوجيًا في شمال غرب الجزائر: سد الشلف (ضحل ويُستخدم للتحويل) وسد كَرّادة (عميق ويُستخدم للتخزين)، واللذين يزودان أكثر من 155 مليون متر مكعب من المياه سنويًا لمحور مستغانم-أرزيو-وهران (M.A.O.)، لتلبية احتياجات أكثر من مليوني نسمة. هدفت الدراسة إلى تقييم تأثير الخصائص المورفومترية والديناميكية المائية وأنظمة المغذيات على جودة المياه وبنية مجتمعات العوالق وحالتها التغذوية، في ظلّ الضغوط البشرية والمناخية المتزايدة. أُجري رصد شهري على مدى عامين لتقييم تركيب ووفرة وتنوّع العوالق النباتية وأهم مجموعات العوالق الحيوانية (الدوّارات، ومجذافيات الأرجل، والكلاوسيرات) (*Rotifera, Copepoda, Cladocera*)، إلى جانب قياسات فيزيائية وكيميائية مفصّلة. أظهرت النتائج فروقًا بيّنة بين النظامين البيئيين؛ إذ تميّز سد الشلف بدرجات حرارة وعكارة وملوحة ومحتوى عضوي أعلى، ما يعكس إثراءً غذائيًا ناتجًا عن المصادر الزراعية والمنزلية. في المقابل، أظهر سد كَرّادة استقرارًا فيزيائيًا-كيميائيًا أكبر، غير أنّه ظلّ حساسًا لتدفّق النترات من حوضه المائي. هيمنت في سد الشلف أنواع من الطحالب الزرقاء (*Cyanophyceae*)، والخضراء (*Chlorophyceae*)، واليوغلينية (*Euglenophyceae*)، بينما هيمنت في سد كَرّادة الطحالب الزرقاء (*Cyanophyceae*) والدينوفلاجيلات (*Dinophyceae*). انعكست هذه الاختلافات أيضًا على مجتمعات العوالق الحيوانية؛ إذ سيطرت الدوّارات (*Rotifera*) في الشلف، بينما احتوى كَرّادة على تجمعات أكثر توازنًا من الكلاوسيرات (*Cladocera*) ومجذافيات الأرجل (*Copepoda*). وأظهرت التحليلات أن عمق السد وتدرّجات المغذيات هما العاملان الرئيسيان المؤثران في ديناميكية العوالق. كما أظهر دمج مؤشر كارلسون (TSI) للحالة التغذوية مع مؤشرات العوالق أن سد الشلف يتراوح بين الإثراء التغذوي العالي وفائق الإثراء، في حين يُعدّ سد كَرّادة متوسط التغذية. تؤكد النتائج أنّ خصائص الخزّانات تتحكّم في بنية العوالق وحالتها التغذوية في البيئات المتوسطة شبه الجافة. ويُحتمل أن يؤدي الإثراء التغذوي في سد الشلف إلى تدهور جودة مياه سد كَرّادة وزيادة تكاليف المعالجة، لذا فإن الحفاظ على الحالة البيئية الجيدة لسد الشلف يُعدّ ضروريًا لاستدامة نظام مياه محور مستغانم-أرزيو-وهران. تمثّل هذه الدراسة أول تقييم بيئي شامل للخزّانات الجزائرية اعتمادًا على العوالق، وتدعم تطوير أدوات رصد واستراتيجيات إدارة فعّالة للتحكّم في المغذيات وضمان استدامة المياه العذبة في المناطق شبه الجافة.

الكلمات المفتاحية: التنوع البيولوجي، وظائف النظم البيئية، الحالة التغذوية، جودة المياه، العوالق، الإدارة المستدامة، التغيرات البيئية العالمية.



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Lidia Baitiche



Dedication

*Above all, I dedicate this thesis to **Myself** as a strong, independent woman, an iron woman who has fought tirelessly to be here, and who will continue striving to achieve more and more. To my wings of ambition, may they soar ever higher.*

*To my **mother**, the eternal wellspring of tenderness and the embodiment of devotion, whose love, prayers, sacrifices, and wisdom have been my refuge and my strength. No words, no dedication, could ever match the greatness of what you deserve. For the noble values, the education, and the unwavering support you bestowed upon me I bow in eternal gratitude.*

*To my **father**, whose pride I carry as a crown, may you see in these pages the harvest of the countless sacrifices and hardships you endured to pave my path forward. This work is the living fruit of your devotion to my education and future.*

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Introduction





Introduction

Limnology, the science of inland waters, encompasses the biological, chemical, and physical characteristics of lakes and other freshwater bodies, treating them as integrated ecosystems. The term derives from the Greek word *limne* (lake or pond). Although it originated with the study of lakes, limnology has since expanded to include rivers, wetlands, and estuarine ecosystems (Gökçe, 2019). Today, it stands as a key branch of ecology because of its focus on environmental constraints, population dynamics, and community interactions in aquatic ecosystems.

Freshwater ecosystems, though covering less than 1% of the Earth's surface, are biodiversity hotspots that provide essential ecosystem services (Dudgeon et al., 2006; Reid et al., 2019; Antão-Geraldes & Calheiros, 2023). They sustain nearly 10% of all known aquatic species, including many vertebrates, and contribute to nutrient cycling, water purification, and climate regulation (Moss, 2010; Reid et al., 2019). Beyond their ecological role, they are vital for human well-being, supplying drinking water, food, energy, raw materials, and cultural benefits (Cabral et al., 2019; Yin et al., 2022; Orduna et al., 2024).

Among these ecosystems, reservoirs have become increasingly relevant due to their role in water supply, irrigation, flood control, hydropower generation, and biodiversity conservation (Tundisi & Matsumura-Tundisi, 2008; Naselli-Flores, 2010; Orduna et al., 2024). Unlike natural lakes, reservoirs are artificial water bodies characterized by distinct morphometry, hydrological regimes, and ecological functioning (Wetzel, 2001; Naselli-Flores & Barone, 2005; Almeida et al., 2020; Pinto et al., 2023). Their water quality and biodiversity patterns are strongly shaped by depth, residence time, catchment features, and management practices (Margalef, 1983; Dokulil & Teubner, 2000; Navarro et al., 2009; Park et al., 2014; Merouche et al., 2020). This issue is particularly critical in Mediterranean and semi-arid regions, where reservoirs are highly vulnerable to climatic extremes, such as irregular rainfall, seasonal droughts, and intense evaporation, all of which intensify fluctuations in physicochemical conditions and nutrient dynamics (Mhamdi et al., 1994; Cherbi et al., 2008; Nasri et al., 2016; Soltani et al., 2020). Anthropogenic pressures, such as agricultural runoff, wastewater discharge, and industrial inputs, further enhance eutrophication risks and promote harmful algal blooms (Simões et al., 2015; Guermazi et al., 2023).



Biological communities, especially plankton, are central to reservoir functioning and water quality assessment. Phytoplankton, as primary producers, form the base of aquatic food webs and respond rapidly to environmental changes, making them sensitive bioindicators (Sommer, 1989; Reynolds, 2006; Beaver et al., 2018; Sendra et al., 2019; Cui et al., 2021; Tokatlı et al., 2024; Molner et al., 2025). Their community structure is shaped by key drivers such as nutrient availability, light, temperature, and hydrodynamics (Reynolds, 1984; Cui et al., 2021; Wang L. et al., 2024). Zooplankton, as primary consumers, link phytoplankton to higher trophic levels and regulate ecosystem stability through grazing and nutrient recycling (Carpenter et al., 1985; Lampert, 2006; Jeppesen et al., 2011; Ger et al., 2016; Stamou et al., 2021; Muñoz-Colmenares et al., 2021; Mallin, 2023; Orduna et al., 2024).

In Algeria, limnological research has primarily focused on phytoplankton taxonomy and the influence of environmental factors on species composition in lakes and reservoirs (Baudrimont, 1974; Hamidi et al., 2009; Djabourabi et al., 2017; Arab et al., 2019; Chaffai et al., 2022; Djerboua et al., 2022; Benmatouk et al., 2024). Zooplankton studies have largely emphasized taxonomic inventories and environmental correlations (Gauthier, 1928; Samraoui et al., 1998; Mouelhi et al., 2000; Cherbi et al., 2008; Bouazzara et al., 2021; Djeddar et al., 2021; Hamil et al., 2021; Smaoune et al., 2021; Ould Rouis et al., 2022; Akli-Bidi et al., 2023; Amar et al., 2023; Djabourabi et al., 2024). However, ecological approaches integrating diversity and abundance to assess community structure and its environmental drivers remain scarce, particularly in semi-arid reservoirs characterized by strong hydrological variability. Reservoir topography was first described for Hamiz reservoir by Ould Rouis et al. (2022), but only as contextual background for sampling design and interpretation. Moreover, studies on the effects of reservoir morphometry and water-level fluctuations on plankton community dynamics are limited. The first investigation in this context was conducted by Baitiche et al. (2025), who examined the influence of morphometry, water levels, and hydrodynamics on zooplankton community structure. Consequently, this thesis represents the first contribution to addressing these gaps in Algeria.

Globally, phytoplankton and zooplankton are increasingly recognized as effective bioindicators of ecological integrity and trophic status (Ejsmont-Karabin, 2012; Kruk et al., 2021). Modern water quality frameworks, such as the European Water Framework Directive (WFD), advocate combining biological, physicochemical, and hydromorphological elements to assess ecological potential (Moss et al., 2003; Montagud et al., 2019). Trophic State Indices (Carlson, 1977) and plankton-based indices, both phytoplankton (Barbe et al., 2003; Feret & Laplace-



Treyture, 2013; Laplace-Treyture & Feret, 2016) and zooplankton (Ejsmont-Karabin, 2012; Ejsmont-Karabin & Karabin, 2013) are widely applied to diagnose eutrophication and ecological functioning. Yet in Algeria, water quality assessments remain largely dependent on physicochemical measures, with limited use of plankton-based indices. Only a few studies have applied such approaches: Guechaoui-El Haouati (2015) used phytoplankton-based indices to evaluate water quality in 65 Algerian reservoirs, while Arab (2021) applied a similar approach to assess Boukourdane reservoir water quality. Doukhandji and Arab (2017), as well as Smaoune et al. (2021), considered rotifers as bioindicators.

Cheliff and Kerrada reservoirs in northwestern Algeria provide a valuable case study for addressing these research gaps. Both reservoirs are central to an inter-basin water transfer project designed to secure irrigation and drinking water for the M.A.O. corridor (Mostaganem, Arzew, Oran), with an annual capacity of ~155 million m³, intended to serve nearly two million inhabitants by 2030 (National Agency for Dams and Transfers [N.A.D.T.], 2017; Baitiche et al., 2025). Cheliff, a shallow diversion reservoir, and Kerrada, a deep storage reservoir, present contrasting morphometric and hydrological characteristics, providing a natural framework to examine how these factors influence plankton dynamics and water quality.

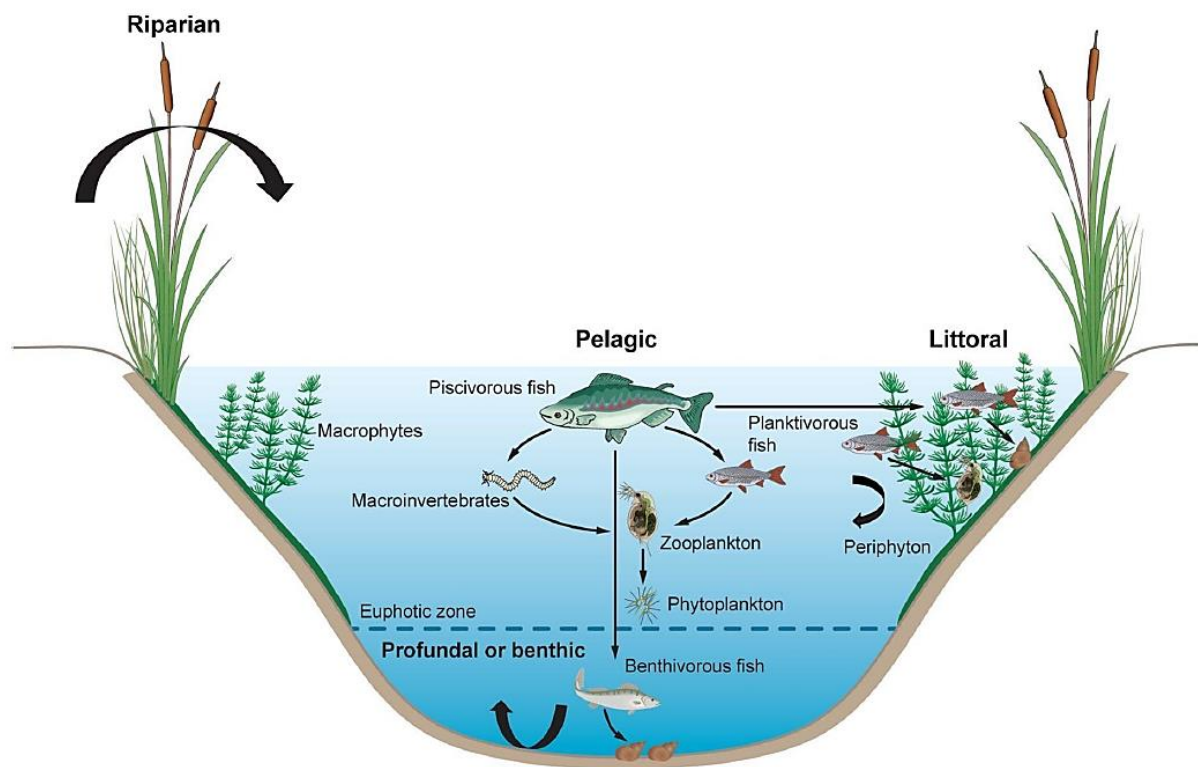
This thesis examines the ecological functioning and water quality of two contrasting reservoirs over two years of monitoring, integrating physicochemical analyses with phytoplankton and zooplankton community studies. The main socioeconomic objective of this research is to support the sustainable management of Algeria's strategic freshwater resources. By analyzing the spatial and temporal dynamics of plankton communities and identifying harmful algal blooms (HABs) in Cheliff and Kerrada reservoirs, the study provides essential insights into ecosystem functioning and water quality regulation. The specific objectives are to: **(i)** characterize the environmental context of Cheliff and Kerrada reservoirs; **(ii)** evaluate phytoplankton and zooplankton diversity, abundance, and community composition in relation to environmental drivers; **(iii)** assess seasonal and spatial variations in water quality using multivariate statistical approaches; **(iv)** apply plankton-based and trophic indices to evaluate ecological potential; and **(v)** translate ecological findings into actionable knowledge for freshwater resource management, with direct socioeconomic relevance to the Mostaganem–Arzew–Oran (M.A.O.) water supply axis. By integrating physicochemical and biological indicators, this research provides a holistic framework for water quality monitoring, early warning systems, sustainable reservoir management, and biodiversity conservation in semi-arid Mediterranean regions.



The thesis is structured into six chapters:

- **Introduction** – presents the research context, knowledge gaps, and objectives.
- **Chapter I. Literature review** – outlines freshwater ecosystem types with emphasis on reservoirs; compares natural lakes and reservoirs, shallow versus deep ecosystems; reviews plankton communities, their roles in food webs and nutrient cycling, their use as bioindicators; and discusses approaches to water quality assessment.
- **Chapter II. Study area** – describes the geographical, hydrological, climatic, and ecological context of Cheliff and Kerrada reservoirs.
- **Chapter III. Environmental parameters** – analyzes spatial and seasonal variability of physicochemical characteristics in the two reservoirs.
- **Chapter IV. Phytoplankton community structure** – examines phytoplankton diversity, abundance, and species composition in relation to environmental variables, reservoir morphometry, depth, and water-level fluctuations.
- **Chapter V. Zooplankton community structure** – explores the structure, diversity and temporal dynamics of Rotifera, Copepoda, and Cladocera groups, highlighting inter-reservoir differences and their responses to environmental fluctuations.
- **Chapter VI. Plankton-based assessment** – applies Carlson's Trophic State Indices and plankton-based metrics (phytoplankton and zooplankton indices) to evaluate ecological quality and trophic status, highlighting the strengths and limitations of each index under Mediterranean conditions.
- The thesis concludes with a **Conclusion and Perspectives**, summarizing the main findings and proposing recommendations for future research and reservoir management.

Chapter I: Literature review



(By Meerhoff & de los Ángeles González-Sagrario, 2021)



This chapter provides a comprehensive review of the theoretical and scientific background underlying the present study. The literature review is structured as follows: an overview of freshwater ecosystem types, with particular attention to reservoirs; a comparison between natural lakes and artificial reservoirs; a comparison of the functioning of shallow and deep reservoirs and the environmental drivers shaping these ecosystems; a review of plankton community structures, their dynamics in reservoirs, their functional roles in food webs and nutrient cycling, their roles as bioindicators, and historical overview of plankton research in Algeria; followed by an outline of water quality assessment approaches and an exploration of reservoir ecosystem functioning.

I.1. Freshwater ecosystems

Freshwater ecosystems, including lakes, rivers, wetlands, and reservoirs, are defined by their low salinity. They serve as critical habitats for biodiversity and essential contributors to global ecological processes (Dijkstra et al., 2014). These freshwater bodies support a disproportionately high share of biodiversity and provide vital ecosystem services such as nutrient cycling, water purification, primary productivity, and habitat provision (WWAP, 2015; Harrison et al., 2016; Carrizo et al., 2017). These ecosystems are shaped by dynamic environmental factors, including precipitation, evaporation, and hydrological flows, which create both stable and variable biomes. However, freshwater ecosystems are among the most threatened globally, facing intense pressure from anthropogenic activities, pollution, hydrological alterations, and climate change (WWF, 2016). Nearly one-third of freshwater species are at risk of extinction (IUCN, 2016), and over 750 million people lack access to safe drinking water, a crisis projected to worsen as global water demand is expected to rise by 400% by 2050 (WWAP, 2015). Human-induced disturbances affect both ecological and evolutionary dynamics, altering productivity, species interactions, and community structures (Currie, 1991; Willig et al., 2003; Krebs, 2009).

Freshwater ecosystems can be classified into two main types based on water movement: **lotic ecosystems** (flowing waters like rivers and streams) and **lentic ecosystems** (standing waters such as lakes and ponds).



I.1.1. Lotic ecosystems (flowing waters)

Lotic ecosystems, including rivers and streams, are defined by the presence of unidirectional flow, which strongly influences sediment transport, oxygenation, and nutrient dynamics (Allan & Castillo, 2007; Chakraborty, 2021). Biological communities in lotic ecosystems are adapted to current velocities and frequent disturbances, which create dynamic habitat conditions (Wilzbach & Cummins, 2008; Mallin, 2024). Primary production is often limited by light availability, due to shading by riparian vegetation or turbidity (Hildrew & Giller, 2023). Recent studies highlight how increasing land-use changes and climate variability further alter light regimes and nutrient inputs (Schneider & Petrin, 2017; Chakraborty, 2021; Tonkin et al., 2021; Morin-Crini et al., 2022).

I.1.2. Lentic ecosystems (standing waters)

Lentic ecosystems refer to bodies of standing water, such as lakes, ponds, and reservoirs, where water movement is generally limited and occurs primarily through internal convective mixing and wind-driven circulation at the surface (Eramma et al., 2023). These ecosystems are characterized by vertical stratification in the water column, with seasonal and depth-dependent variations in temperature, oxygen levels, nutrient concentrations, and light penetration (Kalf, 2002; Bade, 2005). Consequently, the distribution and diversity of aquatic organisms differ significantly across these zones. Lentic ecosystems typically support high biodiversity, especially in the photic zone, where sunlight facilitates primary production by phytoplankton and aquatic macrophytes (Wetzel, 2001). Their productivity and ecological dynamics are largely influenced by internal processes such as sedimentation, decomposition, and biological interactions (Eramma et al., 2023). Shallow lentic ecosystems are often polymictic and nutrient-rich, while deeper ecosystems may exhibit stable thermal stratification during summer months (Wetzel, 2001; Jana, 2024). These structural differences are crucial in shaping ecosystem functioning, particularly for planktonic communities.



I.2. Reservoir ecosystems

I.2.1. Definition and purpose of reservoirs

Water reservoirs are large, open storage structures designed for the collection and storage of water, typically enclosed by embankments or a dam (Pei, 2023). Also referred to as artificial lakes or impoundments. The term ‘dam’ typically refers to the physical barrier that holds back a water body, formed by impounding a river's flowing waters to create a reservoir. Dams give rise to new ecosystems that have hybrid characteristics of lakes and rivers, structured into distinct longitudinal zones within the reservoir (Soballe et al., 1992; García-Chicote et al., 2018).

These man-made water bodies have been constructed for multiple purposes, such as freshwater supply, irrigation, flood control, hydropower generation, and recreational activities (Charles et al., 2012; Pomari et al., 2019; Pinto et al., 2021a; Ma et al., 2022). Reservoirs can be categorized based on their construction purpose and dam location (Galizia-Tundisi, 2018; Nikolić et al., 2020). Considering terrain characteristics, reservoirs can be classified into four main types: **plain reservoirs** (Guo et al., 2020), **valley reservoirs** (Li et al., 2022), **underground reservoirs** (Menéndez et al., 2022), and **coastal reservoirs** (Yang, 2022). Additionally, based on their use features, reservoirs fall into several categories, including **storage reservoirs** (Hong et al., 2010), **flood control reservoirs (diversion)** (Huang et al., 2018), **distribution reservoirs** (Abplanalp, 1976), and **multipurpose reservoirs** (Di Baldassarre et al., 2018). Furthermore, in terms of function, reservoirs can be classified into three primary types: **bank-side reservoirs** (Steel & Duncan, 1999), **valley-dammed reservoirs** (Bábek et al., 2020), and **service reservoirs** (Sathasivan et al., 2010). Thus, reservoirs may be categorized using different classification criteria, leading to variations in terminology.

I.2.2. Reservoir characteristics

Reservoirs exhibit both **riverine** and **lacustrine** characteristics, structured along longitudinal, lateral, and vertical gradients that shape their physical, chemical, and biological dynamics (Cole, 1983; Thornton, 1990; Wetzel, 2001; García-Chicote et al., 2018). Along the **longitudinal gradient**, reservoirs transition through three zones: **riverine**, **transitional**, and **lacustrine** (Fig. 1). The **riverine zone** is narrow, well-mixed, and characterized by high turbidity, significant organic input, and strong advective flow, which limit light penetration and primary production. The **transitional zone** features reduced flow velocity, improved light penetration, and enhanced phytoplankton productivity. The **lacustrine zone** resembles natural lakes, with thermal



stratification influencing planktonic production, nutrient availability, and organic matter sedimentation (Wetzel, 2001; Martin et al., 2007).

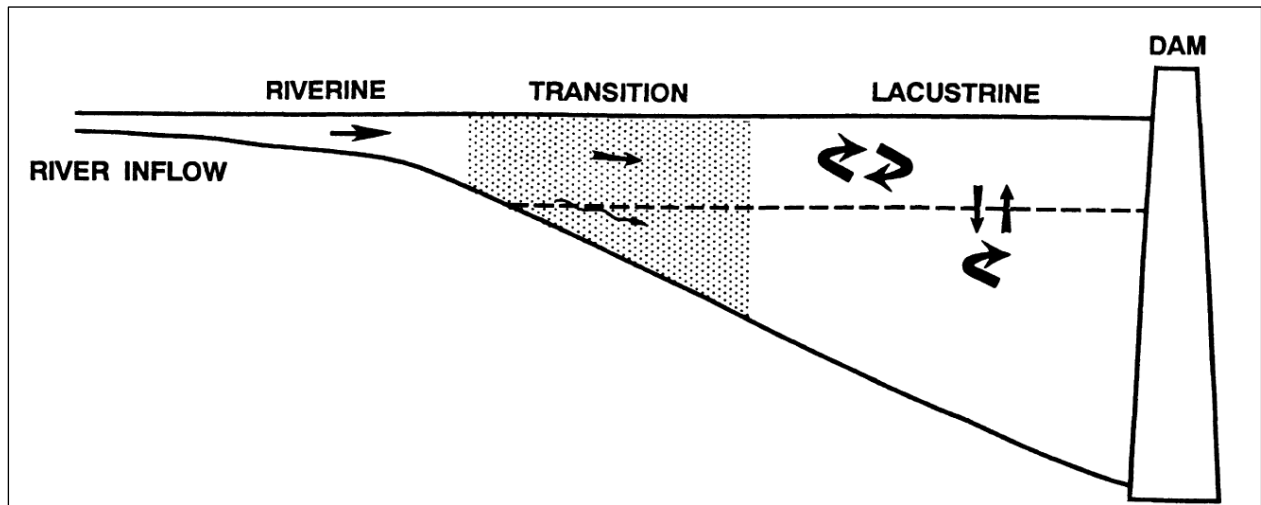


Fig. 1 Generalized zones along longitudinal gradients in reservoirs (Wetzel, 2001).

Lateral and vertical gradients further influence water movement and habitat structure. Flow velocity typically decreases from the inflow to the dam and from the mid-channel to the banks, while vertical thermal stratification results in the formation of distinct layers: **epilimnion**, **thermocline**, and **hypolimnion** (Fig. 2).

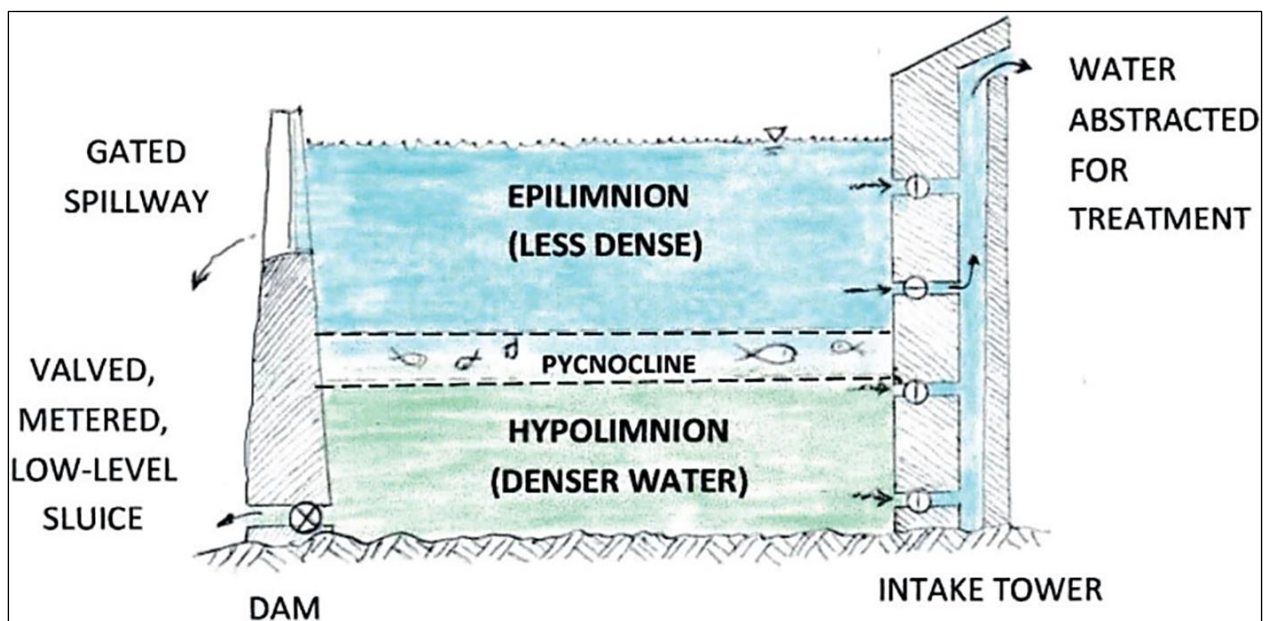


Fig. 2 Schematic profile of a density-stratified reservoir (French, 2022).



I.2.3. Reservoir ecological functioning

Reservoirs are dynamic freshwater ecosystems whose ecological functioning is shaped by a complex interplay of physical, chemical, and biological processes. They are ecologically heterogeneous, varying in size, morphometry, water residence time, and degree of anthropogenic influence. These factors determine stratification regimes, mixing patterns, nutrient cycling, and the composition of biological communities (Tundisi & Matsumura-Tundisi, 2008). Understanding these dynamics is essential for assessing reservoir health, managing water quality, and preserving aquatic biodiversity (Straskraba & Tundisi, 1999; Tundisi et al., 2008; Banerjee et al., 2017). For instance, narrow, river-like reservoirs with short retention times tend to display more lotic characteristics, whereas wider and deeper reservoirs exhibit lentic features such as stable stratification and well-developed pelagic zones (Thornton et al., 1990). Unlike natural lakes, reservoirs are subjected to regulated hydrological regimes, including controlled inflows and outflows, and often experience rapid water-level fluctuations driven by operational requirements (Tundisi & Matsumura-Tundisi, 2008). Consequently, both reservoir design and operational management must be considered when evaluating their ecological status.

I.2.3.1. Physical drivers of ecological functioning in reservoirs

I.2.3.1.1. Wind and rainfall

Wind and rainfall are key physical forces shaping vertical structure and ecological dynamics. Wind promotes vertical mixing, disrupts thermal gradients, and resuspends sediments particularly phosphorus, effects that dominate in dry seasons (Matsumura-Tundisi & Tundisi, 1976; Nogueira & Matsumura-Tundisi, 1996; Wang et al., 2023). Such mixing can trigger algal blooms (Chalar & Tundisi, 2001; Banerjee et al., 2017). Rainfall enhances allochthonous input and turbidity, especially in summer. Both wind and rainfall influence the horizontal and vertical distribution of phytoplankton and zooplankton, as shown in shallow eutrophic ecosystems (George & Edwards, 1976; Blottiere, 2015; Zhang et al., 2017; Deng et al., 2018).

I.2.3.1.2. Thermal stratification

Thermal stratification driven by solar radiation and moderated by wind mixing, is a central feature influencing reservoir ecology. It separates the water column into three layers: the warm, well-mixed epilimnion; the transitional metalimnion; and the cold, relatively quiescent hypolimnion (Lampert & Sommer, 2007). It is most pronounced in reservoirs with residence times >40 days, whereas shallow ecosystems are often polymictic with frequent mixing (Soullignac et al.,



2014; Banerjee et al., 2017). This type of stratification does not apply to all lakes and reservoirs. Some reservoirs may only have an epilimnion and a metalimnion (Bade, 2005). Stratification depth can also be influenced by outlet withdrawal depth and inflows characteristics (Casamitjana et al., 2003). Thermal stratification, typically occurring in warm seasons, it limits vertical mixing, thereby shaping oxygen distribution, light penetration, and nutrient fluxes (Wetzel, 2001). These processes, in turn, regulate phytoplankton composition and productivity (Padisák et al., 2003). In monomictic ecosystems, stratification may persist for several months, followed by complete mixing in winter or early spring.

I.2.3.1.3. Hydrodynamics and water retention

Reservoir hydrodynamics, driven by inflows, outflows, wind mixing, and morphometry, strongly influence spatial heterogeneity, nutrient distribution, and overall biological dynamics (Tundisi et al, 1984; Thornton, 1990) (Fig. 3). Mid-level water intakes can induce hydraulic stratification, modifying temperature and chemical profiles. Tributary inflows often introduce nutrients and suspended solids, while outflow zones tend to be clearer and nutrient-depleted. Outflows may also selectively remove surface-dwelling taxa (e.g., *Protozoa*, *Rotifera*), thereby altering community structure and promoting coexistence of congeneric species (Braga et al., 1998; Rocha et al., 2020).

Water retention time (WRT) is a key driver of biological components, particularly plankton dynamics. Short WRTs flush out phytoplankton, favoring fast-growing taxa, whereas long WRTs promote biomass accumulation, sedimentation, and grazing losses (Perry et al., 1990; Havens et al., 2001). In run-of-the-river dams (RT < 7 days), primary production is limited (Soballe et al. 1992). For zooplankton, a ~6-day WRT can maximize *Cladocera* biomass and enhance phosphorus depletion through grazing (Ejsmont-Karabin et al., 1993; Błędzki & Ellison, 2000). In contrast, short or highly variable WRTs reduce diversity and favor r-strategist species (Schmid-Araya & Zuñiga, 1992).

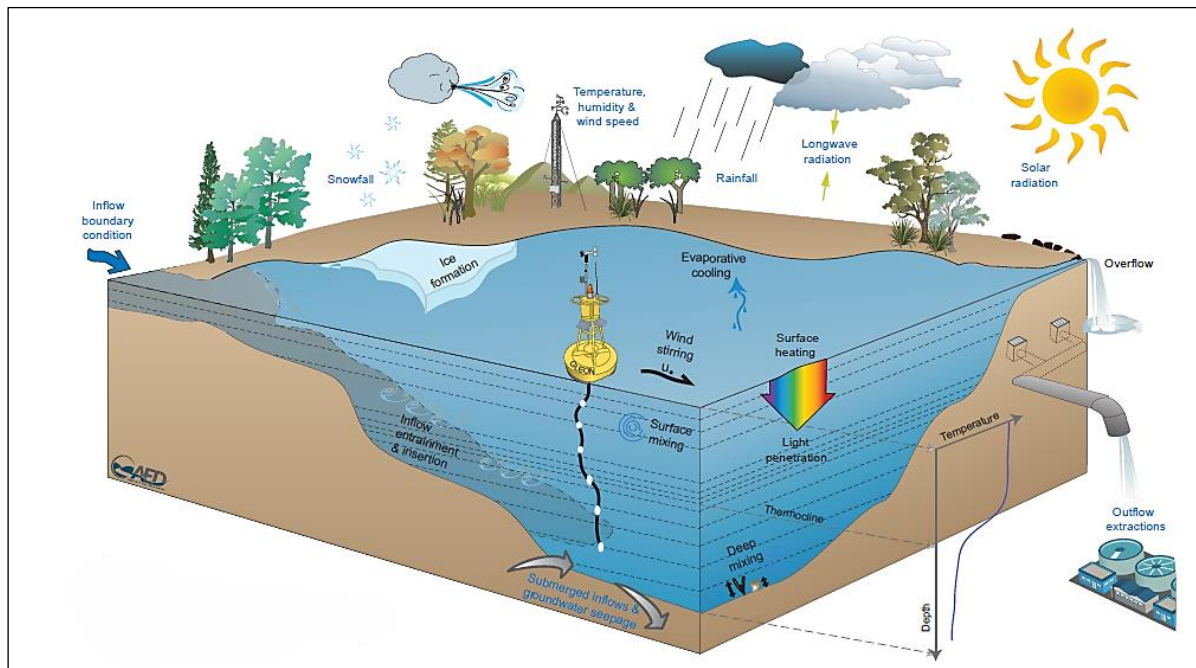


Fig. 3 Physical drivers and hydrodynamic processes shaping reservoirs (Hipsey et al., 2019).

I.2.3.2. Chemical drivers of ecological functioning in reservoirs

Reservoir ecosystems are regulated by a complex interplay of chemical parameters derived from both **allochthonous** (external) and **autochthonous** (internal) inputs. Key controlling factors include water temperature, turbidity, pH, conductivity, dissolved oxygen, and the extent of the euphotic zone, which collectively influence nutrient dynamics and biological productivity (Gemelgo et al., 2009). Conductivity, hardness, alkalinity, and salinity reflect the mineral composition of the water (Straškraba & Tundisi, 1999). Phosphorus is often the primary driver of productivity and eutrophication in temperate reservoirs, while nitrogen compounds, particularly ammonia in riverine zones, also play crucial role. Under low-oxygen conditions, denitrification processes can reduce nitrate concentrations (Schindler, 1974; Correll, 1998; González et al., 2004). Dissolved oxygen typically declines with depth and is influenced by latitude and seasonal stratification (Nogueira et al., 1999; Romero et al., 2004; León-Palmero et al., 2023). Additionally, the littoral zone plays a key ecological role and may act as a source of greenhouse gases like N_2O (Wang et al., 2006; León-Palmero et al., 2023). Silicate availability, essential for diatoms growth, can become limiting due to nutrient retention caused by damming, thereby impacting phytoplankton composition and productivity (Humborg et al., 1997; León-Palmero et al., 2023).



I.2.3.3. Biological components of reservoirs

Water reservoir ecosystems are critical freshwater habitats that support a disproportionately large array of biodiversity relative to their surface area coverage. Freshwater biodiversity encompasses organisms ranging from microbes to fish and provides essential ecosystem services, including food provision, water purification, climate regulation, and cultural values (Lynch et al., 2023). Reservoir biota includes autotrophic or primary producers (phytoplankton, periphyton, macrophytes) and heterotrophic organisms such as zooplankton and benthic invertebrates as primary consumers, as well as fish functioning as secondary and tertiary consumers. These components form interconnected trophic networks.

Phytoplankton are the dominant primary producers in reservoir ecosystems, forming the foundation of aquatic food webs (Almeida et al., 2020; Huang et al., 2024). Their diversity and functional composition respond to nutrient regimes and light availability. Aquatic macrophytes represent significant biomass contributors, providing critical habitat structure and influencing nutrient cycling. Periphyton communities, including epilithic biofilms, play important roles in primary production and are influenced by reservoir operations, with photosynthetic biofilm growth showing seasonal variations downstream of dam structures (Roussel et al., 2024).

Zooplankton communities act as crucial links between primary producers and higher trophic levels in reservoir food webs (Bakhtiyar et al., 2020; Declerck & de Senerpont Domis, 2023). They exhibit adaptations to predation pressure and food quality (Brett et al., 2000). Benthic invertebrates form essential components of reservoir ecosystems, contributing significantly to total ecosystem biomass (Kopylov et al., 2018). Aquatic insects are also important biodiversity elements, with studies documenting 16 species from 14 families and 4 orders in reservoir ecosystems (Kulkarni & Zade, 2020).

Fish communities in reservoirs display complex trophic relationships and serve as both secondary and tertiary consumers (Brönmark et al., 1997; Smith et al., 2018; Córdova-Tapia et al., 2025). Often stocked or influenced by reservoir connectivity, they exert top-down controls on zooplankton and indirectly affect phytoplankton biomass and structure (Jeppesen et al., 2005). Top predators in reservoir ecosystems include piscivorous fish, reptiles, amphibians, and birds. Food web studies reveal complex predator–prey relationships, with trophic levels comprising detritivores, herbivores, microcarnivores/omnivores, carnivores, and apex carnivores (Adite et al.,



2018). Predator presence can significantly influence community structure, with planktivorous fish predation directly impacting zooplankton biomass and composition (Ersoy et al., 2019).

Microbial communities, including heterotrophic bacteria, are regulated by both top-down and bottom-up controls, while viruses play specialized roles in aquatic microbial food webs (Mostajir et al., 2015; Kopylov et al., 2018). These microbes are fundamental to ecosystem processes such as decomposition, nutrient cycling, and energy transfer through food webs.

The integration of these biological compartments is essential for understanding the ecological status, functioning, and resilience of reservoir ecosystems.

I.2.4. Influence of morphometry on physical, chemical, and biological processes

Reservoir morphometry refers to the physical dimensions and shape of a water body, including depth, surface area, volume, and shoreline complexity (Faithfull et al., 2011; Kraemer et al., 2015). These parameters are critical determinants of thermal structure, water circulation, sedimentation patterns, and habitat heterogeneity (Håkanson, 2005). Morphometric characteristics further influence light penetration, nutrient distribution, and the potential for thermal stratification, all of which significantly affect primary production and trophic dynamics (Wetzel & Likens, 1991; Wetzel, 2001). Based on morphometric criteria, this study distinguishes between two types of reservoirs: **shallow diversion reservoirs** and **deep storage reservoirs**.

I.2.4.1. Shallow diversion reservoir

Shallow diversion reservoirs are typically constructed by damming rivers to divert water for irrigation, domestic, or industrial use. These ecosystems are generally less than 10 meters deep and they are typically polymictic, mixing frequently due to wind action and lacking thermal stratification (Grigorszky et al., 2006; Padisák et al., 2006; Banerjee et al., 2017; Wang X. et al., 2022). As a result, they maintain a well-oxygenated but highly dynamic water column, where nutrients are rapidly cycled between sediments and surface waters. The strong benthic–pelagic coupling in these reservoirs enhances their susceptibility to eutrophication, especially under increased nutrient loads (Padisák et al., 2006; Nöges, 2009; Yang et al., 2020). Shallow diversion reservoirs often exhibit high primary productivity and respond quickly to environmental fluctuations, making them particularly sensitive to anthropogenic pressures. Nutrient availability and sediment resuspension are key drivers shaping their biodiversity and community structure. Moreover, water diversion infrastructure can lead to ecological and environmental degradation in



donor regions, including river fragmentation (Grant et al., 2012), increased salinization (Zhuang, 2016), and rising water and energy demands that contribute to unsustainable urban growth (Ming et al., 2017). Damming associated with such projects alters nutrient transport and biogeochemical cycles of nitrogen, phosphorus, and carbon, often resulting in nutrient retention and enhanced greenhouse gas emissions from reservoir systems (Yang et al., 2020; Wang X. et al., 2022). Despite these challenges, shallow diversion reservoirs offer certain advantages, such as lower construction complexity and rapid implementation for short-term water resource management.

I.2.4.2. Deep storage reservoir

Deep reservoirs, typically exceeding 15–20 meters in depth, are characterized by the development of thermal stratification during the warmer months. This stratification results in a distinct layering of the water column, with a warm, well-mixed epilimnion overlying a cold, isolated hypolimnion (Kimmel et al., 1990; Wang H. et al., 2024). The vertical separation of these layers limits mixing and oxygen diffusion to deeper waters, leading to hypoxic or anoxic conditions in the hypolimnion and the accumulation of nutrients such as phosphorus and nitrogen (Tundisi & Matsumura-Tundisi, 2008; Merino-Ibarra et al., 2021). These processes contribute to internal nutrient loading, particularly during periods of stratification breakdown and mixing events. The depth of the thermocline, or mixing depth (Z_{Mix}), is a critical parameter that governs physico-chemical and biological dynamics in stratified reservoirs. Above this depth, in the euphotic zone, light availability supports primary production by phytoplankton, leading to oxygen production and CO₂ uptake. Below this zone, however, decomposition of organic matter and limited oxygen input lead to oxygen depletion and CO₂ accumulation, which can trigger anoxic conditions (Wang H. et al., 2024). Consequently, deep reservoirs tend to have more stable thermal and physical environments than shallow ones, but are more susceptible to hypolimnetic oxygen depletion and associated biogeochemical changes. These vertical gradients influence the structure and function of aquatic communities, which are often vertically zoned in response to temperature, oxygen, and nutrient availability (Wetzel, 2001).



I.3. Phytoplankton communities

Phytoplankton are microscopic, photosynthetic organisms that inhabit aquatic environments and float freely. They are transported either by their own motile structures, such as flagella, or passively by water currents. The term "phytoplankton" originates from the Greek words *phyton* (φυτόν = plant) and *planktos* (πλαγκτός = wanderer or drifter). The broader term "plankton" was introduced by German biologist Victor Hensen in 1887 to describe all living organic particles that drift freely in the open water column, independent of the shoreline or the bottom (Ruttner, 1940; Hutchinson, 1957; Pal & Choudhury, 2014; Wehr et al., 2015).

Phytoplankton form the foundation of aquatic food webs and play a pivotal role in the ecological functioning and trophic dynamics of freshwater ecosystems, including artificial water bodies such as reservoirs. As primary producers, phytoplankton convert solar energy into biomass through photosynthesis and initiate energy flow through aquatic trophic networks. Their composition, abundance, and productivity are tightly linked to environmental variables, making them effective indicators of ecosystem status and change (Reynolds, 2006; Padisák et al., 2009; Pal & Choudhury, 2014; Beaver et al., 2018). In reservoirs, where hydrological regimes and nutrient dynamics are highly variable, phytoplankton communities are especially dynamic and often reflect shifts in ecological conditions as well as anthropogenic influences.

1.3.1. Evolution of phytoplankton taxonomy

The classification of phytoplankton has undergone significant revisions over time, evolving from morphological-based systems to those incorporating cellular ultrastructure and molecular data. Early classical taxonomy, such as that of Fritsch (1935), grouped algae into 11 classes based mainly on pigmentation and morphology, including groups like *Chlorophyceae*, *Bacillariophyceae*, and *Myxophyceae*. Later, Smith (1950) proposed a simplified system with **seven algal divisions**, recognizing broader groups like *Chlorophyta*, *Euglenophyta*, *Cyanophyta*, and *Rhodophyta*. Building on this, Prescott (1984) refined the system by adopting the **phyla concept**, dividing algae into seven phyla with multiple classes and orders, distinguishing groups such as *Chlorophyceae*, *Bacillariophyceae*, and *Dinokontae* based on reproductive modes and cellular features. A major advancement came with Bold and Wynne's (1985) classification, which is among the most widely accepted classical systems. They proposed **nine divisions**, notably introducing *Charophyta* as a distinct division and evolutionary precursor to terrestrial plants. Their classification included detailed orders within divisions such as *Chlorophyta*, *Phaeophyta*, and *Chrysophyta*. Finally, Lee (2008) presented a **modern phylogenetic system**, incorporating



ultrastructural and endosymbiotic origins of chloroplasts. Algae were grouped into four major categories:

1. **Prokaryotes** (*Cyanobacteria*);
2. **Eukaryotic algae with two chloroplast membranes** (*Glaucophyta*, *Chlorophyta*, *Rhodophyta*);
3. **Eukaryotes with one chloroplast endoplasmic reticulum (ER)** (*Euglenophyta*, *Dinophyta*, *Apicomplexa*);
4. **Eukaryotes with two chloroplast ER membranes** (*Cryptophyta*, *Heterokontophyta*, *Prymnesiophyta*).

This progression reflects a transition from **morphological to molecular and evolutionary approaches**, aligning algal taxonomy with broader biological systematics.

1.3.2. Components of freshwater phytoplankton

Phytoplankton comprise two main types of organisms distinguished by their cellular structure: **eukaryotes** (true algae, with a nucleus) and **prokaryotes** (cyanobacteria, without a nucleus) (Prescott et al., 2003; Wehr et al., 2015). In aquatic environments, particularly in reservoir ecosystems, 9 principal microalgal classes are identified based on morphological, cytological, biochemical, and reproductive criteria, following the classification system proposed by Bold and Wynne (1985), which remains one of the most widely accepted classical taxonomic frameworks.

1.3.2.1. Cyanophyceae and Prochlorophyceae

Also referred to as blue-green microalgae, Cyanophyceae are prokaryotic organisms that can form colonies, filaments, or unicellular structures (Fig. 4). Unlike heterotrophic prokaryotes, they contain chlorophyll-a and accessory pigments such as phycocyanin, phycoerythrin, and carotenoids (Ganf et al., 1991; Schagerl & Donabaum, 2003; Colyer et al., 2005; Bernard, 2014). They often exhibit buoyancy due to the presence of gas vesicles and are capable of nitrogen fixation. Morphologically, they lack membrane-bound organelles and typically appear blue-green under light microscopy owing to the phycocyanin pigment. They lack any flagellated structures and devoid of sexual reproduction. Members of this class are structurally simple, reproducing mainly through binary fission or vegetative means.

Identification is based on colony shape, presence of mucilage, filament structure, and the formation of heterocysts (Komárek & Anagnostidis, 2005; Komárek, 2010; Wehr et al., 2015).



Some filamentous cyanophycean possess two types of specialized cells: heterocysts and akinetes. This is characteristic of genera such as *Anabaena* and *Nostoc* (Fogg et al., 1973; Mur et al., 1999). Heterocysts, with their thick walls and transparent cytoplasm, are specialized for atmospheric nitrogen fixation, giving these organisms a competitive advantage in nitrogen-deficient environments. Akinetes are durable, reserve-rich spores that, under favorable conditions, can detach and give rise to new filaments (Bourrelly, 1985).

Several taxa within Cyanophyceae are known toxin producers and exhibit strong responses to thermal stratification and nutrient enrichment. Genera such as *Microcystis*, *Anabaena* (now *Dolichospermum*), and *Planktothrix* are frequently associated with harmful algal blooms (HABs) in warm, eutrophic reservoirs (Paerl & Otten, 2013). Cyanophyceae often dominate eutrophic freshwater bodies due to their capacity to fix atmospheric nitrogen and their resilience to extreme environmental conditions, including high temperatures, intense light, and nutrient imbalances. Their ability to regulate buoyancy through gas vesicles enables vertical migration and facilitates bloom formation. In general, Cyanophyceae preferentially assimilate nitrogen in the form of ammonium (N-NH_4^+), whereas nitrate (N-NO_3^-) is the preferred nitrogen source for most eukaryotic phytoplankton (Blomqvist et al., 1994; Bernard, 2014).

Prochlorophyceae, a small group closely related to Cyanophyceae, are oxygenic photoautotrophs that possess both chlorophyll *a* and *b*, but lack phycobiliproteins, a key feature that distinguishes them from typical Cyanophyceae (Lewin, 2002). These microorganisms are non-flagellated, and their vegetative structures can be unicellular, colonial, or filamentous. Reproduction occurs primarily via binary fission, and in filamentous forms, through fragmentation (De Reviere, 2007). Some species exhibit structural defenses against zooplankton predation, such as colony formation or filamentous morphology (Lampert, 1987; Bouvy et al., 2001).

Although primarily marine, *Prochloron*-like organisms have occasionally been reported in freshwater systems, where they may contribute to primary production in low-light, oligotrophic environments. However, their presence in reservoirs is rare and poorly documented. A milestone in marine microbiology was the discovery of *Prochlorococcus marinus*, a picoplanktonic cyanobacterium found in oligotrophic oceanic waters and now recognized as one of the most abundant photosynthetic organisms in the ocean. While Lewin (1977) first described *Prochloron*, a symbiotic prokaryote with similar pigments, *Prochlorococcus* was first identified as a free-living planktonic organism by Chisholm et al. (1988). Although Prochlorophyta are less common in freshwater habitats, they display similar ecological strategies like Cyanophyceae.

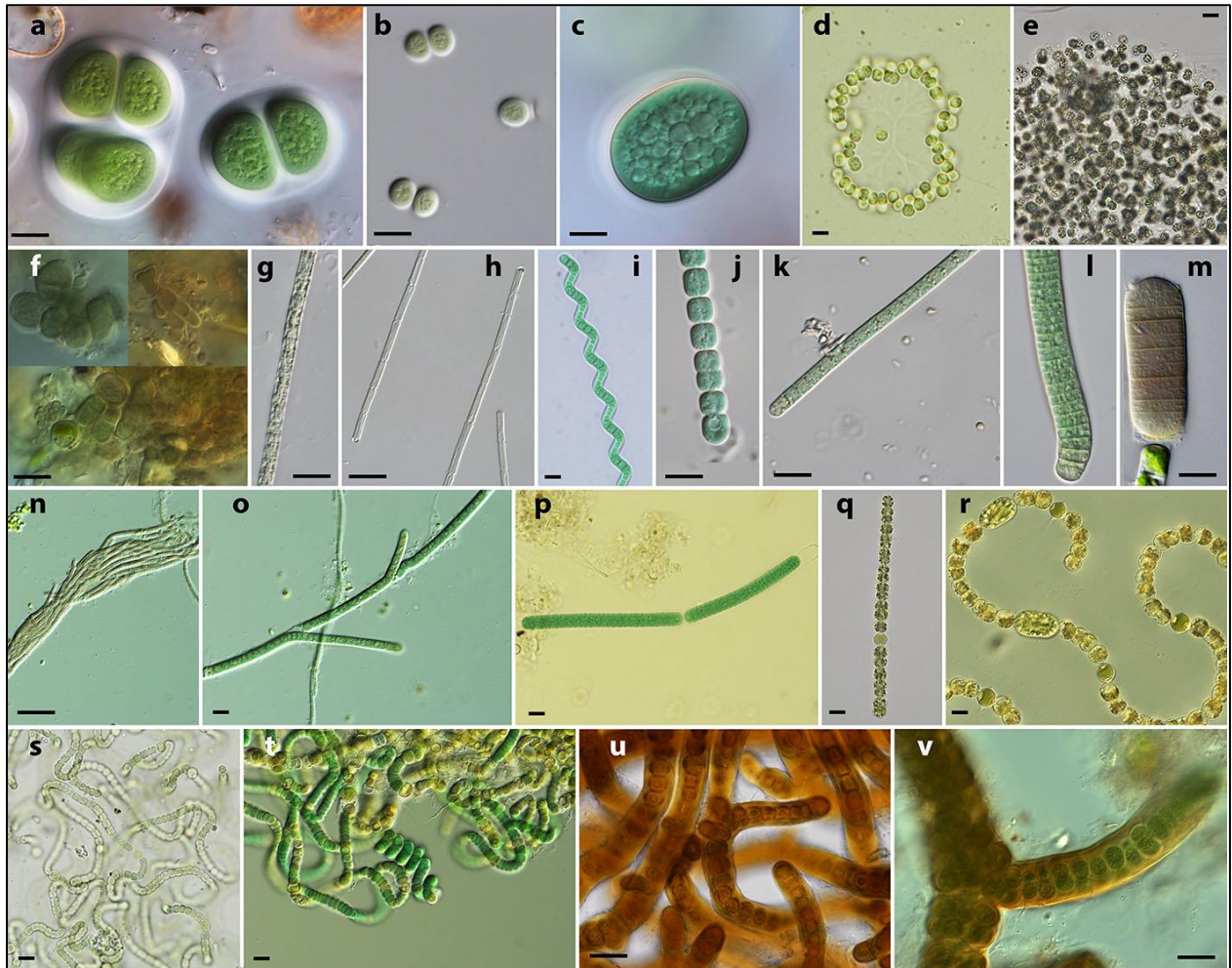


Fig. 4 Cyanophyceae morphological diversity. Groups (orders):

I. Chroococcales: **a** *Chroococcus subnudus*, **b** *Ch. limneticus*, **c** *Cyanothece aeruginosa*, **d** *Snowella litoralis*, **e** *Microcystis aeruginosa*. **II. Pleurocapsales:** **f** *Pleurocapsa minor*. **III. Oscillatoriales:** **g** *Planktothrix agardhii*, **h** *Limnothrix redekei*, **i** *Arthrospira jenneri*, **j** *Johanseninema constrictum*, **k** *Phormidium* sp., **l, m** *Oscillatoria* sp., **n** *Schizothrix* sp., **o** *Tolypothrix* sp., **p** *Katagnymene accurata*. **IV. Nostocales:** **q** *Dolichospermum planctonicum*, **r** *Dolichospermum* sp., **s** *Nostoc* sp., **t** *Nodularia moravica*. **V. Stigonematales:** **u, v** *Stigonema* sp. Scale bar a–u = 10µm, v = 20µm (Dvořák et al., 2015).

1.3.2.2. Chlorophyceae

Chlorophytes, or green microalgae, are eukaryotic microalgae characterized by the presence of chlorophyll *a* and *b* as major pigments, true nuclei, cellulose cell walls, and starch as the primary storage product, formed within the chloroplast (Menzel, 1994; Lee, 2008; Domozych et al., 2012). They can be unicellular (*Chlorella*, *Scenedesmus*), colonial (*Volvox*), or filamentous (*Oedogonium*) (Fig. 5). Motile cells possess equal-length, whiplash-type flagella (commonly two or four) emerging from the anterior end of the cell (John et al., 2002; Turmel et al., 2008). Members of the Chlorophyceae exhibit sexual reproduction, ranging from isogamy to advanced oogamy (Friedl & Rybalka, 2012).



The green coloration of Chlorophyceae, symmetrical morphology, and distinct chloroplast configurations are key diagnostic features. Cellular arrangements (e.g., linear chains, tetrads) and cell wall ornamentation aid in genus-level identification (Graham et al., 2013; Lenarczyk & Wołowski, 2016).

Green microalgae are widespread and very diversified group, particularly thriving under mesotrophic to eutrophic conditions. They exhibit broad ecological plasticity, occupying both planktonic and benthic niches. Overall, Chlorophyceae are typically thermophilic, photophilic microorganisms that prefer environments rich in nitrogenous nutrients (Sane, 2017). Genera such as *Scenedesmus*, *Pediastrum*, and *Chlorella* are commonly found in nutrient-enriched reservoirs (Guiry & Guiry, 2024).

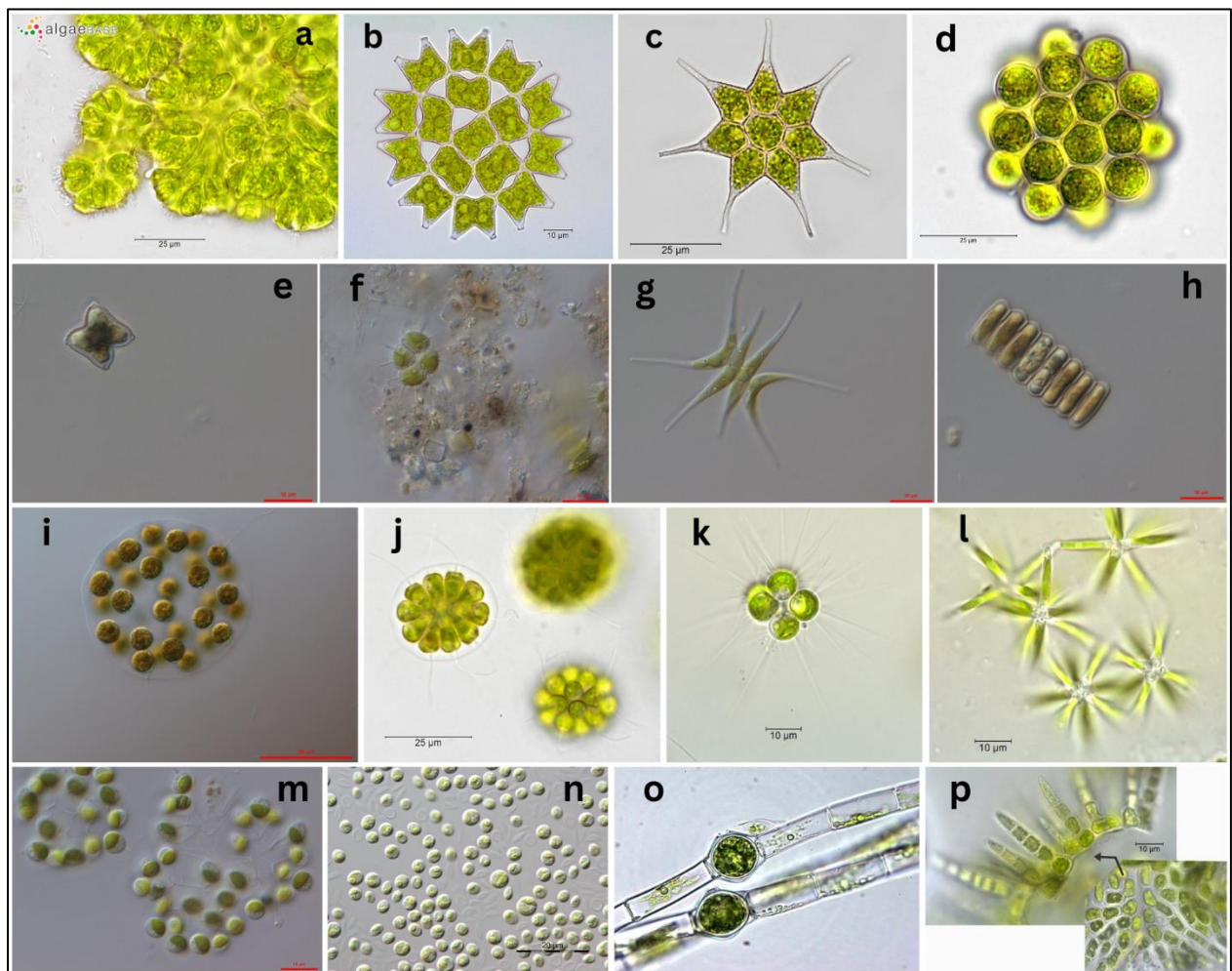


Fig. 5 Chlorophyceae morphological diversity. Groups (orders):

I. Trebouxiales: a *Botryococcus braunii*. **II. Sphaeropleales:** b *Pediastrum duplex*, c *Monactinus simplex*, d *Coelastrum microporum*, e *Tetraëdron minimum*, f *Tetrastrum staurogeniiforme*, g *Tetradesmus lagerheimii*, h *Scenedesmus ellipticus* Corda. **III. Volvocales:** i *Eudorina elegans*, j *Pandorina morum*. **IV. Chlorellales:** k *Micractinium pusillum*, l *Actinastrum hantzschii*, m *Hindakia tetrachotoma*, n *Chlorella vulgaris*. **V. Oedogoniales:** o *Oedogonium braunii*. **VI. Chaetophorales:** p *Stigeoclonium farctum* (Images from AlgaeBase [Guiry & Guiry, 2024]).



1.3.2.3. Zygnematophyceae

Zygnematophyceae is a class within the phylum Charophyta, consisting of conjugating green microalgae that include both filamentous forms and unicellular desmids. The latter traditionally grouped as “*Desmidiophyceae*” but now recognized as a morphological and ecological subgroup within Zygnematophyceae (Brook, 1981; Meesters & Coesel, 2007; Gontcharov, 2008; Wehr et al., 2015; de Araújo et al., 2022) (Fig. 6). Cells possess a rigid cellulose wall and one or more conspicuous chloroplasts containing chlorophylls *a* and *b*, with starch stored in pyrenoids (Brook, 1981; Guiry, 2013; Hall & McCourt, 2015). Filamentous genera such as *Spirogyra*, *Mougeotia*, and *Zygnema* exhibit distinctive chloroplast morphologies, spiral, plate-like, or star-shaped, whereas desmids (e.g., *Cosmarium*, *Closterium*) are typically unicellular, with symmetrical semicells separated by an isthmus and often feature ornate cell wall sculpturing (Meesters & Coesel, 2007; Stancheva et al., 2013).

Members lack flagella at all life stages and reproduce sexually through isogamous conjugation, producing thick-walled zygospores adapted to withstand unfavorable conditions (Mccourt et al., 2000; Gontcharov, 2008; Permann et al., 2021; Grolig, 2024). Asexual reproduction is absent, with population growth occurring through filament or cell fragmentation (Hall & McCourt, 2015).

Zygnematophyceae are widely distributed in freshwater ecosystems, ranging from oligotrophic to eutrophic lakes, ponds, reservoirs, and slow-flowing rivers. Desmids often dominate in slightly acidic, low-conductivity habitats, whereas filamentous taxa can form extensive blooms under stable, low-turbulence, and nutrient-enriched conditions (Reynolds, 2006; Meesters & Coesel, 2007; Stancheva et al., 2013; Hall et al., 2017). Their ecological importance lies in their contribution to primary production, their role as water quality indicators, and their provision of structural habitats for microinvertebrates. However, excessive proliferation, particularly of filamentous species, can lead to oxygen depletion during biomass decay (Reynolds, 2006; Hall & McCourt, 2015; Hall et al., 2017).

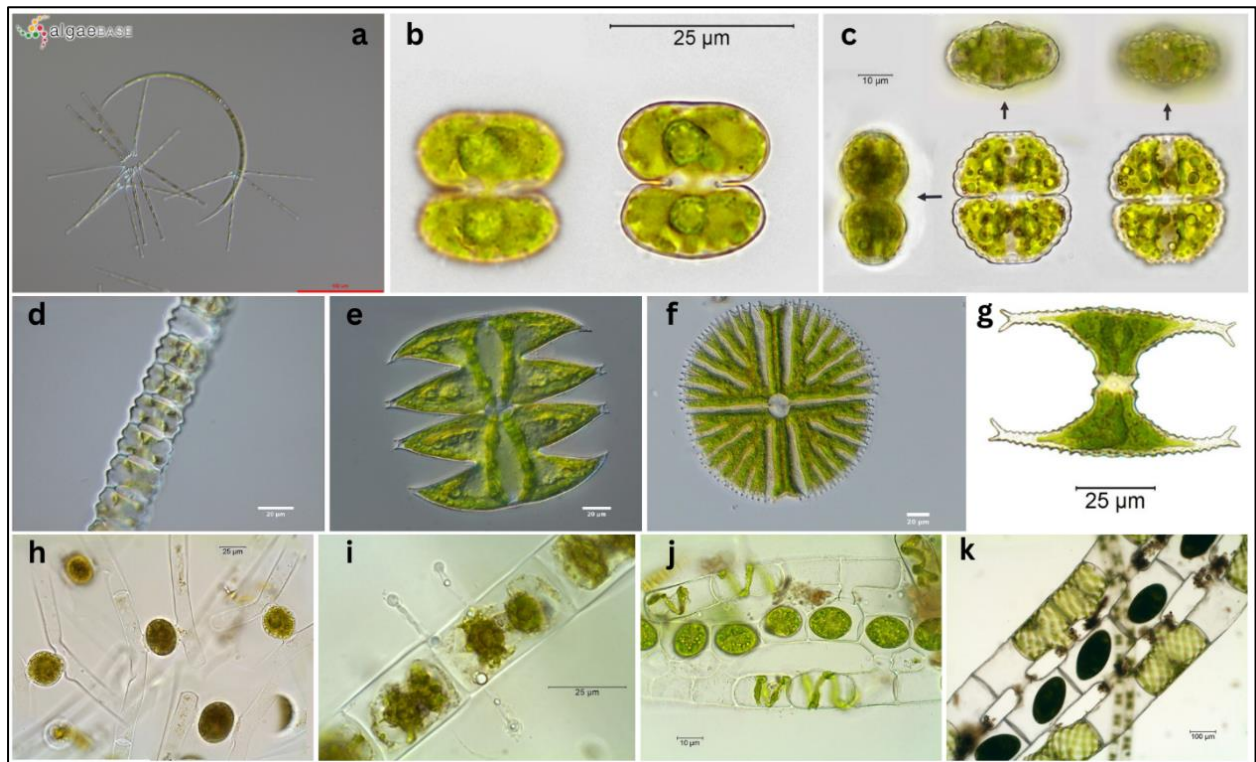


Fig. 6 Zygmatophyceae morphological diversity. Groups (orders):

I. Desmidiaceae: **a** *Closterium acutum*, **b** *Cosmarium bioculatum* var. *depressum*, **c** *Cosmarium subcostatum*, **d** *Desmidium aptogonum*, **e** *Micrasterias laticeps*, **f** *Micrasterias radiosa* var. *elegantior*, **g** *Staurostrum bulbosum*. **II. Zygnematales:** **h** *Mougeotia parvula*, **i** *Zygnema stellinum*. **III. Spirogyrales:** **j** *Spirogyra affinis*, **k** *Spirogyra crassa* (Images from [AlgaeBase](#) [Guiry & Guiry, 2024]).

Within the phylum Chrysophyta, three major algal groups are distinguished, notably **Chrysophyceae**, **Bacillariophyceae**, and **Xanthophyceae**, each exhibiting unique morphological and ecological traits:

1.3.2.4. Chrysophyceae (Golden algae)

Members of this class possess brown to orange-colored chromatophores, which contain one or more accessory pigments, including fucoxanthin (Wehr, 2003; Scoble & Cavalier-Smith, 2014; Wehr et al., 2015). A significant proportion are flagellated and lack a rigid cell wall, giving them a flexible form (Nicholls & Wujek, 2015; Kristiansen & Škaloud, 2017). Motile cells typically bear one or two flagella (occasionally three) inserted at the anterior end. Each cell generally contains one or two parietal chromatophores (Andersen & Wetherbee, 1992; Wehr, 2003; Nicholls & Wujek, 2015; Jo et al., 2016; Bock et al., 2022). In terms of morphology, the most advanced structural form observed is a branched filament (Kristiansen & Škaloud, 2017) (Fig. 7). Sexual reproduction in Chrysophyceae is exceedingly rare and remains poorly understood, with



available observations suggesting only isogamous reproduction (Verma et al., 2015). Asexual reproduction occurs by mitotic division, producing zoospores or autospores, and in colonial forms, by fragmentation (Sandgren, 1988; Nicholls & Wujek, 2015). When present, sexual reproduction involves the fusion of isogametes resembling vegetative cells (particularly in loricate genera), with anisogamy and autogamy reported in *Dinobryon cylindricum* (Nicholls & Wujek, 2015), and meiosis is thought to be zygotic.

This group is widespread in freshwater environments, where they often dominate in oligotrophic lakes, especially under low temperature and low light conditions (Charvet et al., 2012; Remias et al., 2013; Nicholls & Wujek, 2015; Kristiansen & Škaloud, 2017; Schiaffino et al., 2020; Bock et al., 2022). Many species are mixotrophic, capable of both photosynthesis and phagotrophy, which enables survival in nutrient-poor, humic, or acidic environments (Sandgren et al., 1996; Nicholls & Wujek, 2015).

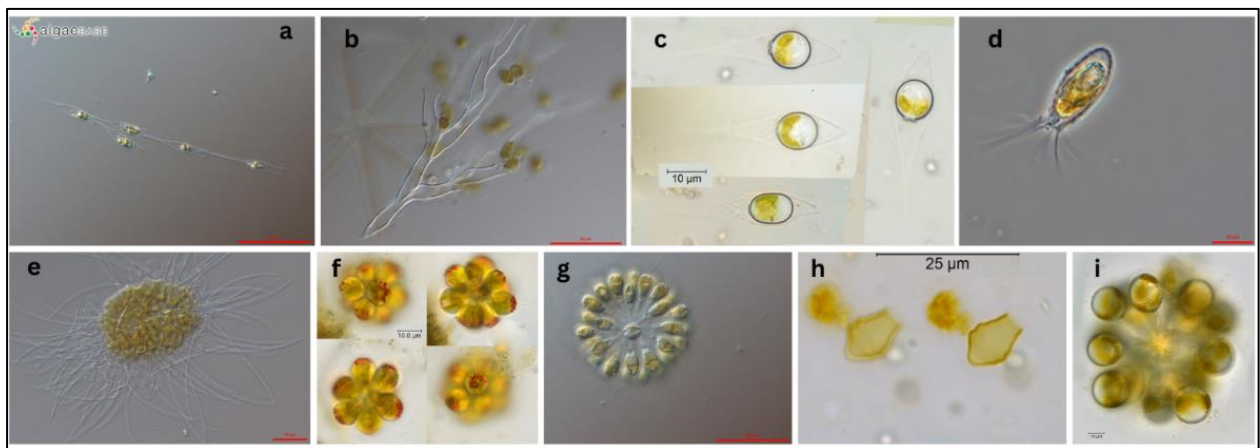


Fig. 7 Chrysophyceae morphological diversity. Groups (orders):

I. Chromulinales: **a** *Dinobryon bavaricum*, **b** *Dinobryon sertularia*. **II. Synurales:** **c** *Mallomonas akrokomu*, **d** *Mallomonas tonsurata*, **e** *Mallomonas caudata*, **f** *Synura sphagnicola*, **g** *Synura uvella*. **III. Chromulinales:** **h** *Pseudokephyrion skujae*. **IV. Ochromonadales:** **i** *Uroglena volvox* (Images from AlgaeBase [Guiry & Guiry, 2024]).

1.3.2.5. Bacillariophyceae (Diatomophyceae)

Diatoms are unicellular or colonial microalgae with silica-based cell walls, known as frustules, composed of two overlapping valves (Medlin, 2009). They exhibit either bilateral symmetry (*Pennales*) or radial symmetry (*Centrales*), with finely detailed surface ornamentation visible under high magnification (Round et al., 1990; Medlin et al., 1996; Townley, 2011) (Fig. 8). Members of this group contain yellow to golden-brown chromatophores, which include accessory



brown pigments such as fucoxanthin (Medlin, 2009; Jones, 2025). In *Pennales*, a unique form of sexual reproduction occurs involving the fusion of protoplasts from vegetative cells. Sexual reproduction is typically isogamous, involving the union of a flagellated and a non-flagellated gamete (Kooistra et al., 2003).

Accurate identification of diatoms requires knowledge of valve morphology, raphe structure, and striae patterns. Common freshwater genera include *Navicula*, *Fragilaria*, and *Cyclotella* (Round et al., 1990; Wehr et al., 2015).

Diatoms are ubiquitous and ecologically dominant in freshwater ecosystems, occupying both planktonic and benthic zones. These siliceous microalgae are abundant in early spring and thrive in well-mixed, nutrient-rich waters (Chaïb et al., 2011; Townley, 2011; Chaïb & Tison-Rosebery, 2012). Their preference for low-light and high-silica conditions makes them particularly suited to deep, oligotrophic to mesotrophic reservoirs (Round et al., 1990; Ziouch et al., 2025).

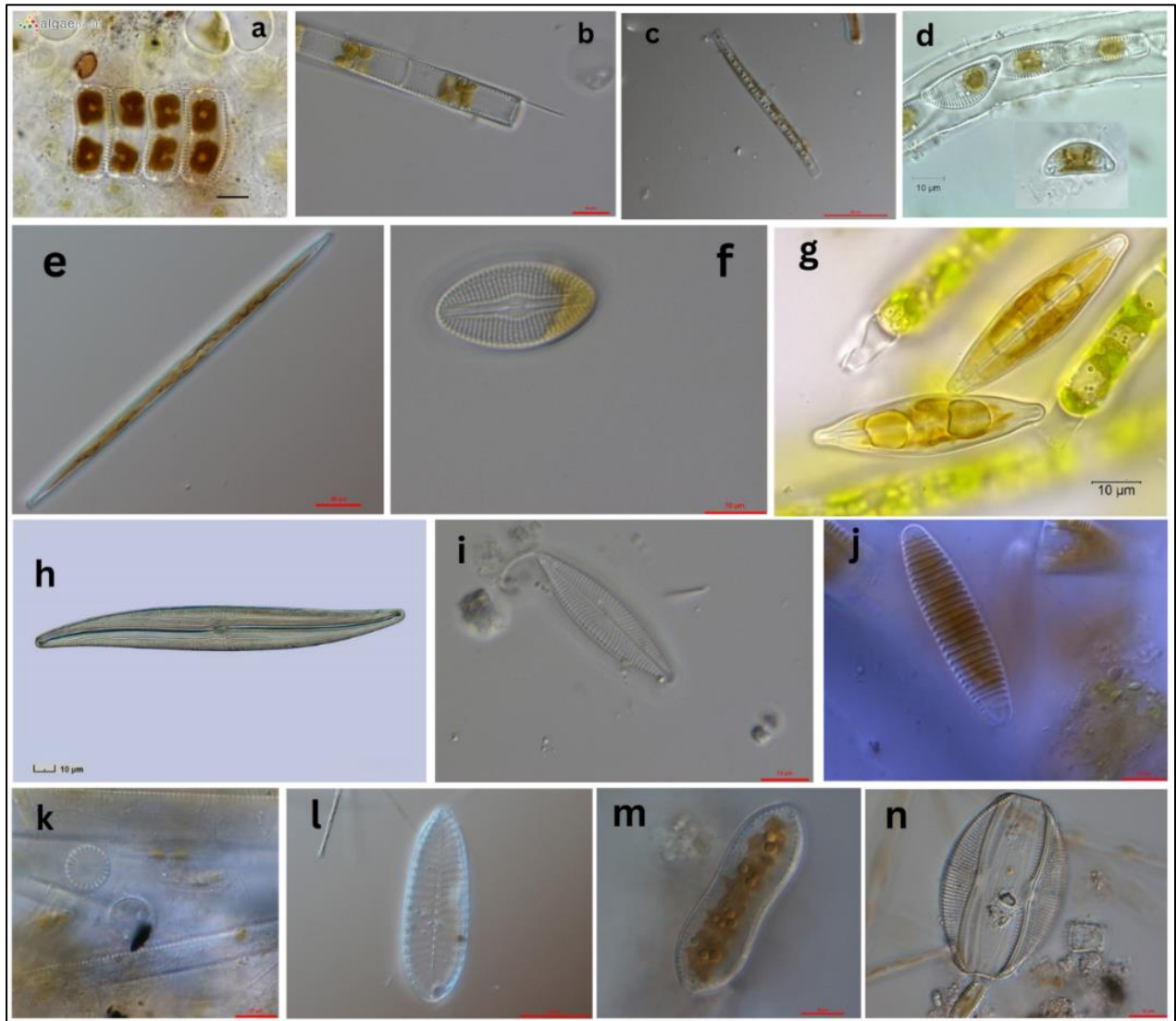


Fig. 8 Bacillariophyceae morphological diversity. Groups (orders):

I. Achnanthales: a *Achnanthes adnata*. **II. Aulacoseirales:** b *Aulacoseira granulata*. **III. Bacillariales:** c *Nitzschia sigmoidea*. **IV. Cymbellales:** d *Encyonema cespitosum*. **V. Fragilariales:** e *Ulnaria ulna*. **VI. Naviculales:** f *Diploneis elliptica*, g *Frustulia saxonica*, h *Gyrosigma attenuatum*, i *Navicula trivialis*. **VII. Rhabdonematales:** j *Diatoma vulgare*. **VIII. Stephanodiscales:** k *Stephanocyclus meneghinianus*. **IX. Surirellales:** l *Iconella tenera*, m *Surirella librile*. **X. Thalassiophysales:** n *Amphora ovalis* (Images from AlgaeBase [Guiry & Guiry, 2024]).

1.3.2.6. Xanthophyceae (Heterokontophyceae)

Xanthophyceae, or yellow-green microalgae, are a class of heterokont flagellates characterized by the presence of xanthophylls as major pigments, along with chlorophyll *a* and *c*, but lacking fucoxanthin, which gives them their yellow-green appearance (Stransky & Hager, 1970; Whittle, 1976; Hibberd, 1981; Maistro et al., 2016) (Fig. 9). The cells may be unicellular, colonial, or filamentous. Cell walls are often composed of two overlapping halves made of cellulose or silica (Rybalka et al., 2009). Most members possess two unequal flagella arising from



the anterior end, one tinsel-type and one smooth (Lokhorst & Star, 2003; Zuccarello & Lokhorst, 2005). Sexual reproduction is isogamous (Stransky & Hager, 1970; Maistro et al., 2016).

Although primarily found in freshwater habitats, their occurrence is usually limited. *Vaucheria* and *Botrydiopsis* are occasional freshwater representatives (Graham et al., 2013). These yellow-green algae are infrequent in freshwater habitats, generally appearing in nutrient-rich benthic environments or as epiphytes on submerged vegetation (Maistro et al., 2016).

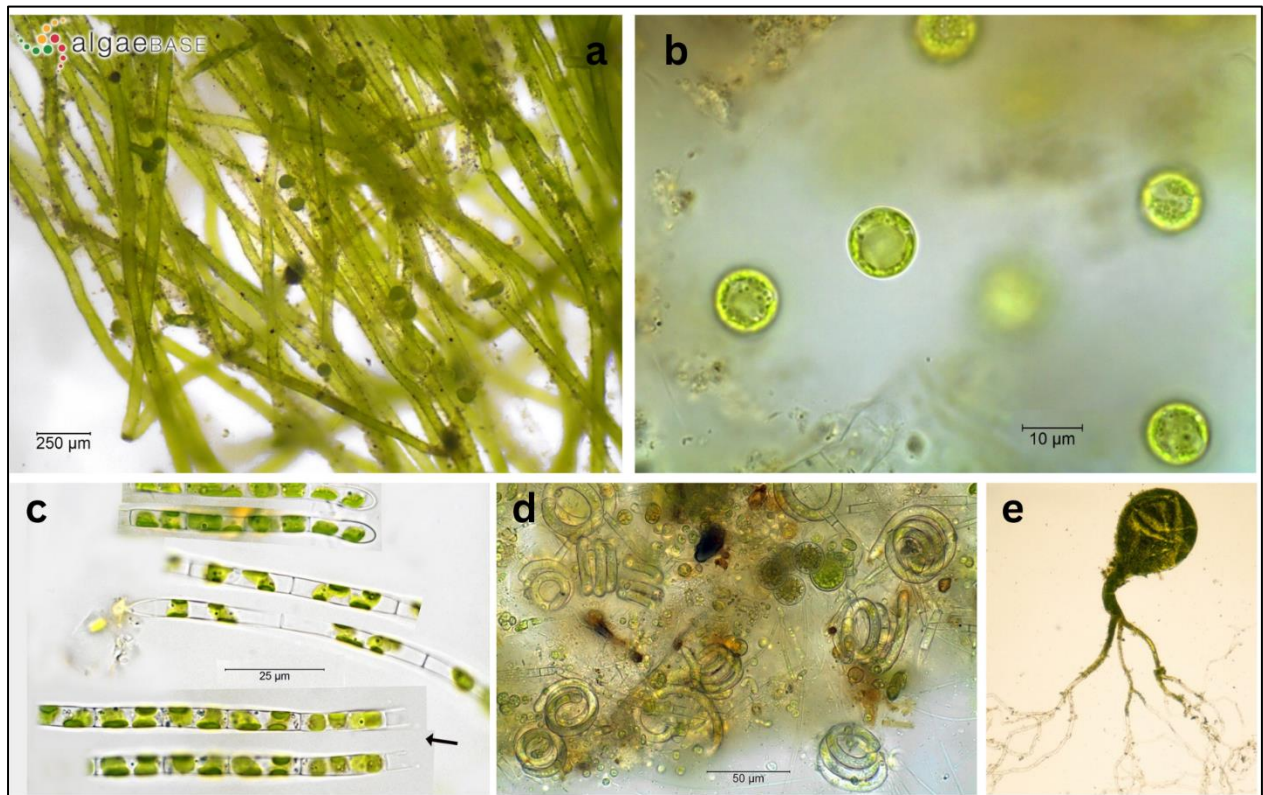


Fig. 9 Xanthophyceae morphological diversity. Groups (orders):

I. Vaucheriales: a *Vaucheria geminata*. **II. Mischococcales:** b *Botrydiopsis arhiza*. **III. Tribonematales:** c *Tribonema minus*. **IV. Mischococcales:** d *Ophiocytium parvulum*. **V. Botrydiales:** e *Botrydium granulatum* (Images from AlgaeBase [Guiry & Guiry, 2024]).

1.3.2.7. Euglenophyceae (Euglenoids)

Members of the Euglenineae are unicellular and characterized by a flexible pellicle surrounding the cell, allowing for shape changes, and their red eyespot (stigma) used for phototaxis (Cavalier-Smith, 1981; Leander et al., 2017; Schwartzbach & Shigeoka, 2017) (Fig. 10). They contain pure green chromatophores, and each cell typically possesses several pyrenoid-like bodies (Triemer & Zakryś, 2015). Chloroplasts in many genera are grass-green, discoid, band-shaped, or stellate, and may be present with or without pyrenoids (Zakryś et al., 2017). Only flagellated forms are known in this group, and most species exhibit active motility via one or two flagella that emerge



from the base of a canal-like invagination at the anterior end (Bicudo & Menezes, 2016). Cells contain a complex vacuolar system, a large prominent nucleus, and one or two contractile vacuoles located at the anterior end. These vacuoles are connected to a reservoir, which in turn opens to the exterior through a narrow gullet (e.g., in *Euglena*) (Schwartzbach & Shigeoka, 2017; Zakryś et al., 2017). Sexual reproduction is rare and not well established; available reports suggest possible isogamy, though these observations remain unconfirmed (Triemer & Zakryś, 2015; Schwartzbach & Shigeoka, 2017).

Euglenoids are frequently associated with eutrophic, organically enriched, or polluted environments (Triemer & Zakryś, 2015). Genera such as *Euglena* and *Phacus* display mixotrophic capabilities, alternating between autotrophic and heterotrophic modes of nutrition, which enhances their adaptability to variable environmental conditions (Brown et al., 2002; Esson & Leander, 2008; Zoltner & Field, 2022).

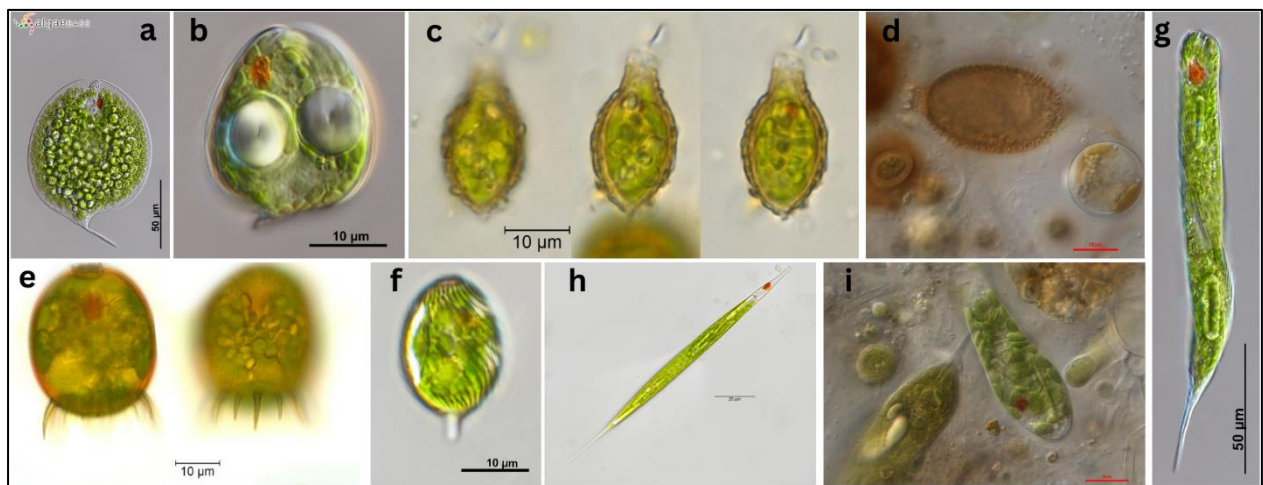


Fig. 10 Euglenophyceae morphological diversity. One group (order):

Euglenales: **a** *Phacus gigas*, **b** *Phacus curvicauda*, **c** *Strombomonas acuminata*, **d** *Trachelomonas hispida*, **e** *Trachelomonas armata*, **f** *Lepocinclis ovum*, **g** *Lepocinclis oxyuris*, **h** *Lepocinclis acus*, **i** *Euglena gracilis* (Images from AlgaeBase [Guiry & Guiry, 2024]).

1.3.2.8. Dinophyceae (Pyrrophyceae)

Dinoflagellates are unicellular, motile protists characterized by two flagella located in perpendicular grooves, one transverse and one longitudinal, producing a distinctive spinning motion (Leadbeater & Dodge, 1967; Wehr et al., 2015; Maruyama et al., 2023). They occur in two structural forms: thecate (armored) with sculptured cellulose plates, and athecate (naked), sometimes exhibiting branched filaments. Cells typically possess discoid chromatophores containing brown to dark-yellow pigments, including peridinin and other accessory pigments (Taylor et al., 2008; Carty & Parrow, 2015) (Fig. 11). Nutrition varies widely: while many are



autotrophic, others are colorless saprophytes or holozoic, and some are parasitic (Taylor et al., 2008). Sexual reproduction, likely isogamous, remains poorly understood and is rarely observed (Carty & Parrow, 2015).

Identification of Dinoflagellates relies on the arrangement of thecal plates, cell symmetry, and flagellar insertion (Carty, 2003; Adl et al., 2012).

Dinoflagellates occur sporadically in freshwater, with some species such as *Ceratium* and *Peridinium* forming blooms in stratified, nutrient-rich reservoirs (Reynolds, 2006; Taylor et al., 2008; Burkholder, 2009). Their ecology is closely linked to thermal stability and light penetration, favoring late summer proliferation.

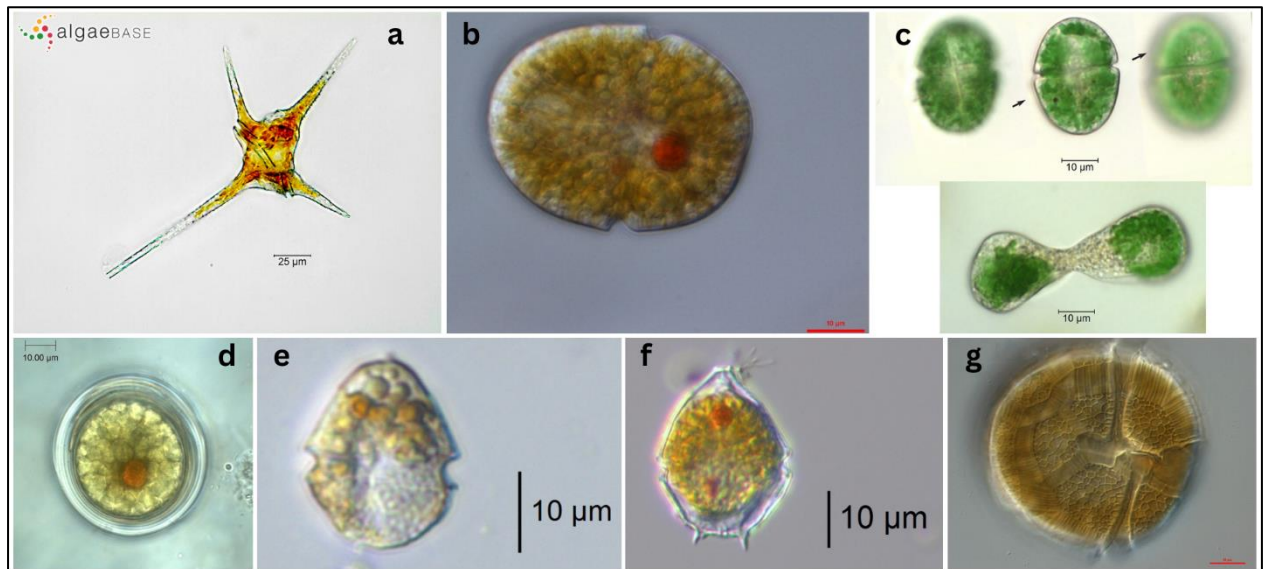


Fig. 11 Dinophyceae morphological diversity. Groups (orders):

I. Gonyaulacales: a *Ceratium hirundinella*. **II. Gymnodiniales:** b *Gymnodinium uberrimum*, c *Nusuttodinium aeruginosum*. **III. Suessiales:** d *Hemidinium nasutum*. **IV. Peridinales:** e *Parvodinium goslaviense*, f *Peridiniopsis quadridens*, g *Peridinium cinctum* (Images from AlgaeBase [Guiry & Guiry, 2024]).

1.3.2.9. Cryptophyceae (Cryptomonads)

Cryptophytes are unicellular, dorsiventrally flattened flagellates with a distinct asymmetry and two slightly unequal flagella. They contain chlorophylls *a* and *c*, as well as phycobiliproteins, which give them a reddish to brownish coloration. Cells typically have two large parietal chromatophores with variable pigmentation and a complex vacuolar system (Kugrens et al., 1986; Fraunholz et al., 1997; Clay, 2015; Hoef-Emden & Archibald, 2016) (Fig. 12). The usual mode of



reproduction is asexual, taking place through mitotic division followed by cytokinesis. Sexual reproduction is oogamous, and the life cycle of some cryptomonads can be quite complex (Hoef-Emden & Melkonian, 2003; Clay, 2015; Hoef-Emden & Archibald, 2016).

A characteristic identification feature is the presence of a gullet and ejectisomes (Clay, 2015; Wehr et al., 2015). Classification of cryptomonads based solely on morphological traits has proven inconsistent (Deane et al., 2002; Hoef-Emden & Melkonian, 2003; Clay, 2015; Hoef-Emden & Archibald, 2016).

This class is relatively small and occurs in both marine and freshwater ecosystems, though often in low abundance (Hill, 1991; Hoef-Emden, 2007; Metfies et al., 2010; Clay, 2015). *Cryptomonas* and *Rhodomonas* are frequently observed in freshwater reservoirs and lakes, particularly in mesotrophic to eutrophic conditions and in cooler, mixed waters (Camacho et al., 2001; Metfies et al., 2010). Their cryptic pigmentation allows efficient light absorption across depths (Hoef-Emden, 2007; Clay, 2015). They are important components of the spring and fall phytoplankton assemblages, often co-dominant with chlorophytes and chrysophytes (Camacho et al., 2001; Hoef-Emden & Archibald, 2016).



Fig. 12 Cryptophyceae morphological diversity. Groups (orders):

I. Cryptomonadales: **a** *Cryptomonas curvata*, **b** *Cryptomonas erosa*, **c** *Cryptomonas ovata*, **d** *Tetrabaena socialis*. **II. Pyrenomonadales:** **e** *Rhodomonas pusilla* (Images from AlgaeBase [Guiry & Guiry, 2024]).



1.3.3. Phytoplankton ecology

Phytoplankton growth in reservoirs is controlled by physical, chemical, and biological factors; however, unlike natural lakes, it is also highly affected by hydrological fluctuations and operational management (Kimmel et al., 1990; Tolotti et al., 2010; Rangel et al., 2012; Fadel et al., 2015). Major influencing drivers include climatic conditions, watershed nutrient inputs, basin morphometry, and the structure of the food web (Naselli-Flores & Barone, 2000; Galizia-Tundisi & Matsumura-Tundisi, 2003; Firsova et al., 2023). Phytoplankton organisms inhabit almost all aquatic environments, including extreme habitats such as hypersaline waters, thermal springs, and polar lakes, and they are more abundant in nutrient-rich inland waters than in the open sea (Padisák & Naselli-Flores, 2021; Calbet, 2024; Pereira, 2025). Diatoms are generally the most abundant group, followed by dinoflagellates and cyanobacteria, of which only about 6% are toxin producers (Caruana & Amzil, 2018; Vincent & Bowler, 2020). Microalgae are ecologically diverse, occurring as planktonic or benthic forms, and some live in symbiosis with animals, plants, or fungi (Villareal, 1994; Couté & Bernard, 2001; Janson, 2002; De Reviers, 2003). They can also attach to birds, crustaceans, or turtles (Bennett, 1920; Croll & Holmes, 1982; Carman & Dobbs, 1997; Majewska et al., 2015; López-Fuerte et al., 2024; Lin et al., 2025).

Phytoplankton species exhibit diverse ecophysiological responses to environmental factors, with variations in growth rates, grazing and nutrient requirements. To cope with such fluctuations, they have evolved a range of adaptive strategies, including (Grogg, 2012):

- Various mechanisms promoting mobility and migration towards areas rich in nutrients and light (phototaxis, vertical migration, swimming, aggregation);
- Mixotrophic modes of nutrition;
- Interspecific competition through the production of allelochemical substances;
- Enzyme production for nutrient acquisition, secretion of chelators (e.g., siderophores);
- Defense mechanisms against predation (allelopathy).

1.3.4. Phytoplankton dynamics

Phytoplankton communities undergo significant changes over the course of a single year, a process widely known as *seasonal succession* (Margalef, 1979; Rolland et al., 2009; Wei et al., 2020). The first conceptual model of this succession in freshwater lakes, known as the PEG model (*Plankton Ecology Group*), was developed after several years of discussion among about thirty plankton ecologists, based on observations from 24 lakes (Sommer et al., 1986). The PEG model describes the seasonal succession of plankton communities in temperate freshwater lakes as a



sequence of ~24 predictable events regulated by nutrient availability, light conditions, grazing pressure, and physical processes (Sommer et al., 1986; Padisák et al., 1993; Scheffer et al., 2003; Sommer et al., 2012). The different stages can be summarized as follows:

- **Late winter:** Vertical mixing replenishes nutrient concentrations in the euphotic zone, and increasing light availability stimulates phytoplankton growth, dominated by diatoms and cryptophytes (Reynolds, 2006; Mariani et al., 2013). Then an intensive grazing by herbivorous zooplankton that reduces phytoplankton biomass, producing the characteristic *clear-water* phase during which nutrients are remineralized (Sommer et al., 1986; Mariani et al., 2013).
- **Early summer:** As grazing pressure declines, a more diverse phytoplankton assemblage develops, comprising both small edible species and large inedible taxa. Chlorophyceae often thrive on the remaining nutrients and resist grazing. Progressive nutrient depletion then favors species adapted to low-nutrient conditions, such as dinoflagellates exploiting hypolimnetic phosphorus, nitrogen-fixing heterocystous cyanobacteria, and silica-limited diatoms (Reynolds et al., 2002; Rolland et al., 2009; Sommer et al., 2012).
- **Mid-late summer:** Community shifts toward species adapted to stable, nutrient-depleted, and warmer conditions, often flagellates or cyanobacteria depending on the trophic status. Cyanobacteria and phytoflagellates typically dominate under stratification, high temperatures, and with the advantage of vertical migration (Reynolds et al., 2002; Sommer et al., 2012; Wei et al., 2020).
- **Autumn:** Increased mixing depth, nutrient enrichment, and declining light availability promote the appearance of large inedible species and autumn diatoms, accompanied by edible forms. A general reduction in primary production follows, leading to the winter biomass minimum. The return of mixed conditions and lower temperatures lead to a sharp decline in cyanobacteria and phytoflagellates, with a resurgence of diatoms (Sommer, 1991).

While widely applied, the PEG model does not account for all processes influencing phytoplankton dynamics, notably physical destabilization of the water column (Pannard et al., 2007; Pannard et al., 2008; Paparella & Vichi, 2020), microbial loop-mediated nutrient regeneration (Pomeroy et al., 2000; Li et al., 2014), and the effects of viral lysis or parasitism (Suttle, 2000; Brussaard, 2004; Gleason et al., 2008; Sommer et al., 2012), despite the recognized ecological significance of these interactions. Reynolds and colleagues emphasized that succession



patterns are strongly shaped by physical drivers (mixing regimes, stratification depth, light climate) interacting with species-specific adaptive traits such as buoyancy regulation, motility, and nutrient storage capacity (Reynolds, 1988; 2006).

1.3.5. Bioindicative role of phytoplankton

Phytoplankton, being sensitive to environmental changes, serve as excellent indicators of water quality. They are widely recognized as reliable bioindicators due to their short life cycles, rapid response to environmental fluctuations, and strong dependence on water chemistry and light regimes (Salmaso et al., 2006; Anneville et al., 2008; Yang et al., 2012; Sahoo et al., 2024; Tokatli et al., 2024). The composition and dominance patterns of phytoplankton communities reflect trophic status (from oligotrophic to hypereutrophic), mixing conditions, and nutrient availability, particularly nitrogen and phosphorus (OECD, 1982; Padisák et al., 2006; Bhagat et al., 2025). Their community composition, species richness, and the presence or absence of sensitive taxa help assess the ecological status of aquatic ecosystems (Blandin, 1986; Belin & Daniel, 2013).

Specific taxa serve as indicator species: for example, the dominance of *Microcystis* or *Dolichospermum* may signal nutrient over-enrichment and thermal stability, whereas *Cyclotella* or *Asterionella* suggest better oxygenation and silica-rich environments (Reynolds et al., 2002). Certain microalgal species also exhibit high tolerance to pollution, including *Euglena viridis*, *Nitzschia palea*, *Synedra ulna*, *Scenedesmus quadricauda*, *Stigeoclonium tenue*, *Ankistrodesmus falcatus*, *Pandorina morum*, *Oscillatoria limosa*, *O. tenuis*, and *O. chlorina* (Sahoo et al., 2024). Phytoplankton metrics (e.g., chlorophyll-a concentration, diversity indices, functional group classification) are also integral to ecological status assessments under frameworks such as the Water Framework Directive (WFD) (European Commission, 2000). Their role in water quality monitoring is particularly valuable in reservoirs, where rapid changes in nutrient input, flow regimes, and temperature can cause bloom formation or community shifts (Lyche-Solheim et al., 2024). Reynolds et al. (2002) identified **31 phytoplankton assemblages** grouped into **functional groups**, reflecting species' responses to abiotic factors such as temperature, light, nutrients, and predation. Phytoplankton is often the first biological component to respond to eutrophication, especially in lake environments (Padisák et al., 2006; Lyche-Solheim et al., 2024). Recognizing this role, the Water Framework Directive (European Commission, 2000) classified phytoplankton as a key element of biological quality in lakes. Their proliferation can alter ecological dynamics and, in some cases (e.g., cyanobacteria), release toxins that threaten human and animal health (Paerl & Otten, 2013; Chandel et al., 2024).



1.3.6. Functional role of phytoplankton

1.3.6.1. Nutrients cycling

Functionally, phytoplankton drive primary production and play a central role in biogeochemical cycling within freshwater ecosystems. They regulate carbon and nutrient dynamics not only through uptake but also via sedimentation and release processes during decomposition (Pal & Choudhury, 2014). By assimilating inorganic nutrients such as nitrogen, phosphorus, and silica, phytoplankton convert them into organic matter through photosynthesis, thereby forming the base of the aquatic food web. This organic matter is subsequently respired and remineralized into inorganic forms by bacteria, archaea, and zooplankton (Robinson, 2017). The resulting biomass sustains higher trophic levels while recycling nutrients back into the water column through excretion, decomposition, and sedimentation, thus maintaining ecosystem productivity and balance (Reynolds, 2006; Cloern et al., 2014). Their photosynthetic activity also modulates oxygen dynamics, contributing to oxygen supersaturation during daylight hours and influencing vertical DO profiles in stratified reservoirs (Wetzel, 2001). Owing to their rapid turnover and sensitivity to environmental fluctuations, phytoplankton act as key regulators of nutrient availability and reliable indicators of ecosystem status (Falkowski & Raven, 2013). In freshwater ecosystems, phosphorus is generally the principal limiting nutrient (Wagner, 2010; Dillon & Molot, 2024). Phytoplankton mainly assimilate orthophosphate, which is often rapidly depleted in the epilimnion during growth periods (Wetzel, 2001). Under phosphorus-deficient conditions, many algae and cyanobacteria produce extracellular phosphatases to mobilize organically bound phosphorus, although the broader ecological implications of this mechanism remain insufficiently understood (Štrojsová et al., 2003; Dillon & Molot, 2024).

1.3.6.2. Trophic food web

As the first trophic level, they provide the base energy source for zooplankton, filter feeders, and higher consumers such as fish, by converting solar energy into organic matter (Beaver et al., 2018; Shcherbak et al., 2024). This energy is transferred through successive trophic levels, including fish and other predators, making phytoplankton essential for sustaining aquatic biodiversity and productivity (Reynolds, 2006; Molner et al., 2025). Their composition and abundance directly influence food web structure and energy flow in freshwater and marine ecosystems (Gaedke, 2009; Winder & Sommer, 2012). The efficiency of energy transfer from phytoplankton to higher trophic levels varies, with factors such as nutrient availability and species composition playing significant roles (Katechakis, 2005; Paerl et al., 2024). Certain taxa exhibit



allelopathic effects or exude compounds affecting zooplankton grazing and microbial communities, thereby playing a regulatory role in community structure and food web stability (Legrand et al., 2003; Katechakis, 2005; Reynolds, 2006). Human-induced nutrient enrichment can lead to shifts in phytoplankton composition, potentially resulting in harmful algal blooms that disrupt ecosystem health (Shcherbak, 2024). While phytoplankton are essential for sustaining aquatic biodiversity, their overabundance due to nutrient pollution can lead to detrimental effects on water quality and ecosystem health, highlighting the need for balanced nutrient management in aquatic ecosystems.

1.3.7. Phytoplankton blooms

Phytoplankton blooms refer to a rapid proliferation or accumulation of algae within aquatic ecosystems. These blooms are primarily composed of various planktonic genera and typically occur when phytoplankton populations grow in distinct pulses. A bloom is generally identified when cell concentrations surpass the annual mean or background levels by several times, or when they reach a defined threshold, commonly around 5×10^6 cells L^{-1} . Such blooms may also visibly alter the color of the water (Wetzel, 2001).

In temperate regions, the first notable bloom usually takes place in spring, stimulated by increasing sunlight availability. This growth phase is typically curtailed in autumn due to reduced light penetration. In contrast, tropical ecosystems may experience near-continuous phytoplankton growth throughout the year, provided that nutrients are sufficient (Reynolds, 2006; González-Gil et al., 2022). In eutrophic temperate lakes, holoplanktonic algae, those consistently existing in planktonic form, often dominate during spring. Prominent genera such as *Asterionella*, *Fragilaria*, and *Tabellaria* contribute significantly to these spring blooms. However, several biotic factors such as chytrid fungal infections, grazing by zooplankton, or protozoan predation can suppress bloom development (Sommer et al., 2012).

Meroplanktonic species like *Melosira*, which exist only intermittently in the plankton, are often observed in high abundance during winter. Their bloom is typically a result of the resuspension of viable resting spores from sediment layers during favorable conditions. Despite their relatively slow growth, *Melosira* species can thrive under nutrient-rich, low-competition, and low-grazing environments (Reynolds, 2006).

Cyanobacterial genera such as *Aphanizomenon*, *Dolichospermum* (*Anabaena*), and *Microcystis*, also classified as meroplanktonic, commonly dominate summer and fall



phytoplankton communities in warm lakes. These taxa often form resting cells during winter to survive unfavorable conditions. Cyanobacteria also possess gas vesicles that allow buoyancy regulation, enabling them to adapt to different depths across seasons depending on thermocline positioning (Paerl & Otten, 2013). Furthermore, many cyanobacteria and certain eukaryotic algae are highly efficient in absorbing nutrients such as phosphate and ammonia even at low concentrations. Some cyanobacterial species also have the ability to fix atmospheric nitrogen; a trait absent in most other algal groups. Consequently, cyanobacterial blooms often emerge even in nitrogen-depleted environments. Among phytoplankton groups, cyanobacteria and dinoflagellates are especially notable for frequently forming blooms under diverse ecological conditions and are ecologically significant due to their impacts on nutrient cycling and aquatic food webs (Reynolds, 2006; Paerl et al., 2024).

1.3.8. Historical overview of phytoplankton research in Algeria

Study of freshwater algae and phytoplankton in Algeria has a long but fragmented history. Early investigations date back to the late 18th and 19th centuries, beginning with Desfontaines (1799), followed by Montagne (1846) and Ehrenberg (1854), who provided some of the first descriptions of the algal flora of the region (Baudrimont, 1971; Nehar et al., 2015). A major milestone was the work of Baudrimont (1974), who documented 356 species, varieties, and forms of diatoms from inland waters, estimating that this represented only 17.5% of the world's known flora at the time. His work remains a cornerstone in the study of Algerian freshwater diatoms.

In the following decades, research became more targeted, focusing on specific groups and aquatic ecosystems. Significant contributions include the taxonomic and ecological studies of Chaïb et al. (2011), Chaïb & Tison-Rosebery (2012), and Nehar et al. (2015), which enriched the Algerian diatom database by reporting new taxa from El-Hammam stream and the estuary of the Cheliff River. More recent studies have broadened knowledge of phytoplankton diversity in Algerian wetlands and reservoirs: El Haouati et al. (2015) recorded 24 species in Lake Reghaia, Chabaca et al. (2020) reported 18 genera in Lake Oubeira, and Heramza et al. (2021) and Negadi et al. (2021) described diatom assemblages from Chott-Chergui and Ain Dalia Dam.

Research on Cyanophyceae has also expanded considerably, with studies by Bensouilah et al. (2003), Nasri et al. (2003, 2008), Soumati et al. (2005), Souissi et al. (2004), Amri et al. (2010), and Ouarts et al. (2011) focusing mainly on northeastern Algeria, particularly Lake Oubeira. Similarly, Bidi-Akli et al. (2017) investigated cyanobacterial communities and their impact on water quality.



In recent years, phytoplankton research in Algerian reservoirs and lakes has increasingly addressed ecological questions, including the effects of physicochemical parameters and heavy metals on community structure (Hamidi et al., 2009; Boussadia, 2017; Djabourabi et al., 2017; Khereif et al., 2018; Arab et al., 2019; Draredja et al., 2019). Studies integrating taxonomy with water quality monitoring have contributed to understanding biodiversity patterns and eutrophication processes (Sehili et al., 2020; Heramza et al., 2021; Djerboua et al., 2022; Arif et al., 2023 a, b; Belhaoues et al., 2024; Hammana et al., 2024).

Despite these advances, many Algerian aquatic ecosystems remain poorly explored, and comprehensive ecological assessments integrating phytoplankton diversity, abundance, and functional composition are still limited.



I.4. Zooplankton communities

Zooplankton are microscopic, heterotrophic organisms that inhabit aquatic environments and drift passively with water currents, although many possess limited mobility through appendages such as antennae or swimming setae. The term “zooplankton” derives from the Greek words *zoon* (ζῷον = animal) and *planktos* (πλανκτός = wanderer or drifter), and, like phytoplankton, they were included in the broader category of “plankton” defined by Victor Hensen in 1887 to describe organisms suspended in the open water column, independent of the bottom or shoreline (Ruttner, 1953; Hutchinson, 1971; Wetzel, 2001; Santhanam et al., 2018).

Zooplankton occupy a central position in aquatic food webs, functioning as primary consumers that link phytoplankton and microbial production to higher trophic levels such as fish. They influence energy transfer, regulate algal biomass through grazing, and contribute to nutrient recycling via excretion, decomposition, and sedimentation (Sterner, 2009; Ger et al., 2016; Bakhtiyar et al., 2020). Zooplankton diversity and distribution are regulated by abiotic factors such as temperature, salinity, dissolved oxygen, nutrient availability, and hydrodynamics, as well as by biotic interactions such as grazing and fish predation (Gliwicz, 2003; Kehayias et al., 2008; Błędzki & Rybak, 2016; Rosińska et al., 2019; Baitiche et al., 2025). Consequently, their community composition, abundance, and seasonal dynamics are tightly coupled with environmental conditions and predation pressure, making them reliable indicators of ecosystem functioning and trophic state (Pontin, 1978; Dumont, 1994). In reservoirs, zooplankton communities are highly dynamic, often reflecting both natural seasonal variability and anthropogenic impacts on water quality and habitat conditions.

1.4.1. Evolution of zooplankton taxonomy

The classification of zooplankton has evolved considerably, shifting from morphology-based systems to approaches integrating ultrastructural, ecological, and molecular data. Early taxonomic frameworks were largely descriptive, relying on visible traits such as body form, segmentation, and appendages. Foundational works by Sars (1903a; 1903b), Apstein (1907; 1910), and Edmondson (1959) provided the first comprehensive morphological descriptions of cladocerans, copepods, and rotifers, establishing the basis for freshwater zooplankton taxonomy.

During the mid-20th century, syntheses by Hutchinson (1967) and Ruttner (1975) emphasized ecological and functional perspectives, broadly grouping freshwater zooplankton into Rotifera, Cladocera, and Copepoda as the dominant representatives. Around the same period,



Smirnov (1976; 1984) and Pontin (1978) refined morphological classifications and developed ecological identification keys, incorporating biogeographical and adaptive traits.

Further advancements were made by Fernando (1980a; 1980b), who analyzed zooplankton communities across tropical freshwater habitats, and by Dumont et al. (1981), Dumont (1983), and Dumont and Tundisi (1984), who formalized ecological and biogeographical frameworks for freshwater zooplankton.

In the following decades, the application of advanced microscopy and ultrastructural studies, particularly those by Korovchinsky (1992) and Wallace & Snell (2001), allowed finer taxonomic resolution within Cladocera and Rotifera, where features such as lorica morphology and reproductive modes became key diagnostic characters.

More recently, molecular phylogenetics has transformed zooplankton taxonomy by revealing cryptic diversity and evolutionary relationships not evident from morphology alone (Schwenk et al., 2005; Fontaneto et al., 2009). DNA barcoding and genomic approaches have uncovered high levels of hidden diversity in rotifers, copepods, and cladocerans, reshaping the understanding of species boundaries and evolutionary lineages.

Today, zooplankton taxonomy reflects an integrated approach that combines morphological, ecological, and molecular evidence, aligning it with broader advances in evolutionary biology and systematics.

1.4.2. Components of freshwater zooplankton

Reservoirs usually have zooplankton composed mainly of two major taxonomic groups: **rotifers** and **microcrustaceans**, which include **cladocerans** and **copepods** (Armengol, 1982; Dussart et al., 1984; Armengol & Miracle, 1999; Picapedra et al., 2019). The ubiquity and abundance of rotifers, cladocerans, and copepods explain why these three groups are the main components of freshwater zooplankton in limnological studies (Cabral et al., 2019; Picapedra et al., 2019; Bakhtiyar et al., 2020; Muñoz-Colmenares et al., 2021). A basic understanding of essential anatomical terminology is required to accurately identify these organisms.



1.4.2.1. Rotifera

Rotifers are microscopic unsegmented, bilaterally symmetrical invertebrates that play a vital ecological role in freshwater ecosystems, as primary consumers in planktonic food webs and as effective bioindicators. They are multicellular and range in size from about 50 to 2000 μm . They are widespread filter feeders of algae, bacteria, and detritus (Wallace & Snell, 2001; Segers, 2007; Rico-Martínez et al., 2016; Shumka et al., 2018). Commonly known as "**wheel animals**," rotifers are distinguished by their ciliated corona used for locomotion and feeding. The name "**rotifer**" derives from the rotating appearance of the **cilia** on the corona. Their digestive system typically includes a complex jaw apparatus (**trophi**) used to seize and grind food particles. Trophi morphology varies among species and is often used for taxonomic identification (Guerrero-Jiménez et al., 2013) (Fig. 13).

About three-quarters of rotifer species are sessile, living attached to littoral substrates, while roughly 100 species are entirely planktonic (Wetzel, 2001; Rico-Martínez et al., 2016). Rotifers are considered opportunistic due to their high tolerance to environmental changes and rapid reproductive rates (Allan, 1976; Rico-Martínez et al., 2016). Their feeding behavior is diverse, rotifers may act as generalist or specialist feeders depending on the morphology of their buccal apparatus (Sarma & Nandini, 2019; Gilbert, 2022), and they consume bacteria, heterotrophic flagellates, and small ciliates (Arndt, 1993). In particular, bacteria and phytoplankton are significant components of their diet in reservoir environments (Rico-Martínez et al., 2016). Additionally, trait-based studies categorizing rotifers by trophi type and feeding strategies have advanced our understanding of their functional roles in aquatic ecosystems (Obertegger & Wallace, 2023). Rotifers are now widely recognized as sensitive indicators of trophic state, responding swiftly to changes in nutrient levels, pollution, and habitat conditions, making them invaluable for monitoring freshwater quality, especially in small water bodies (Kuczyńska-Kippen et al., 2025).

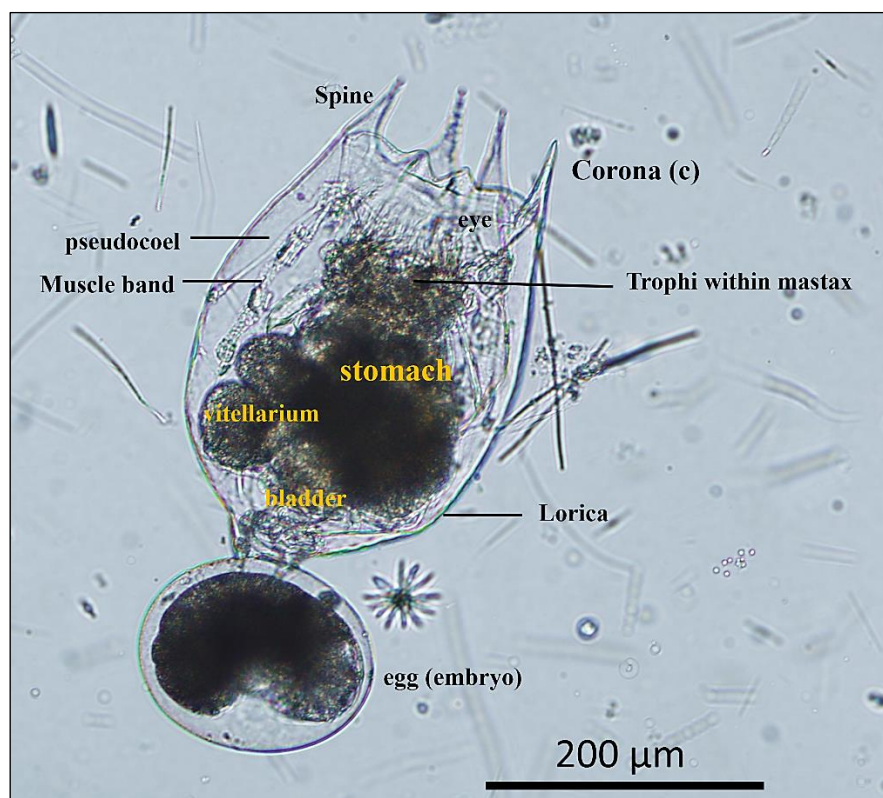


Fig. 13 Dorsal view of *Brachionus calyciflorus*.

1.4.2.2. Microcrustaceans

In the freshwater ecosystems considered in this study (reservoirs), microcrustaceans are mainly composed of cladocerans and copepods. They inhabit a wide range of aquatic environments, including leaf litter, mosses, springs, groundwater, and even caves (Błędzki & Rybak, 2016).

1.4.2.2.1. Cladocerans

Cladocerans, or **water fleas**, are small freshwater microcrustaceans (0.2–3.0 mm). They are primarily algal filter feeders and serve as key prey for visual predators (Balcer et al., 1984; Błędzki & Rybak, 2016). They are distinguished by a bivalve carapace, compound eye, and large second antennae used for swimming, with thoracic appendages generating feeding currents (Fig. 14). Most species reproduce parthenogenetically, with eggs developing in a brood chamber and juveniles resembling adults, though sexual reproduction occurs periodically. Cladocerans also display pronounced seasonal polymorphism and cyclomorphosis, which are important adaptive traits, mostly confined to epilimnetic species or to those that migrate there daily, influencing both buoyancy and predator avoidance (Jacobs, 1977; Wetzel, 2001; Błędzki & Rybak, 2016).



Cladocerans are widely distributed in freshwater reservoirs, inhabiting both the open water (pelagic zone) and littoral regions where vegetation provides shelter (Forró et al., 2007; Lampert & Sommer, 2007). They are especially abundant in the epilimnion, where food resources such as phytoplankton and bacteria are concentrated (Gliwicz, 2003; Karuthapandi & Rao, 2016). *Daphnia* often dominates open waters, while smaller taxa such as *Bosmina* and *Ceriodaphnia* occur nearshore, with vertical migrations allowing them to exploit resources and evade predators (Gliwicz, 2003; Lampert, 2006; Forró et al., 2007; Lampert & Sommer, 2007; Karuthapandi & Rao, 2016). Thus, cladocerans occupy diverse habitats within reservoirs, reflecting both resource availability and ecological pressures.

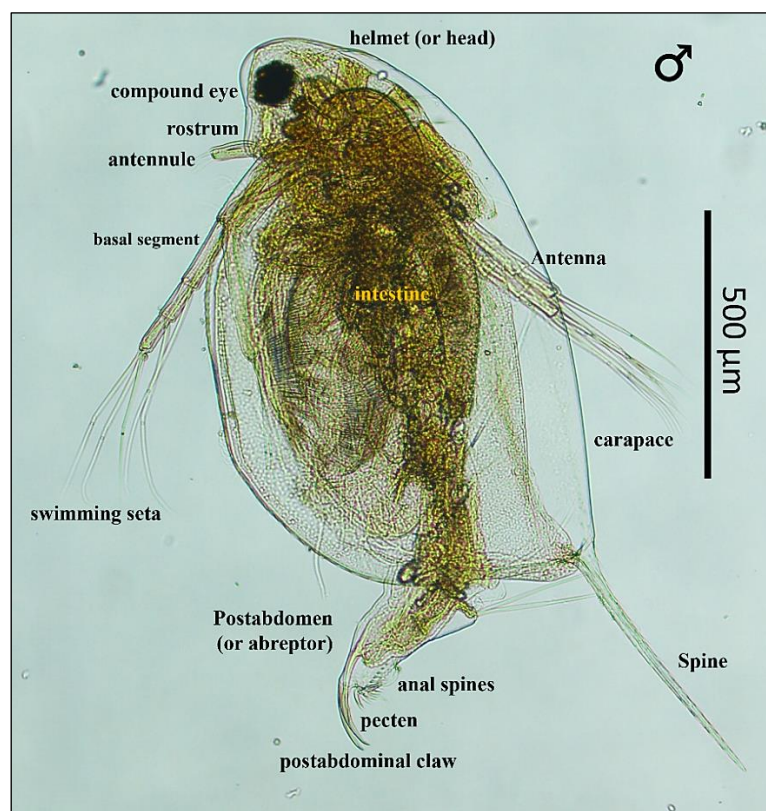


Fig. 14 Lateral view of a *Daphnia longispina*.

1.4.2.2.2. Copepods

Copepods, commonly referred to as **oarsmen**, range in size from 0.3 to 3.2 mm. In freshwater reservoirs, the most common pelagic representatives are **calanoids** and **cyclopoids**, which are distinguished by differences in their first antennae, urosome, and fifth leg morphology. The body is divided into an anterior metasome (cephalothorax), which includes the head with five pairs of appendages (antennae and mouthparts) and the thorax with six pairs of mainly swimming legs, and a posterior urosome composed of abdominal segments. In females, the first abdominal segment is modified as a genital segment, and the terminal caudal rami bear terminal setae (Wetzel,



2001; Błędzki & Rybak, 2016) (Fig. 15). Reproduction is sexual, involving internal fertilization, with females carrying external egg sacs until hatching.

Ecologically, calanoids are generally considered herbivorous or omnivorous filter feeders, whereas cyclopoids are more raptorial and predatory, although feeding strategies vary with species and environmental conditions. Together, they act as important regulators of phytoplankton and protozoan populations and serve as prey for higher trophic levels (Gliwicz, 2003; Min et al., 2021). Freshwater copepod diversity is substantial, with over 2,800 species identified globally. The Palaearctic region exhibits the highest species richness, and many species are endemic to specific regions, reflecting historical and ecological factors (Boxshall & Defaye, 2008).

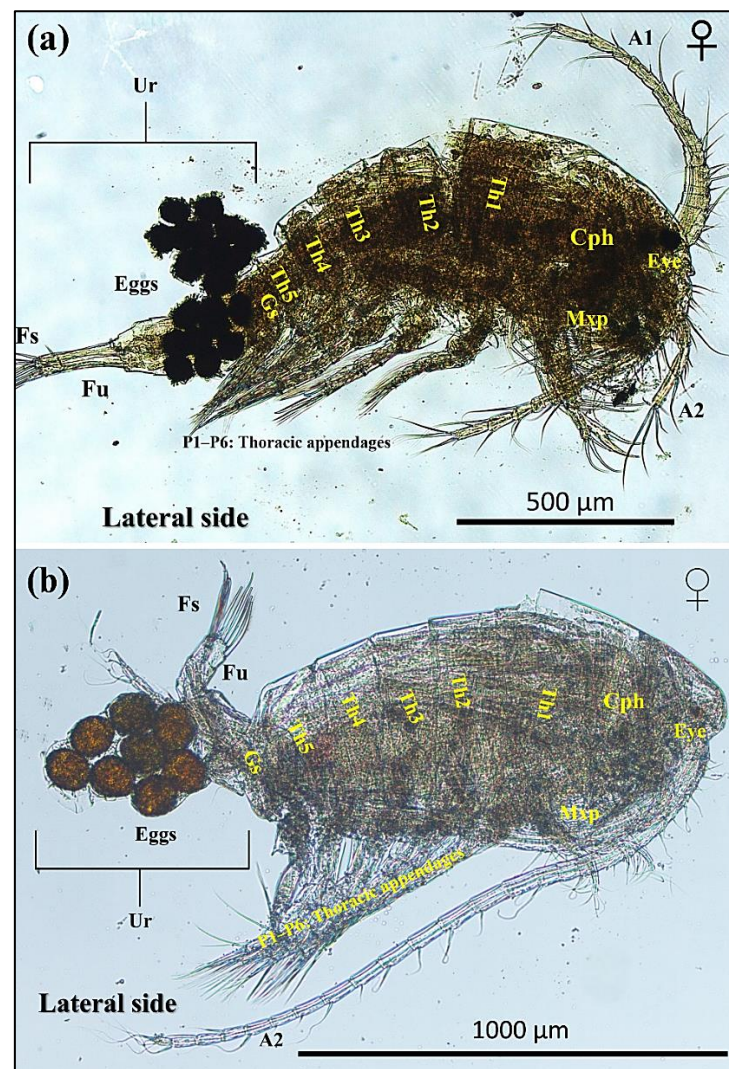


Fig. 15 Lateral view of Cyclopoida species: (a) *Cyclops abyssorum* and Calanoida species: (b) *Copidodiaptomus numidicus*.

Abbreviations: A1: Antennule, A2: Antenna, Cph: Cephalothorax, Fs: Furcal setae, Fu: Furcal rami, GS: Genital double somite, Mxp: Maxilliped, P1–P6: Thoracic appendages, paired swimming legs, Ur: Urosome.



1.4.3. Zooplankton ecology

Zooplankton dynamics in reservoirs are driven by physical, chemical, and biological factors, but unlike natural lakes, they are particularly influenced by hydrological regulation and water level fluctuations (Armengol, 1982; Picapedra et al., 2020; Zhikharev et al., 2022). Their distribution and abundance are shaped by climatic conditions, nutrient inputs, reservoir morphometry, and predation pressure from higher trophic levels (Baranyi et al., 2002; García-Chicote et al., 2018; Stamou et al., 2019; Bakhtiyar et al., 2020). Zooplankton inhabit nearly all aquatic environments, from alpine lakes to temporary ponds, and often reach higher densities in nutrient-enriched inland waters compared to oligotrophic ecosystems (Hutchinson, 1971; Pourriot, 1977; Wetzel, 2001). Among the dominant groups are rotifers, cladocerans, and copepods, which show distinct life-history traits and ecological roles (Ruttner-Kolisko, 1975; Cabral et al., 2019; Muñoz-Colmenares et al., 2021). Zooplankton are ecologically diverse, including planktonic and benthic forms, parasitic and symbiotic species, and even resistant resting stages that ensure survival under unfavorable conditions (Allan, 1976; Hairston, 1996; Vargas et al., 2024). They exhibit a wide range of adaptive strategies to cope with environmental variability (Allan, 1976; Lampert, 1989; Sommer et al., 2012), including:

- Vertical migration to optimize feeding and reduce predation risk (Bandara et al., 2021; Murphy et al., 2023);
- Reproductive strategies such as cyclical parthenogenesis and diapause/resting eggs (Breckenridge & Pakhomov, 2024; Vargas et al., 2024);
- Selective feeding and shifts in diet under food limitation or phytoplankton blooms (Sommer et al., 2012; Picapedra et al., 2020);
- Morphological defenses such as helmets, spines, and body size plasticity (Diel et al., 2021; Octorina et al., 2022);
- Physiological tolerance to stressors (temperature, hypoxia, salinity) (Sommer et al., 2012).

1.4.4. Zooplankton dynamics

Zooplankton communities also undergo marked seasonal succession, largely synchronized with phytoplankton dynamics, environmental drivers, temperature, predation, and interspecific interactions (Sommer et al., 2003; Lampert & Sommer, 2007). The **PEG model**, initially developed for phytoplankton, was later extended to include zooplankton, emphasizing their pivotal role in structuring planktonic food webs (Sommer et al., 1986; Sommer et al., 2012):



- **Late winter–spring:** Herbivorous cladocerans and copepods increase in abundance following the phytoplankton bloom, exerting strong grazing pressure and contributing to the clear-water phase (Sommer et al., 1986; Lampert & Sommer, 2007). Rotifers typically dominate early due to their rapid reproduction, while cladocerans peak later with higher grazing efficiency (Adrian et al., 2006; Sommer et al., 2012).
- **Early summer:** As edible phytoplankton decline, competitive interactions intensify among zooplankton. Larger cladocerans and calanoid copepods dominate, but selective feeding and food limitation reduce community biomass (Sommer et al., 2003; Hansson et al., 2013; Straile, 2015).
- **Mid–late summer:** Increased predation by planktivorous fish shifts community structure toward smaller-bodied species such as rotifers and cyclopoid copepods, a phenomenon known as size-selective predation (Brooks & Dodson, 1965; Jeppesen et al., 2000). Under eutrophic conditions, cladocerans may persist longer, while oligotrophic systems favor copepods (Gliwicz, 2003; Straile, 2015).
- **Autumn:** Declining temperatures, mixing events, and a resurgence of diatoms allow for a secondary increase in zooplankton, although biomass generally declines toward winter minima (Adrian et al., 2006; Sommer et al., 2012).

Zooplankton succession cannot be explained solely by resource dynamics; physical mixing regimes, predator–prey interactions, and adaptive life-history strategies (such as diapause and vertical migration) play fundamental roles (Jeppesen et al., 1997; Reynolds, 2011; Winder & Sommer, 2012; Mariani et al., 2013; Liu et al., 2022).

1.4.5. Bioindicative role of zooplankton

Zooplankton are valuable for characterizing aquatic ecosystems, with indicator species commonly used to assess changes in trophic state (Ejsmont-Karabin, 1995; Duggan et al., 2001; Jeppesen et al., 2011; García-Chicote et al., 2019; Rosińska et al., 2019; Muñoz-Colmenares et al., 2021). Their advantages as indicators stem from their biological and ecological traits, as well as their relative ease of study compared to phytoplankton. Zooplankton respond sensitively to environmental changes, integrating physical, chemical, and biological factors over time. Microzooplankton communities shift with eutrophication due to their small size and high metabolic rates (Park & Marshall, 2000; Duggan et al., 2001; Pawlowski et al., 2016), while their slower responses compared to phytoplankton allow them to integrate past episodes (Attayde & Bozelli, 1998; Sousa et al., 2008; Stamou et al., 2019). Positioned between primary producers and higher



trophic levels, zooplankton respond to both nutrient availability (bottom-up) and predation pressure (top-down) (Carpenter et al., 1985; Caroni & Irvine, 2010; Jeppesen et al., 2011; Bakhtiyar et al., 2020; Lomartire et al., 2021). Their short generation times enable rapid detection of community changes, as in rotifers, making them suitable for short-term assessments while also reflecting long-term conditions (Payne, 2013; García-Chicote et al., 2018). Their wide geographical distribution (Shurin et al., 2006; Sarma et al., 2019) ensures that local differences are more attributable to environmental drivers than to dispersal limitations (Whitman et al., 2004). Easy identification, high abundance, and cost-effectiveness further enhance their monitoring value (Whitman et al., 2004). Ecologically, freshwater zooplankton are integral to lake and reservoir food webs and are sensitive to anthropogenic impacts (Caroni & Irvine, 2010). When sampled from both water and sediments, they offer an indicator value unmatched by fish or phytoplankton alone, without substantial investment (Jeppesen et al., 2011). Due to their trophic significance and unique responses to specific environmental dynamics, zooplankton are effective bioindicators for measuring the impact of disturbance in aquatic ecosystems.

1.4.6. Functional role of zooplankton

1.4.6.1. Nutrients cycling

Zooplankton, present in high biomass in aquatic environments, occupy a central trophic position and act as key mediators of energy and elemental fluxes (Sterner, 2009; Stamou et al., 2019; Bakhtiyar et al., 2020; Lomartire et al., 2021). While functional classifications for zooplankton communities have emerged (Barnett et al., 2007; Merico et al., 2009; Litchman et al., 2021; Shchapov et al., 2021), most lack quantitative analyses of trait relationships and often omit physiological traits critical for ecosystem functions, such as nutrient recycling (Richon & Tagliabue, 2021; Schenone et al., 2021; Butts et al., 2022). Traits like respiration rate, excretion, fecal pellet composition, and sloppy feeding can strongly influence C, N, and P cycles, with their ecosystem impact depending on population size and trait expression.

Zooplankton contribute to nutrient availability and energy regeneration via direct physiological processes (assimilation, fecal pellet production, excretion) and indirect effects on prey biomass and community structure (Vanni, 2002; Butts et al., 2022). Through grazing, they ingest phytoplankton and microbes, assimilate organic matter, and release nutrients such as nitrogen and phosphorus back into the water via excretion and sloppy feeding, often in more bioavailable forms (Sterner, 2009; Gerphagnon et al., 2019). These processes stimulate phytoplankton growth, enhance microbial activity, and sustain primary production, particularly in



nutrient-limited ecosystems (Elser & Urabe, 1999; Boersma & Meunier, 2020). Most zooplankton are planktonic filter feeders that play a vital role in transferring organic matter, including up to 50% of fixed carbon, to higher trophic levels. Other species are benthic omnivores or predators (Błędzki & Rybak, 2016). Zooplankton also contribute to vertical nutrient redistribution through diel migration, transporting nutrients from surface to deeper layers and releasing them at depth (Sarma et al., 2016; Bandara et al., 2021; Wan et al., 2023). Their carcasses, fecal pellets, and molts further support microbial decomposition and remineralization, driving benthic–pelagic coupling and sediment nutrient fluxes (Hessen & Kaartvedt, 2014; Tang et al., 2014). By modulating the timing, location, and form of nutrient availability, zooplankton play a central role in maintaining ecosystem productivity and resilience (Sterner, 2009; Butts et al., 2022).

1.4.6.2. Trophic food web

Functionally, zooplankton occupy a central position in aquatic food webs, transferring energy from primary producers to higher consumers such as planktivorous fish and invertebrate predators (Jeppesen et al., 2011; Stamou et al., 2019; Bakhtiyar et al., 2020; Lomartire et al., 2021). By selectively grazing on phytoplankton, bacteria, and protists, they regulate primary production and microbial dynamics while channeling organic matter to higher trophic levels (Sommer & Sommer, 2006; Sterner, 2009; Schenone et al., 2021). Their community composition, body size, and feeding strategies strongly influence the efficiency of energy transfer and the structure of aquatic food webs (Katechakis et al., 2005). Large-bodied cladocerans often enhance energy transfer efficiency, whereas dominance of small rotifers or copepods may reduce grazing pressure on phytoplankton and alter trophic pathways (Sommer et al., 2003; Ger et al., 2019).

Predation by fish exerts top-down control on zooplankton, shaping size structure and producing cascading effects on phytoplankton biomass and water clarity (Jeppesen et al., 2011; Schulze, 2011; Ersoy et al., 2019; Liu et al., 2020). Visually oriented fish (e.g., carp, roach, bream) selectively prey on large-bodied zooplankton, particularly *Daphnia*, thereby reducing average size and fecundity of zooplankton populations (Brooks, 1968; Williams & Moss, 2003). Lower fish biomass is often associated with increases in large cladocerans and a corresponding suppression of algal biomass, unless high nutrient loading overrides this effect (Jeppesen et al., 1997; Tátrai et al., 2003). Fish community composition determines whether top-down effects dominate or if nutrient-driven phytoplankton growth prevails (Persson et al., 1999; Tátrai et al., 2003).



Additionally, non-predatory mortality factors such as disease, senescence, or toxin-induced death can equal or even exceed predation pressure in some reservoirs, significantly influencing zooplankton dynamics (Tang et al., 2014).

Human-driven eutrophication and climate change can further modify zooplankton–phytoplankton–fish interactions, destabilizing trophic networks and altering ecosystem functioning (Winder & Sommer, 2012; Zhao et al., 2022). Overall, zooplankton act as both regulators of lower trophic levels and essential prey for higher consumers, maintaining biodiversity and trophic balance in freshwater ecosystems (Shurin & Seabloom, 2005; Sterner, 2009; Butts et al., 2022).

1.4.7. Historical overview of zooplankton research in Algeria

The study of freshwater zooplankton in Algeria developed later than phytoplankton research and remains relatively fragmented. Early contributions were primarily descriptive and taxonomic, focusing on inventories of rotifers, cladocerans, and copepods from lakes, rivers, and reservoirs. Some of the earliest studies include those of Maupas (1889) and Gauthier (1928). Later, Cherbi (1984) investigated zooplankton assemblages in three reservoirs: Boughzoul, Hamiz, and Ghrib. Akli (1992) presented a Magister thesis on the systematics and geographical distribution of freshwater copepods (Microcrustacean) in northern Algeria, while Ould Rouis (1995) focused on the systematics and geographical distribution of cladocerans (Microcrustacean) in Algeria; both works serve as key references for species identification. A major milestone was the work of Samraoui et al. (1998), who inventoried Rotifera, Cladocera, Copepoda, and Ostracoda, several taxa were reported for the first time in Algeria and even in the Maghreb. This species list was subsequently expanded by Samraoui (2002).

During the 2000s and 2010s, research became more structured, integrating taxonomy with ecological analyses. Notable contributions include the inventory by Mouelhi et al. (2000) across North Africa, as well as ecological and taxonomic studies by Hamaidi et al. (2008) on the Chiffa River; Cherbi et al. (2008) on Boukourdane, Foug El Gherza, and Djorf Torba reservoirs; Bouzidi et al. (2010) on Lake Sidi M'hamed Benali; Bidi-Akli et al. (2014) on Zeralda reservoir; Doukhandji (2017) on rotifers from several Algerian reservoirs; Alliouche and Arab (2018) on the effects of physicochemical parameters on the dynamics of *Acanthocyclops robustus* in four reservoirs of different trophic states; Hamil et al. (2018; 2021) on Ghrib reservoir; Smaoune (2021) on nine Algerian reservoirs; and Ould Rouis et al. (2022) on Hamiz reservoir. In semi-arid regions, Bouazzara et al. (2021) also contributed to zooplankton research.



More recent studies have increasingly emphasized community ecology and the bioindicator role of zooplankton. For example, Amar et al. (2023) compared the dynamics of cladocerans and rotifers in Lake Sidi M'hamed Benali and the Sarno reservoir in relation to abiotic parameters, providing valuable insights into species richness and seasonal variability. The study by Baitiche et al. (2025) represents a major advance, assessing biodiversity and community structure in two northwestern Algerian reservoirs with contrasting morphometry.

Despite these advances, many Algerian aquatic ecosystems remain poorly explored, and comprehensive ecological assessments integrating species composition, abundance, functional groups, and environmental pressures are still rare.



I.5. Water quality assessment

Water quality assessment is a central theme in freshwater ecology and management, as it determines both ecosystem health and the suitability of aquatic resources for human use. A commonly accepted definition describes water quality as the sum of its physical, chemical, and biological characteristics (Alley, 2007; Spellman, 2008). It is typically assessed in relation to the needs of various living organisms and/or specific human uses and purposes (Shah & Joshi, 2017; Hassan-Omer, 2020).

For reservoirs, which are artificial ecosystems, the Water Framework Directive (WFD) introduced the term **ecological potential** instead of **ecological status**, applied to natural water bodies. This ecological potential is assessed through an index integrating biological, hydromorphological, and physicochemical indicators. Key variables include chlorophyll-a concentration, phytoplankton composition, cyanobacteria occurrence, and zooplankton composition, all of which are monitored due to their relevance for ecosystem functioning and potential health risks. Indicator values are then used to assign each water body to an ecological potential category ranging from poor to very good. Since the implementation of the WFD, several tools have been developed to classify water quality, building on earlier criteria (OECD, 1982; Palmer et al., 1992; Premazzi & Chiaudani, 1992; Carvalho et al., 2006; UKTAG, 2007; García-Chicote et al., 2018). Aquatic organisms, in particular, have been extensively used as biological indicators (Margalef, 1965; Saksena, 1987; Salachna & Olearczyk, 2020; Krokowski, 2021).

I.5.1. Physicochemical parameters

The assessment of water quality in freshwater ecosystems such as reservoirs is fundamental to understanding their ecological potential, biotic integrity, and suitability for human use (Wetzel, 2001). Physicochemical parameters directly influence biological processes, species composition, and overall ecosystem dynamics. Accurate monitoring of these parameters is therefore critical for evaluating anthropogenic impacts, trophic status, and ecosystem functioning (Allan & Castillo, 2007; Chapman, 2021). Below are some of the key physicochemical parameters that significantly influence reservoir water quality:

I.5.1.1. Temperature

Temperature is a fundamental parameter in the study of water quality. It affects the solubility of salts and gases (notably dissolved oxygen), and influences electrical conductivity and pH, and modulates key chemical reactions that control aquatic ecosystem functioning (Rodier,



2009; Benedini & Tsakiris, 2013). Higher temperatures can lead to decreased dissolved oxygen levels, with studies showing a strong negative correlation between temperature and dissolved oxygen (Danladi-Bello et al., 2017; Rajesh & Rehana, 2022). As a master abiotic driver, temperature regulates metabolic and enzymatic activity in aquatic organisms, impacting respiration, growth, and energy balance (Reynolds, 2006; Kültz, 2022). Recorded ranges of ~28–30 °C influence stratification, dissolved oxygen concentrations, and productivity, particularly in stratified ecosystems where temperature gradients shape nutrient cycling and plankton dynamics (Wetzel, 2001; De Stasio et al., 2009; Nasir et al., 2024).

I.5.1.2. Hydrogen potential (pH)

pH is a measure of water acidity, reflecting the concentration of hydrogen ions (H^+) (APHA, 2005; Sawyer et al., 2003). It indicates how acidic or alkaline the water is: acidic water contains an excess of hydrogen ions (H^+), whereas alkaline (basic) water contains more hydroxide ions (OH^-) (Alley, 2007). pH reflects the stability of the carbonic acid system and is closely linked to the buffering capacity provided by carbonates and bicarbonates (Fontela et al., 2023). As an indicator of acid-base balance, it determines the presence of specific chemical species and influences the bioavailability of nutrients and potentially toxic elements (Hussain et al., 2024). Extremely acidic or basic conditions can disrupt aquatic life. pH is influenced by photosynthesis, respiration, and chemical weathering (Rodier et al., 2009; Mihai et al., 2023). Most freshwater organisms thrive within a pH range of 6.5–8.5, and deviations can cause stress or even mortality in sensitive species (Pinheiro et al., 2021).

I.5.1.3. Dissolved oxygen

Dissolved oxygen is a key parameter for evaluating the quality of aquatic ecosystems, serving as a fundamental link between the mineral and organic components of lake environments (Wetzel, 2001; Crossman et al., 2021). Its concentration is influenced by multiple environmental factors, including water temperature, depth, the day–night light cycle, organic matter content, nutrient inputs, and the abundance of macrophytes, phytoplankton, and heterotrophic microorganisms (Kalff, 2002; Zhao et al., 2021; Lin et al., 2024). Oxygen enters freshwater ecosystems primarily through two mechanisms: gas exchange with the atmosphere and photosynthesis by aquatic primary producers such as algae and submerged plants (Cole et al., 1994; Marcé et al., 2024). Low dissolved oxygen levels can lead to hypoxic or even anoxic conditions, strongly affecting biological communities and altering biogeochemical processes (Dodds & Whiles, 2010). Its concentration is a sensitive indicator of ecosystem health, with hypoxic



conditions frequently associated with eutrophication and organic matter decomposition (Baker et al., 2019; Abdul-Aziz & Gebreslase, 2023).

I.5.1.4. Electrical conductivity

Electrical conductivity is a key water quality indicator that reflects the ability of water to conduct an electrical current, which depends on the total concentration of dissolved ions, such as cations (calcium, magnesium, sodium, potassium) and anions (chlorides, sulfates, nitrates, bicarbonates) (APHA, 2005). It provides an indirect measure of ionic content and salinity: the higher the conductivity, the greater the concentration of dissolved salts (Wetzel, 2001; Woszczyk et al., 2023). Conductivity is influenced by natural inputs (soil erosion, rock weathering) or anthropogenic sources (agricultural, industrial, or domestic discharges), making it a useful indicator of contamination or environmental disturbance (Allan & Castillo, 2007; Chapman, 2021). It also correlates with total dissolved solids (TDS) (Allan & Castillo, 2007). Abnormally high conductivity values may indicate pollution or ongoing eutrophication, potentially disrupting the ecological balance of aquatic ecosystems (Dodds & Whiles, 2010).

I.5.1.5. Turbidity

Turbidity is a measure of water transparency and is influenced by suspended particles such as clay, silt, sediments, phytoplankton, organic matter, and microorganisms (Wetzel, 2001; Alley, 2007). It reflects the cloudiness of water and serves as an important indicator of aquatic ecosystem quality (APHA, 2005). Turbidity controls light penetration in the water column; high turbidity reduces light availability, thereby limiting photosynthesis and altering biological communities, especially primary producers like phytoplankton. It also affects sedimentation rates and nutrient dynamics and it can signal pollution or ecological imbalance (Davies-Colley & Smith, 2001). Elevated turbidity increases water temperature due to greater absorption of solar radiation by suspended particles. In turn, this reduces dissolved oxygen concentrations, as warmer water holds less oxygen, ultimately affecting food availability for aquatic organisms (Cole et al., 1994; Power & Hailey, 2003). Increases in turbidity are often associated with surface runoff, erosion, and anthropogenic disturbances, which can reduce primary productivity and modify predator–prey interactions (Wetzel, 2001; Chapman, 2021). Due to these effects, turbidity is widely used as a monitoring parameter in freshwater ecosystems. Turbidity levels above 5 NTU are generally visible to the human eye, while values exceeding 100 NTU are typical of turbid or muddy waters (APHA, 2005).



I.5.1.6. Particulate matter

Particulate matter, or suspended matter, comprises a heterogeneous mixture of organic and inorganic particles in the water column, including detritus, plankton, mineral sediments, and anthropogenic inputs (Allan & Castillo, 2007). High suspended matter concentrations are often considered a form of pollution (Rodier et al., 2009). Their levels directly influence light penetration, primary production, and habitat quality, and they also affect the transport of nutrients and contaminants through adsorption processes (Bilotta & Brazier, 2008). High suspended matter levels can impair filter-feeding organisms, alter zooplankton community structure, and contribute to sedimentation that degrades benthic habitats (Davies-Colley & Smith, 2001). In reservoirs, particulate matter dynamics are shaped by hydrological conditions, watershed inputs, and internal resuspension processes (Wetzel, 2001; Ferreira et al., 2019; Fan et al., 2023).

I.5.1.7. Organic matter

Organic matter in aquatic ecosystems originates from autochthonous sources, such as phytoplankton and macrophyte production, as well as allochthonous inputs from the watershed, including plant debris and soil-derived material (Wetzel, 2001; Uehlinger & Zah, 2003; Hotchkiss & DelSontro, 2024). It plays a central role in nutrient cycling, serving as a substrate for microbial activity and influencing oxygen dynamics through its decomposition (Wan et al., 2023). High OM levels can increase biochemical oxygen demand (BOD), leading to hypoxic or anoxic conditions that negatively affect aquatic fauna and alter community structure (Smith, 2003). In reservoirs, OM accumulation is closely linked to hydrology, trophic status, and sedimentation processes, with significant implications for water quality and ecosystem functioning (Jargal et al., 2021).

I.5.1.8. Total dissolved solid (TDS)

Total dissolved solids denote the combined concentration of organic and inorganic ions, salts, and micro-granular matter, small enough ($\leq 2 \mu\text{m}$) to pass through filters. TDS serves as a key aggregate indicator of water quality rather than a direct pollutant (Grafton et al., 2012; Zukhruf et al., 2024). It affects aesthetics, technical water use, ecosystem health, and organism physiology, altering taste, causing scaling, influencing osmoregulation, and potentially suppressing spawning or increasing mortality in sensitive species (Chaudhary & Dwivedi, 2024; Verma et al., 2025). Major sources include natural mineral dissolution, agricultural and urban runoff, and industrial discharges.



I.5.1.9. Nutrient compounds

Nutrient compounds, mainly phosphorus and nitrogen, play a critical role in reservoir water quality by influencing biological productivity and the functioning of aquatic ecosystems. At moderate concentrations, they promote the growth of aquatic plants and phytoplankton, which are essential components of the food web (Smyth et al., 2022). However, an excess of these nutrients, often from agricultural fertilizers or urban pollution, can lead to eutrophication, characterized by dense algal blooms, particularly toxic cyanobacteria, that degrade water quality and threaten biodiversity (Rodier et al., 2009; Naeem et al., 2014; Jose et al., 2023). Managing nutrient compounds (nitrogen and phosphorus) alongside dissolved salts is essential for preserving lakes and their role as biodiversity reservoirs (Saccò et al., 2021; Grochowska & Augustyniak-Tunowska, 2025).

Among nitrogen forms, **nitrites** (NO_2^-) are intermediates in the nitrogen cycle, formed by the oxidation of ammonium or reduction of nitrates. High nitrite concentrations may indicate organic or toxic pollution (De Villers et al., 2005; Rodier et al., 2009; Chamoli et al., 2024). **Nitrates** (NO_3^-), the end products of nitrification, mainly originate from agricultural runoff or urban effluents. As major plant nutrients, their accumulation contributes to eutrophication (Balejčíková et al., 2020). **Ammonium** (NH_4^+), a reduced form of nitrogen, also serves as an indicator of recent pollution. **Orthophosphates** (PO_4^{3-}) originate from natural sources, agriculture (fertilizers), or domestic activities (detergents). Often the limiting factor in primary production, they play a key role in eutrophication. Excessive inputs of nitrate (NO_3^-), ammonium (NH_4^+), and orthophosphate (PO_4^{3-}) stimulate phytoplankton growth, potentially leading to harmful algal blooms (Smith, 2003; Paerl, 2017; Wurtsbaugh et al., 2019). Nutrient ratios, such as N:P, further influence algal community composition and trophic dynamics (Redfield, 1958; Sardans et al., 2012; Schindler, 2012; Hofmann et al., 2021).

I.5.2 Trophic state of reservoirs

The trophic state is one of the most important parameters for characterizing aquatic ecosystems (Wen et al., 2011; Neverova-Dziopak et al., 2023). It reflects the biological productivity of lakes and is generally assessed using indicators such as total phosphorus, chlorophyll-a concentration, and water transparency measured with a Secchi disk (Carlson, 1977). Today, biomonitoring is increasingly recognized as a key approach for evaluating water quality, particularly in studies assessing pollution (Mohan & Omana, 2007; Ogidi et al., 2024).



Eutrophication, driven by excessive nutrient inputs such as nitrates and phosphates, leads to declines in dissolved oxygen and major ecological imbalances (Paerl et al., 2017).

1.5.2.1. Trophic state indices

1.5.2.1.1. Carlson's Trophic State Index (TSI)

The trophic state of aquatic ecosystems, including reservoirs, is commonly assessed using Carlson's Trophic State Index (TSI) (Carlson, 1977). This widely-used metric classifies ecosystems based on three parameters, chlorophyll-a concentration, Secchi disk transparency, and total phosphorus, which respectively indicate algal biomass, light penetration, and nutrient availability. By integrating these metrics, the TSI enables standardized classification and long-term monitoring of trophic dynamics, categorizing waters from **oligotrophic** (low productivity) through **eutrophic** to **hypereutrophic** (high productivity). Chlorophyll-a provides the most direct measure of phytoplankton biomass, total phosphorus represents the limiting nutrient governing algal growth, and Secchi transparency reflects the combined effects of biological and non-biological turbidity. High concentrations of these parameters signal nutrient enrichment and are often associated with water-quality degradation (Carlson, 1977). Threshold values for classifying trophic categories are summarized in the Supplementary Material (Table S1).

1.5.2.1.2. Phytoplankton-based indices

Phytoplankton, due to its sensitivity to variations in nutrients (nitrogen, phosphorus), light, pH, and temperature, serves as an excellent bioindicator of water quality (Padisák et al., 2009; Chandel et al., 2024). Its rapid response to eutrophication makes it one of the most effective indicators of the trophic status of lakes and other aquatic environments (Reynolds, 2006; Lyche-Solheim et al., 2013; Katsiapi et al., 2016; Stamou et al., 2019). Phytoplankton has been officially included as a Biological Quality Element (BQE) in the EU Water Framework Directive (WFD), reflecting its central role in nutrient dynamics and eutrophication processes (Moss, 2008; Poikane et al., 2019).

Phytoplankton-based assessment methods rely on a variety of indices that integrate species composition, abundance, diversity, and functional traits. Among the most widely applied is **Phytoplankton Trophic Index (PTI)**, based on indicator taxa sensitive to nutrient enrichment (Phillips et al., 2013; Mischke et al., 2016; Meyer et al., 2024). It was the first phytoplankton index developed for water bodies, standardized in 1990 (Barbe et al., 1990). However, PTI often overestimated water quality, showed abrupt shifts with slight changes in chlorophyll, and lacked



methodological clarity (Cergrene, 1994; Ochocka & Pasztaleniec, 2016). It was later replaced by **Planktonic Index (IPL)**, which assigns a quality score (Q_i) to each algal group (e.g., Cyanobacteria, Chlorophytes) based on its association with eutrophication (Barbe et al., 2003; Ochocka & Pasztaleniec, 2016). In addition, **Lacustrine Plankton Index (IPLAC)** (Feret & Laplace-Tretyure, 2013; Laplace-Tretyure & Feret, 2016) incorporate biomass, taxonomic composition, functional groups, and the ecological preferences of algal assemblages, providing a more integrative measure of nutrient status and ecological integrity. Another widely used tool is **Palmer's Algal Pollution Index**, which evaluates water quality based on the presence and abundance of pollution-tolerant genera (e.g., *Oscillatoria*, *Scenedesmus*, *Chlorella*), each assigned a tolerance score (Palmer, 1969). More recent indices focus on phytoplankton functional groups, capturing ecological responses to physical and chemical drivers (Padisák et al., 2009; Salmaso et al., 2015), while multimetric approaches combine biomass metrics (chlorophyll-a), taxonomic composition, and bloom-forming cyanobacteria to provide a more robust evaluation of ecological status (Lyche-Solheim et al., 2013; Poikane et al., 2019; Crossman et al., 2021).

Phytoplankton indices remain among the most reliable indicators for ecological assessment; however, their interpretation is often challenged by high temporal variability, hydrodynamic influences, and difficulties in taxonomic resolution (Reynolds, 2006; Padisák & Naselli-Flores, 2021). Nevertheless, phytoplankton-based metrics continue to serve as the backbone of ecological monitoring worldwide, providing integrative and early-warning signals of water quality degradation (Phillips et al., 2013; Poikane et al., 2020).

1.5.2.1.3. Zooplankton-based indices

Zooplankton, as a primary consumer in aquatic food chains, is also a reliable indicator of environmental alterations. Its community composition varies with trophic conditions and water quality (Jeppesen et al., 2011; García-Chicote et al., 2019; Muñoz-Colmenares et al., 2021). Despite its key role in pelagic food webs, zooplankton has not been included as a Biological Quality Element (BQE) under the WFD (Moss, 2007; Caroni & Irvine, 2010; Davidson et al., 2011; Jeppesen et al., 2011; Ejsmont-Karabin, 2012; Stamou et al., 2019; Montagud et al., 2019). This omission has often been attributed to its historical absence from monitoring programs, possibly due to financial constraints. Caroni & Irvine (2010) outlined possible reasons for its exclusion from the WFD:



- As an intermediate link between phytoplankton and higher trophic levels, zooplankton is influenced by both bottom-up and top-down pressures, which complicates the interpretation of anthropogenic effects.
- It has been considered relevant only for standing waters, although this contradicts the WFD itself, which acknowledges that not all biotic groups apply to every water body type, yet still includes phytoplankton in Annex V.
- No zooplankton expert participated in drafting the Directive, regarded by the authors as the most plausible reason and reportedly common in early WFD discussions.

The use of zooplankton as indicators of trophic status and water quality dates back to at least 1978 (Gannon & Stemberger). Ejsmont-Karabin (2012) developed rotifer-based indices to assess trophic status in north-eastern Poland, reflecting bottom-up control by phytoplankton biomass and reduced top-down regulation from competitors. In South America, cyclopoid copepods have been used to assess reservoir water quality, with certain species strongly correlated with chlorophyll-a and total phosphorus (Perbiche-Neves et al., 2021). Cladocerans are also widely applied in paleolimnological studies to reconstruct historical trends in trophic status and environmental change (Leoni et al., 2021). Zooplankton communities, being abundant and diverse, provide valuable information on ecosystem functioning. They can reflect environmental productivity (phosphorus) (Simões et al., 2015), fish community structure (Hessen et al., 2006), water chemistry, water body size (de Bie, 2008), anthropogenic impacts, and climate change (Almeida et al., 2020). Community composition is shaped by species-specific requirements for total organic carbon, chlorophyll-a, pH, and depth (Skála, 2015). As chemical conditions shift with climate change, zooplankton communities respond through changes in phenology, abundance, distribution, size spectra, and structure (Vadadi-Fülöp et al., 2012).

Over the past two decades, numerous studies have confirmed zooplankton's value as an excellent bioindicator of the ecological integrity of reservoir ecosystems (Boix et al., 2005; Carpenter et al., 2006; Ejsmont-Karabin, 2012; Ejsmont-Karabin & Karabin, 2013; Haberman & Haldna, 2014; Ochocka & Pasztaleniec, 2016; Doukhandji & Arab, 2017; García-Chicote et al., 2019; Muñoz-Colmenares et al., 2021; Smaoune et al., 2021; Karpowicz et al., 2025).

The most commonly used zooplankton-based indices are those developed by Ejsmont-Karabin (2012), which include the trophic state index based on rotifer abundance (TSI_{Rot}). Ejsmont-Karabin and Karabin (2013) also proposed indices based on crustacean abundance (TSI_{Cru}) and the cyclopoid-to-calanoïd ratio ($TSI_{Cy/Ca}$) to assess water quality. Additional



zooplankton-based metrics account for species richness, diversity, and functional traits sensitive to nutrient enrichment (Jeppesen et al., 2011; Błędzki & Rybak, 2016). Such indicators provide integrative responses to environmental conditions and serve as early-warning tools for ecological degradation.

I.5.3. Historical overview of water quality assessment by phytoplankton and zooplankton in Algeria

In Algeria, water quality assessments have historically focused predominantly on physicochemical variables (nutrients, oxygen, conductivity, turbidity) rather than biological indices. As a result, the use of plankton-based trophic indicators remains scarce. Research on phytoplankton has primarily targeted harmful algal blooms (HABs), particularly cyanobacteria such as *Microcystis*, *Planktothrix agardhii*, *Cylindrospermopsis raciborskii*, and *Aphanizomenon issatschenkoi*, which have been frequently reported in Mediterranean reservoirs (Nasri et al., 2008; Bidi-Akli et al., 2014; Bouhaddada et al., 2016; Saoudi et al., 2017; Benayache et al., 2022; Guellati et al., 2024). These studies emphasized the risks of cyanotoxins for water use and human health but did not translate into the development of systematic trophic indices.

Zooplankton-based assessments are even more limited. Only a handful of studies have considered rotifers as bioindicators in Algerian reservoirs, most notably those by Doukhandji and Arab (2017) and Smaoune et al. (2021). These pioneering works highlighted the potential of rotifer community structure to reflect trophic gradients, yet such approaches remain underdeveloped compared to international practices.

Overall, while the global scientific community has progressively established robust phytoplankton- and zooplankton-based trophic indices since the late 20th century, in Algeria the application of these methods is still at an early stage. Most reservoir monitoring continues to emphasize physicochemical parameters, with only sporadic integration of plankton communities. Strengthening plankton-based biomonitoring in Algeria would therefore represent an important step toward aligning with international standards and enhancing ecosystem-based water management.

Chapter II: Study area





II.1. Study area

Cheliff and Kerrada reservoirs are located in northwestern Algeria, approximately 30 km northeast of Mostaganem, within Zahrez Cheliff hydrographic basin in wilaya of Mostaganem. Kerrada reservoir lies about 9 km upstream of Cheliff reservoir. Cheliff reservoir is situated at $35^{\circ}59'12.5''\text{N}$, $0^{\circ}26'20.6''\text{E}$, while Kerrada reservoir is positioned at $36^{\circ}03'30.3''\text{N}$, $0^{\circ}23'05.3''\text{E}$, both at an altitude of approximately 500 m (Baitiche et al., 2025) (Fig. 16, Fig. 17). Mostaganem, the 27th wilaya of Algeria, is located on Mediterranean coast, 350 km west of Algiers and 80 km east of Oran. It encompasses the mountainous Dahra region, characterized by a predominantly rural population, as well as the Mostaganem plateau (Said, 2019). Cheliff reservoir serves as the primary water source for Kerrada reservoir, where water is temporarily stored before undergoing treatment and being regularly transferred to urban and industrial centers according to demand. These reservoirs are central to a large-scale water transfer project designed to supply irrigation and drinking water to the M.A.O. corridor, which includes three major provinces: Mostaganem, Arzew, and Oran.

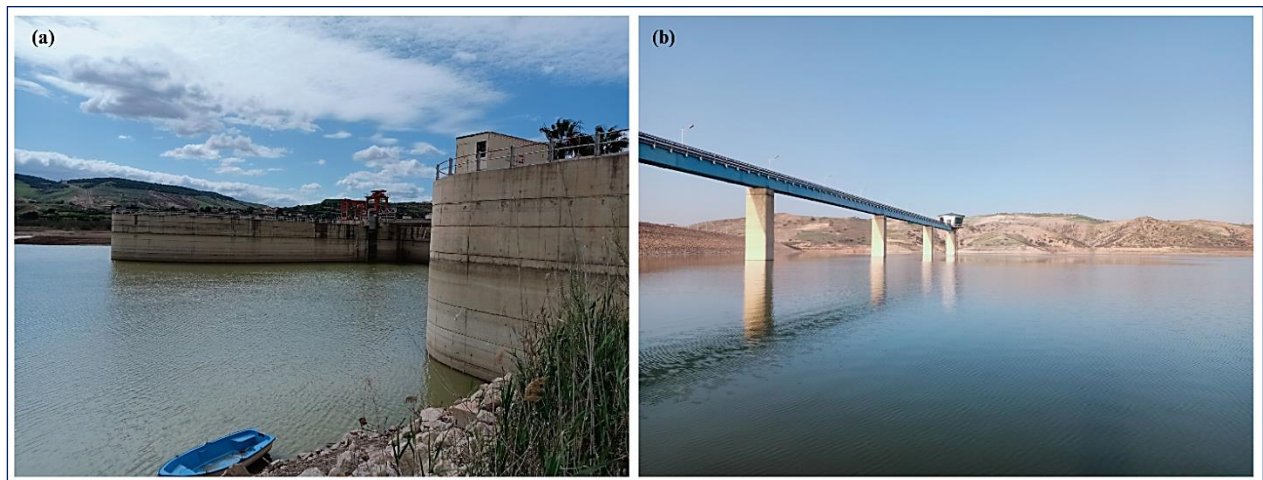


Fig. 16 Study area: (a) Cheliff reservoir; (b) Kerrada reservoir.

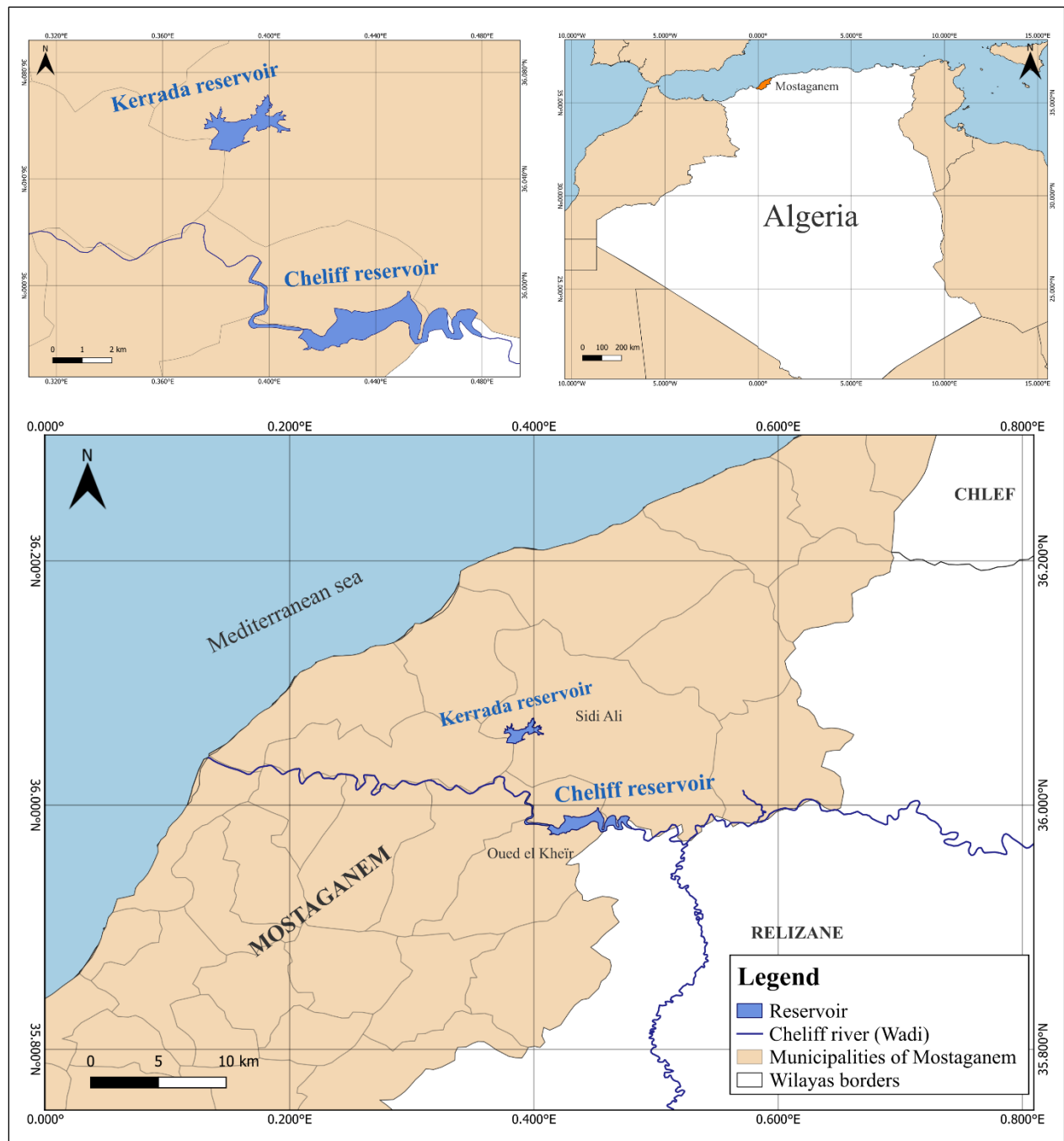


Fig. 17 Location map of Cheliff and Kerrada reservoirs.



II.2. Morphometric characteristics

Morphometrically, Cheliff reservoir is a shallow diversion reservoir constructed on Cheliff Wadi, characterized by a relatively large watershed area and low storage capacity. It is designed primarily to regulate river flow and transfer water downstream. The structure is of mixed type, comprising a concrete weir with a surge basin and a zoned earthen embankment with a watertight core, with a dam crest elevation of 35 m (NGa). In contrast, Kerrada reservoir functions as a deep storage reservoir, with a larger storage volume and depth that enable long-term regulation and water supply. It is a zoned embankment reservoir with a symmetrical central core and sandstone riprap counterweights, equipped with a tower-type water intake with multiple intake openings. The maximum discharge capacity of the structure is $6.5 \text{ m}^3 \text{ s}^{-1}$ (National Agency for Dams and Transfers [N.A.D.T.], 2017; 2018; Baitiche et al., 2025) (Table 1).

Table 1 Morphometric details and use of Cheliff and Kerrada reservoirs (National Agency for Dams and Transfers [N.A.D.T.], 2017; 2018; Baitiche et al., 2025).

Reservoir	Year of construction	Year of impounding	Altitude (m)	Mean depth (m)	Maximum height (m)	Watershed area (km ²)	Maximum storage capacity (hm ³)
Cheliff	2007	2009	480	6	26	44	50
Kerrada	2007	2010	540	115	133.5	19	70

II.3. Hydrological features

Hydrographically, the reservoirs form part of **Cheliff–Zahrez watershed**, which integrates three major hydrographic sub-basins: Cheliff (43,750 km²), endorheic Zahrez (8,989 km²), and coastal Dahra (3,201 km²), classified as a moderate relief system. Cheliff hydrographic basin, extending over 700 km, originates from the confluence of two rivers, Touil Wadi (16,338 km²) and Nahr Ouassel (3,651 km²), which merge to form the Cheliff Wadi (Meddi & Mehaiguen, 2007). Bordered by the Mediterranean to the north, the Ouarsenis mountain chain to the south, and the Algiers and Oran watersheds to the east and west, respectively, the basin represents Algeria's largest hydrographic system (Sadeuk Ben Abbes et al., 2020; Brahimi et al., 2023). Cheliff Wadi itself, originating in the Saharan Atlas near Aflou (Jebel Amour), stretches for about 750 km before discharging into the Mediterranean, making it the longest river in Algeria and a critical axis for the socioeconomic development of the region (Elmeddahi et al., 2016; Achite et al., 2021; Brahimi et al., 2023).



The main inflows to Cheliff reservoir come from the Cheliff Wadi. Its net evaporation (evaporation minus precipitation) is estimated at $1,000 \text{ mm yr}^{-1}$, with an annual runoff of 87 mm , equivalent to 61 hm^3 . Sediment deposition on the riverbed represents about 30% of suspended sediment, with an average density of 1.4 t m^{-3} . The peak flood discharge with a 10,000-year return period is estimated at $6,770 \text{ m}^3 \text{ s}^{-1}$. In contrast, Kerrada reservoir is supplied exclusively by Cheliff reservoir, making it a single-source system. Its mean annual net evaporation reaches $2,217.6 \text{ mm yr}^{-1}$, with an estimated liquid inflow of $1.47 \text{ hm}^3 \text{ yr}^{-1}$ and a solid inflow of $5,200 \text{ m}^3 \text{ yr}^{-1}$. Flood discharges and volumes for different return periods are estimated at $59.1 \text{ m}^3 \text{ s}^{-1}$ and 0.32 hm^3 for 20 years, $101.2 \text{ m}^3 \text{ s}^{-1}$ and 0.59 hm^3 for 200 years, and $175.6 \text{ m}^3 \text{ s}^{-1}$ and 1.07 hm^3 for 10,000 years (National Agency for Dams and Transfers [N.A.D.T.], 2017; 2018).

II.4. Description of MAO transfer system

Cheliff–Kerrada–WTP (Water Treatment Plant transfer system diverts water from Cheliff wadi, enabling short-term (weekly) regulation of Wadi inflows. Water is first stored in Cheliff diversion reservoir and then transferred to Kerrada storage reservoir, which reduces spillage and sediment accumulation while improving diversion efficiency (Baitiche et al., 2025). From there, water is treated at Sidi Hadjel Water Treatment Plant ($6.5 \text{ m}^3 \text{ s}^{-1}$) and supplied to Mostaganem–Arzew–Oran (M.A.O.) corridor through an 88.5 km gravity-fed pipeline. The system infrastructure includes Cheliff diversion reservoir (50 hm^3 , 20 hm^3 usable), Kerrada storage reservoir (70 hm^3 , 65 hm^3 usable), Cheliff pumping station ($9.5 \text{ m}^3 \text{ s}^{-1}$) with a 6.7 km pipeline to the WTP, and a reversible Cheliff–Kerrada pipeline ($9.5 \text{ m}^3 \text{ s}^{-1}$, 8.7 km). The annual regulated volume ($155 \text{ hm}^3 \text{ yr}^{-1}$) is distributed between Mostaganem (45 hm^3) and Oran (110 hm^3). Hydrological assessments indicate that available inflows are sufficient to meet potable and industrial demands, excluding additional upstream contributions from reservoirs such as Gargar and Kramis (Sembenelli & Poletini, 2010; National Agency for Dams and Transfers [N.A.D.T.], 2017; 2018).

II.5. Climatic context

Climate plays a fundamental role in the dynamics and distribution of ecosystems (Grimm et al., 2013; Sharma et al., 2025; Soifer et al., 2025). It directly influences species distribution by shaping the environmental conditions required for survival and reproduction (Tahri et al., 2020). Among the most widely used climatic indices, temperature and precipitation are the most influential and best-documented environmental factors (Gutiérrez-Hernández & García, 2021). Water availability and scarcity depend directly on precipitation, one of the key climatic and



hydrological drivers (Stenseth et al., 2002; Nasri et al., 2016; Pfister et al., 2018; Derdous et al., 2021). Extreme events such as drought, declining precipitation, and global temperature rise negatively impact water availability for different socio-economic sectors (Taïbi et al., 2021; Brahimi et al., 2023).

The Mediterranean basin is climatically influenced by the interaction of mid-latitude and tropical processes, being located in a transition zone between arid climate of North Africa and temperate, rainy climate of Europe. Consequently, it is recognized as a climate change hotspot, subject to pronounced warming and drying, with significant effects on the spatial and temporal distribution of precipitation (Achite et al., 2021).

In Algeria, particularly in the northwestern region, recent climatic trends show a marked decline in precipitation, estimated at about 12.4 mm per month per century since the mid-1970s, along with an increase in mean annual temperatures of approximately 2.7 °C over the same period (World Bank, 2021; Zaibak & Meddi, 2022). This trend of declining hydrological regimes has been confirmed across several catchments in western Algeria (Meddi & Talia, 2013; Elmeddahi et al., 2016). Reduced rainfall has directly impacted dam filling and the natural recharge of aquifers (Meddi & Eslamian 2021; Zaibak & Meddi, 2022). This progressive aridification exerts considerable pressure on local ecosystems by reducing water availability and altering the distribution of plant and animal species (Tahri et al., 2020).

For this study, meteorological data covering the period from 2012 to 2022 were obtained from the **Algerian National Meteorological Office** (O.N.M.) (Algerian National Meteorological Office [ONM], 2022), which provides detailed records of the relevant climatic parameters for the study area.

II.5.1. Temperature

Temperature variations have a significant impact on aquatic ecosystems, influencing processes such as evaporation, water levels, and biodiversity. Higher temperatures increase evaporation rates, leading to lower water levels in reservoirs, which can negatively affect biodiversity by reducing the availability of habitats for aquatic species (Grant, 2012; Magee & Wu, 2017; Boyd, 2020; Sanwal et al., 2024). Conversely, cooler temperatures during wet periods can promote reservoir enrichment, thereby enhancing ecosystem diversity (Jeppesen et al., 2015; Winton et al., 2019; Hassan et al., 2020; Hassan Omer, 2020).



According to the data presented in the graph (Fig. 18), January is the coldest month, with an average temperature of 11.2 °C, while August is the warmest, at 27.4 °C. These variations illustrate the seasonal temperature fluctuations in the study area, which have important implications for the structure and functioning of aquatic ecosystems.

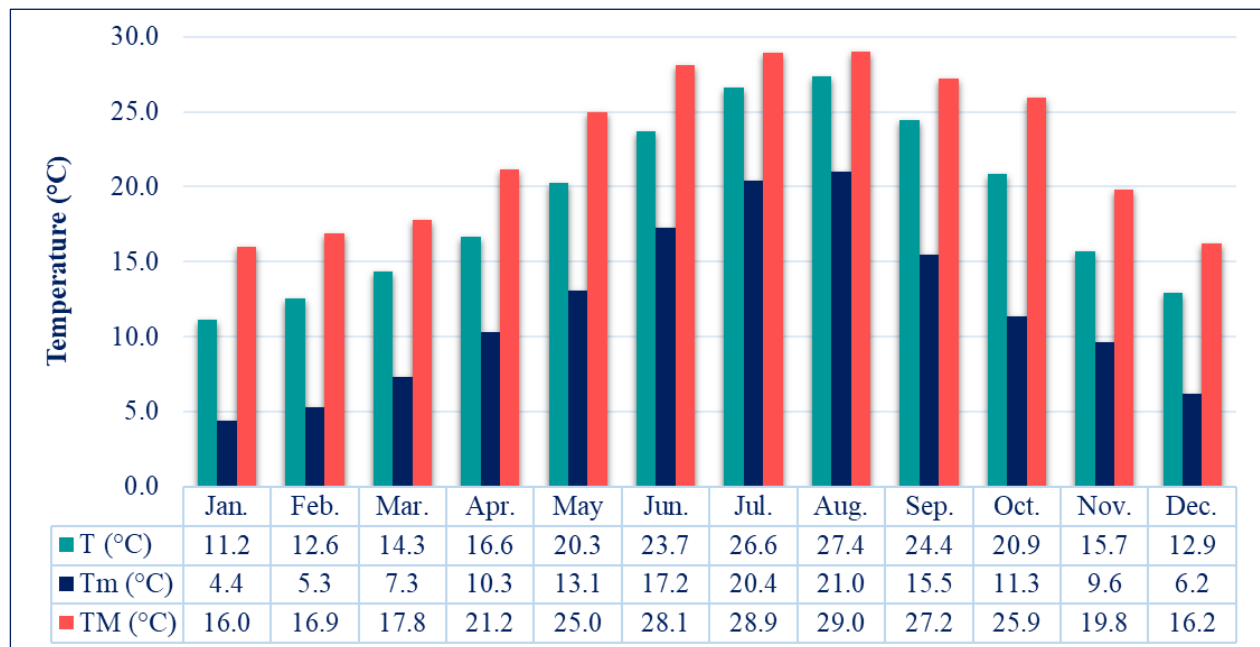


Fig. 18 Monthly mean temperature variations in the study area during the 2012–2022 period.

II.5.2. Precipitation

Precipitation significantly influences the distribution and growth of plant and animal species, acting as a key climatic factor ([Gutiérrez-Hernández & García, 2021](#)). The relationship between water availability and species distribution is complex, as it not only affects growth but also shapes ecological niches and evolutionary processes. Recent research shows that water availability is a primary driver of species turnover, particularly in tropical regions, where up to 95% of speciation occurs within specific precipitation niches ([Ringelberg et al., 2023](#)). This relationship is further complicated by climate change, which alters precipitation patterns and increases the frequency of droughts and floods, thereby affecting plant productivity and survival ([Rahman, 2019](#)).

Algeria, like other Maghreb and Mediterranean countries, has experienced a marked reduction in rainfall and recurrent drought events since the mid-1970s ([Meddi & Hubert, 2003](#); [Longobardi & Villani, 2010](#); [Elmeddahi et al., 2016](#); [Achour et al., 2020](#); [Tramblay et al., 2020](#); [Achite et al., 2021](#); [Zaïbak & Meddi, 2022](#)). This trend is particularly pronounced in semiarid regions. Cheliff watershed illustrates this spatial variability: average annual rainfall decreases from



north to south, reflecting the influence of altitude and relief in the Tellian ranges. Rainfall declines from 778.8 mm yr⁻¹ in the north to 249.1 mm yr⁻¹ in the southern highlands (Elmeddahi et al., 2016).

In the study area, precipitation exhibits marked monthly variability. The highest rainfall occurs in winter and autumn, with 79.47 mm in January and 69.21 mm in November. The lowest precipitation is recorded in July, with only 0.46 mm (Fig. 19).

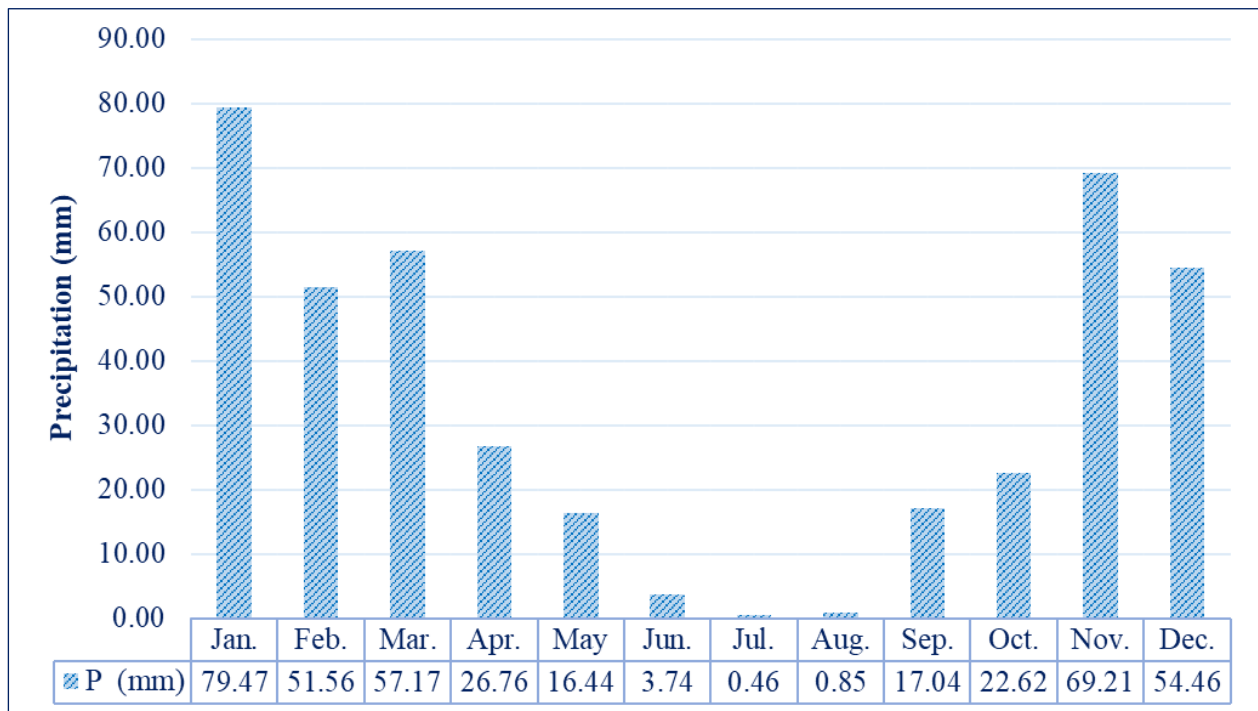


Fig. 19 Monthly mean precipitation variations in the study area during the 2012–2022 period.

II.5.3. Ombrothermic diagram

Ombrothermic diagram is a relevant tool for visualizing and quantifying dry and wet periods, based on the climatic criterion $P < 2T$, according to the method developed by Bagnouls and Gaussen (1957). In this type of graph, the months of the year are shown on the x-axis, while mean monthly temperatures and precipitation are plotted on the y-axis, using a scale in which 20 mm of precipitation is equivalent to 10 °C, thus providing a double scale for precipitation. The dry period is identified when the temperature curve lies above the precipitation curve, whereas the wet period corresponds to the months in which the precipitation curve exceeds the temperature curve (Derdous et al., 2021).



According to the ombrothermic diagram (Fig. 20), two distinct periods can be observed:

- **A dry period (D.P.)** lasting seven months, from April to October;
- **A wet period (W.P.)** beginning at the end of October and extending until the beginning of April.

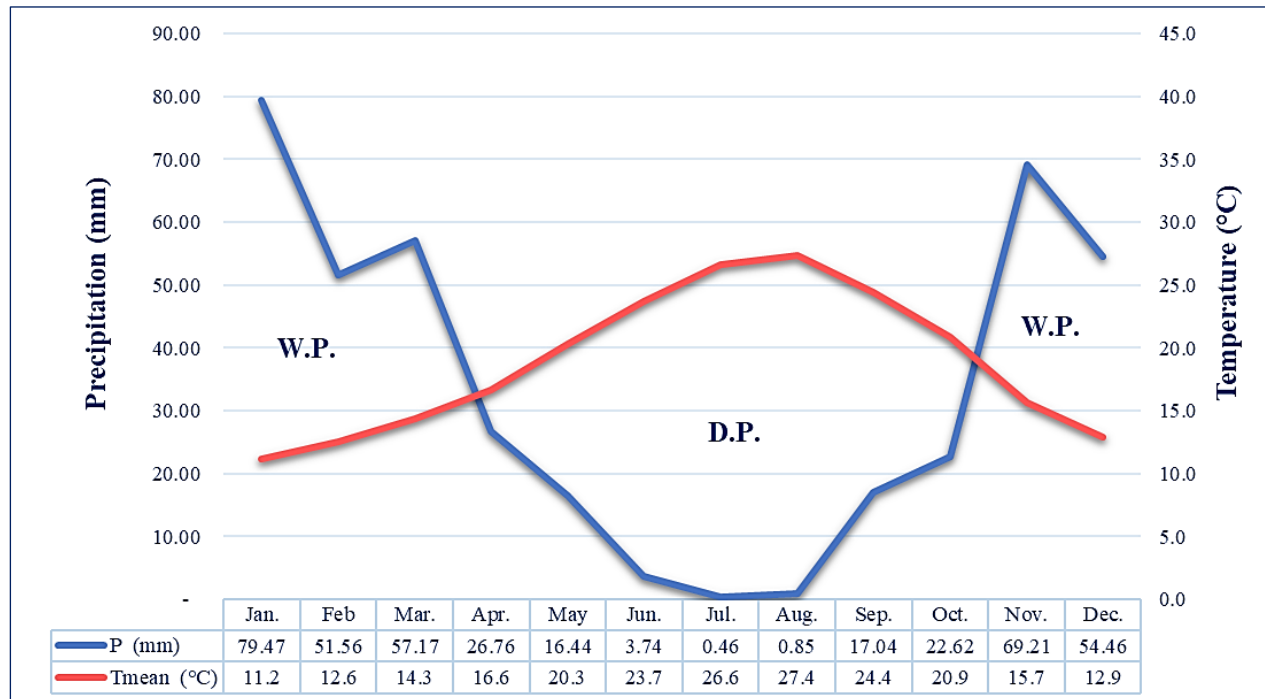


Fig. 20 Ombrothermic diagram of the study area for the period 2012–2022.

II.5.4. Emberger's pluviometric quotient

According to Emberger (1955), the Mediterranean region can be subdivided into five bioclimatic zones, defined based on the relationship between precipitation and temperature. This classification is still supported by recent studies in the region (Derdous et al., 2021). To characterize the bioclimatic zone of the study area, Emberger's pluviometric quotient (Q_2) was calculated. This quotient provides a better understanding of local climatic conditions by integrating annual precipitation along with the mean temperatures of the hottest and coldest months. The formula of the quotient is expressed as follows:

$$Q_2 = \frac{1000.P}{(M+m)(M-m)/2} = \frac{2000.P}{M^2 - m^2}$$

Where:

- **P**: mean annual precipitation (mm).
- **M**: mean temperature of the hottest month (K).
- **m**: mean temperature of the coldest month (K) (see Table S2, Supplementary material).



Based on Emberger climagram, the study area is characterized by a Mediterranean climate and belongs to the **semi-arid bioclimatic stage** with a temperate winter (Emberger, 1955) (Fig. 21).

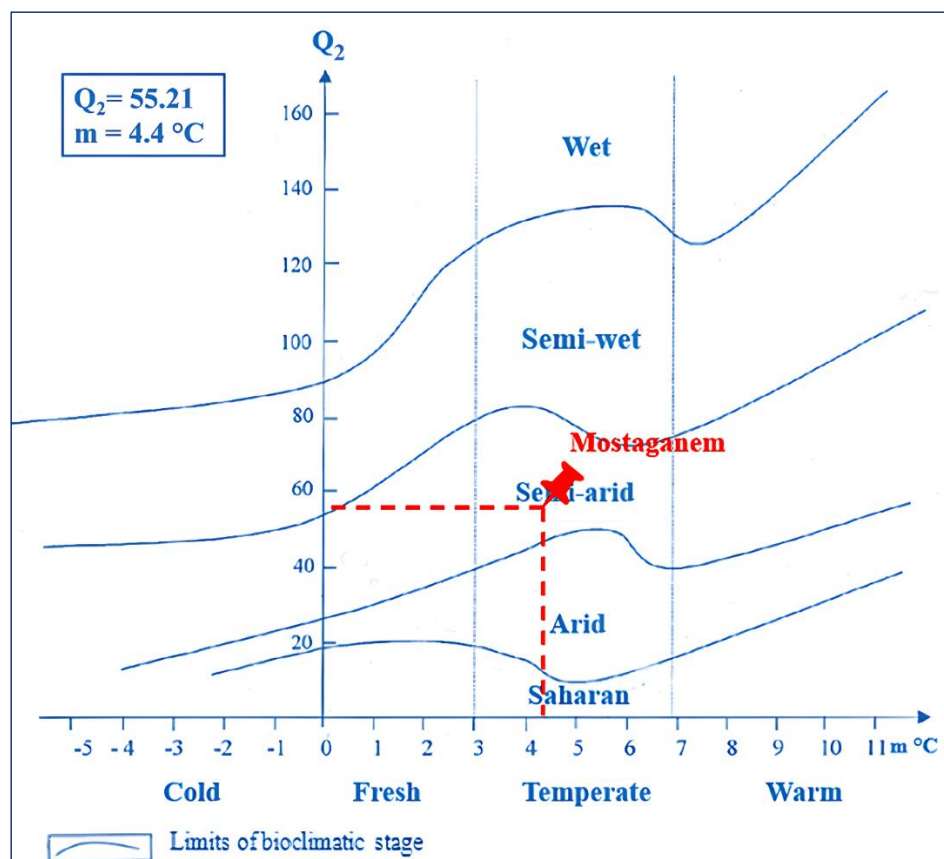


Fig. 21 Bioclimatic stage of the study area based on the Emberger climagram for the period 2012–2022.

II.5.5. Wind

Wind is a key factor in shaping Mediterranean climate, defined by both its speed and direction, and it exerts a significant influence on aquatic ecosystems. In northern Algeria, winds typically range from northwest to northeast, with storm events frequently originating from the north-northeast between December and March (Houta & Gadiyatov, 2023). This atmospheric dynamic plays a critical role in the circulation and oxygenation of reservoir waters. The interaction between surface winds and water promotes turbulence at the air–water interface, enhancing oxygen diffusion and facilitating the dispersion of planktonic organisms, processes essential for maintaining water quality and ecosystem resilience. Recent climate data for Algeria indicate that mean regional wind speeds generally average around 2.3 m s^{-1} in the north (Boudia & Santos, 2019). Wind-driven mixing enhances aerobic conditions in reservoirs and supports plankton distribution, thereby fostering biodiversity. However, strong winds may also trigger adverse



processes such as sediment resuspension, destabilization of pelagic communities, and alterations in nutrient dynamics (Kies & Kerkouf, 2014).

Average wind speed ranges from 6.2 km h⁻¹ in October to 9.1 km h⁻¹ in April (Fig. 22).

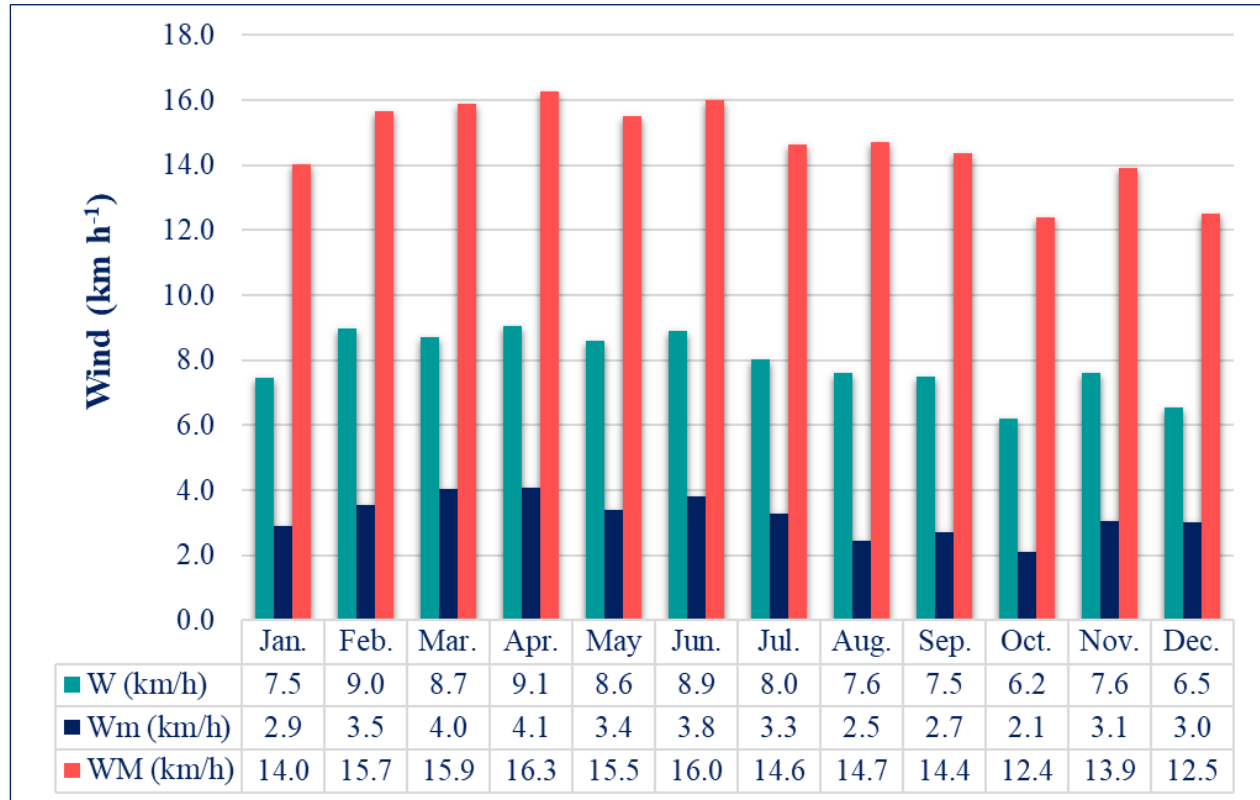


Fig. 22 Monthly mean wind speed variations in the study area during the 2012–2022 period.

II.6. Site geology

Cheliff reservoir area is characterized by Miocene and Lower Quaternary sedimentary formations, with more recent deposits of alluvium, colluvium, and terraces. The main lithological units include river alluvium (fine sands, sandy silts, and rounded pebbles), ancient terrace alluvium (silty to sandy clays with lenticular sandy interbeds), consolidated sandy–silty clays, and colluvium with carbonate sandstone blocks in a sandy–clayey matrix. On the right bank, quartz-rich carbonate sandstones, strongly folded and fractured with interbedded organogenic limestones, are present. The sequence also includes marls alternating with sandstones and gypsum (highly weathered and fractured), overlying fresher blue marls of the substratum. Structurally, the site is complex. (Sembenelli & Poletini, 2010; Bouderbala & Gharbi, 2017; National Agency for Dams and Transfers [N.A.D.T.], 2017).



However, Kerrada reservoir site is located on a syncline composed of alternating sandstone and marl beds, intersected by numerous faults and joints, mainly within the sandstone. Dam rests on sandstone on the left bank and on partially decompressed marls with thin sandstone layers on the right bank. Sandstone layers are permeable to highly permeable due to fracturing, especially on the left bank where outcropping sandstones are in direct contact with reservoir water, creating a high risk of significant seepage and bypass flows unless carefully treated. On the right bank, sandstone is overlain by thick marl, though seepage may still occur from an outcropping sandstone bed upstream of the dam axis. Construction materials are of good quality, but clays are highly plastic and sandstones fragile. Over time, prolonged clay consolidation and sandstone disintegration under submersion are expected to cause significant and long-lasting settlements (Sembenelli & Poletini, 2010; National Agency for Dams and Transfers [N.A.D.T.], 2018).

II.7. Vegetation characteristics

Cheliff and Kerrada reservoirs are predominantly surrounded by agricultural lands, where cereals and market gardening crops such as artichokes, watermelon, potatoes, and melon are cultivated (Djezzar, 2015). However, Cheliff reservoir, supports a higher vegetation diversity, particularly along its upstream banks and on the right margin. In Mostaganem study area sector, the dominant plant families are Asteraceae, Fabaceae, Brassicaceae, and Lamiaceae, with Asteraceae exhibiting the highest species richness and endemism (Mama et al., 2022). Endemic and regionally restricted species include *Bituminaria tunetana*, *Brassica fruticulosa* ssp. *glaberima*, *Erysimum semperflorens* ssp. *elatum*, *Senecio pinguiculus*, *Malcolmia arenaria* s.s., and *Salvia balansae* (Mostari et al., 2020; Mama et al., 2022). Upstream of Cheliff reservoir, the banks are bordered by double linear stands of *Tamarix Africana* with an herbaceous understory of *Otospermum glabrum*, alongside steppe-like dry grasslands rich in Maghreb and Algero-Moroccan endemic Brassicaceae (Mostari et al., 2020). The broader vegetation cover includes woodland formations dominated by junipers (*Juniperus oxycedrus*, *J. phoenicea*), Barbary thuya (*Tetraclinis articulata*), Aleppo pine (*Pinus halepensis*), and stone pine (*Pinus pinea*) (Santa et al., 1958; Houari et al., 2022). Riparian and aquatic vegetation is also well developed, represented by reeds, cattails, *Myriophyllum alterniflorum*, *Sparganium erectum*, and *Schoenoplectus lacustris*, which can exceed 2 m in height (Fig. 23). In contrast, Kerrada reservoir is characterized by lower floristic diversity and is almost entirely surrounded by cultivated lands without significant natural vegetation cover.

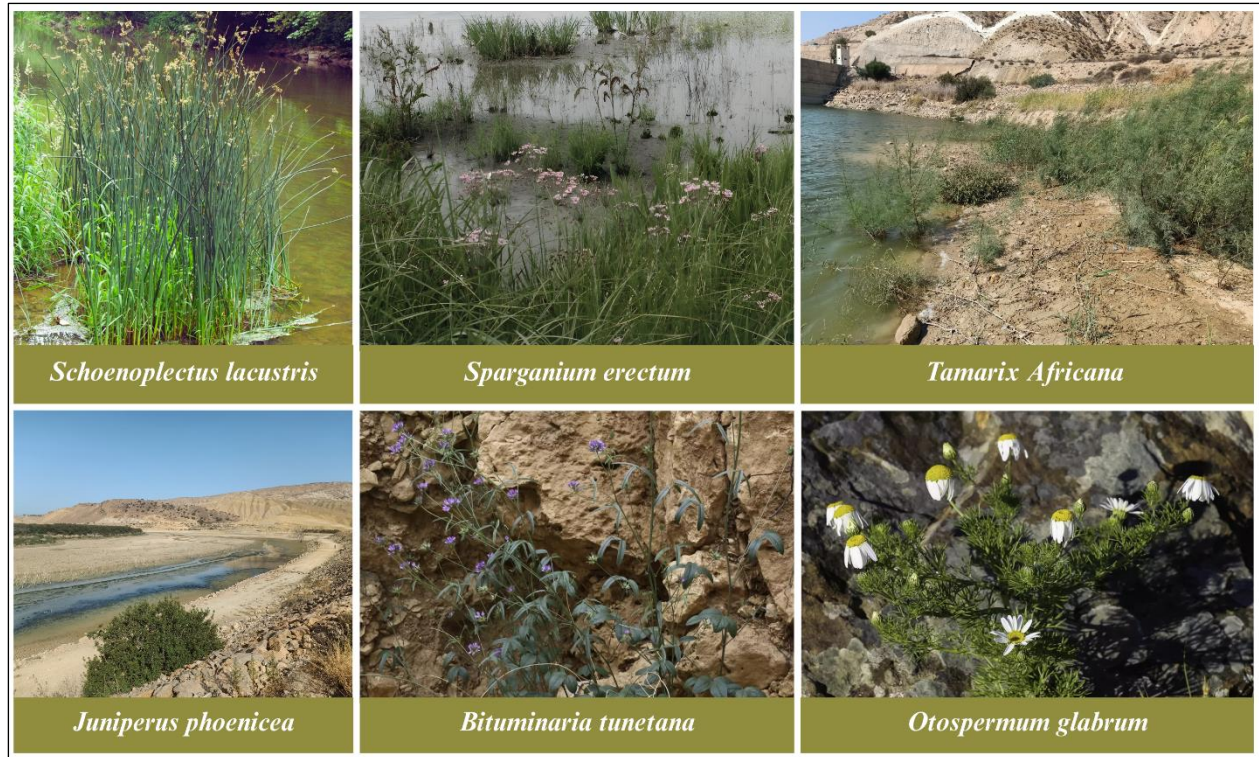


Fig. 23 Riparian and surrounding vegetation in the study area.

II.8. Anthropogenic pressures

Mostaganem, with more than 14,000 ha of agricultural land, is considered the province with the highest fertilizer consumption in Algeria, with an annual average of 8,375 T (Kies et al., 2018). Cheliff and Kerrada reservoirs are subjected to multiple anthropogenic pressures, mainly associated with the surrounding agricultural landscape. In Cheliff reservoir, fishing activity is practiced year-round, and nets are frequently observed in the waterbody, highlighting a continuous exploitation of fish resources. In contrast, fishing is strictly prohibited in Kerrada reservoir. On the left bank of Cheliff reservoir, the discharge of untreated urban wastewater, along with fertilizer inputs and runoff from nearby agricultural fields, contributes significantly to nutrient enrichment and water contamination. Cheliff reservoir is particularly affected by untreated wastewater and agricultural runoff, resulting in elevated nitrate and ammonium concentrations (Bettahar et al., 2009; Kies & Kerkouf, 2014). Numerous pumps are also installed along its banks to divert water for irrigation, further intensifying agricultural pressure on the system. Consequently, Cheliff reservoir is more exposed to contamination from fertilizers and domestic effluents, whereas Kerrada reservoir is mainly influenced by agricultural runoff, though at a lower intensity and more localized upstream. Moreover, the lack of wastewater treatment systems in rural areas contributes to microbiological contamination of water bodies, as observed in several lakes and reservoirs in



northeastern Algeria ([Majour et al., 2024](#)). Although Cheliff reservoir faces significant contamination challenges, Kerrada's stricter regulations may provide a useful model for mitigating agricultural impacts on water quality. However, the overall agricultural practices in the region still pose risks to both reservoirs, underscoring the need for improved management strategies.



Chapter III: Environmental parameters of the studied ecosystems





This chapter examines the spatial and seasonal patterns of physicochemical parameters in the studied reservoirs. By integrating descriptive analyses with principal component analysis (PCA), we aim to identify the key environmental drivers that shape water quality dynamics and assess how morphometric and hydrological differences influence ecosystem responses to climatic and anthropogenic pressures. Understanding these parameters provides essential context for interpreting later plankton community structure and evaluating the impact of environmental variability on overall ecosystem functioning.

III.1. Material and methods

III.1.1. Sampling procedure

To provide a comprehensive assessment of water quality and its temporal variations, water sampling was conducted monthly from November 2020 to October 2022. Four sites were selected in each reservoir (Cheliff reservoir: ST1, ST2, ST3, and ST4; Kerrada reservoir: ST5, ST6, ST7, and ST8) to capture spatial variability along the gradient from the tail to the dam (Fig. 24, Fig. 25). Site selection was based on several factors, including reservoir depth, presence of vegetation, inflows from main tributaries, upstream and downstream positions, and proximity to agricultural land (Table 2). At each site, water samples were collected from three layers: surface, middle, and near-bottom (1 m above the sediment), according to the depth of the station.

Meteorological data on monthly rainfall and air temperature for 2020, 2021, and 2022 were obtained from the Algerian National Meteorological Office (O.N.M.) ([Algerian National Meteorological Office \[ONM\], 2022](#)) to evaluate their potential influence on hydrodynamic processes and nutrient enrichment in the reservoirs.

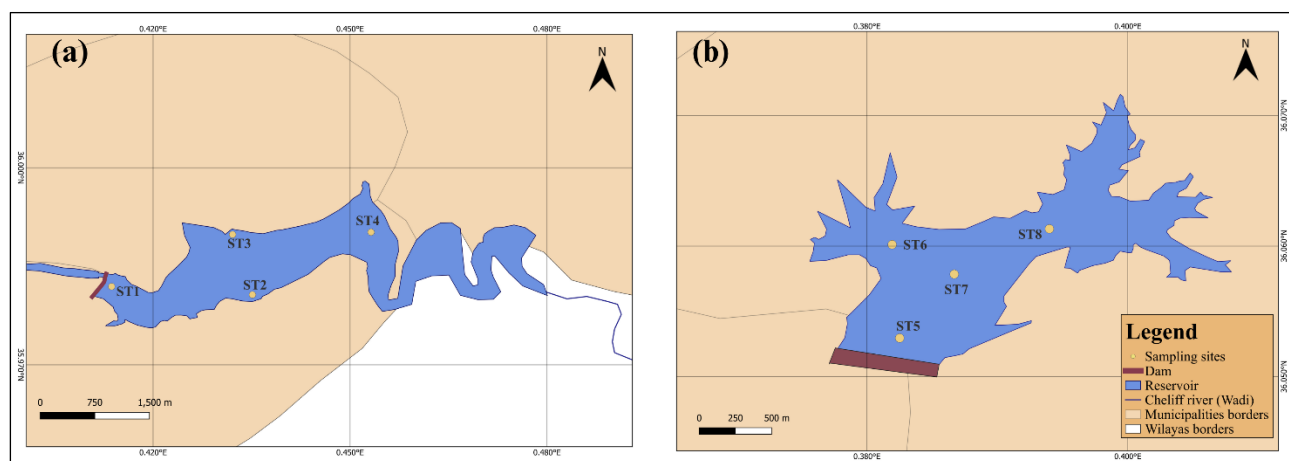


Fig. 24 Sampling sites in (a) Cheliff reservoir and (b) Kerrada reservoir.



Table 2 Characteristics of the sampling sites in Cheliff and Kerrada Reservoirs (Baitiche et al., 2025).

Reservoir	Sites	Characteristics	Coordinates	Depth (m)
Cheliff	ST1	Dam side.	35°59'00.9"N 0°24'48.4"E	Max = 12 Min = 4
	ST2	Influenced by farmland and sewage effluent.	35°58'51.1"N 0°26'09.8"E	Max = 4 Min = 0.3
	ST3	Bare of vegetation.	35°59'25.9"N 0°25'42.6"E	Max = 4.5 Min = 0.2
	ST4	Upstream side; this is the shallowest station in the reservoir. The location was selected to ensure study continuity in case of summer drought. It is characterized by submerged dead eucalyptus trees and farmland on the left bank.	35°59'08.0"N 0°27'18.0"E	Max = 2.5 Min = 0
Kerrada	ST5	Intake side of the reservoir (water intake tower).	36°03'05.7" N, 0°22'56.9" E	Max = 132 Min = 13
	ST6	Characterized by surrounding farmland.	36°03'40.6" N, 0°22'53.1" E	Max = 40 Min = 5
	ST7	Located in the middle of the reservoir and features a small island.	36°03'22.2" N, 0°23'05.6" E	Max = 50 Min = 8
	ST8	Characterized by agricultural land and shallower than the other stations.	36°03'39.7" N, 0°23'37.00" E	Max = 25 Min = 0

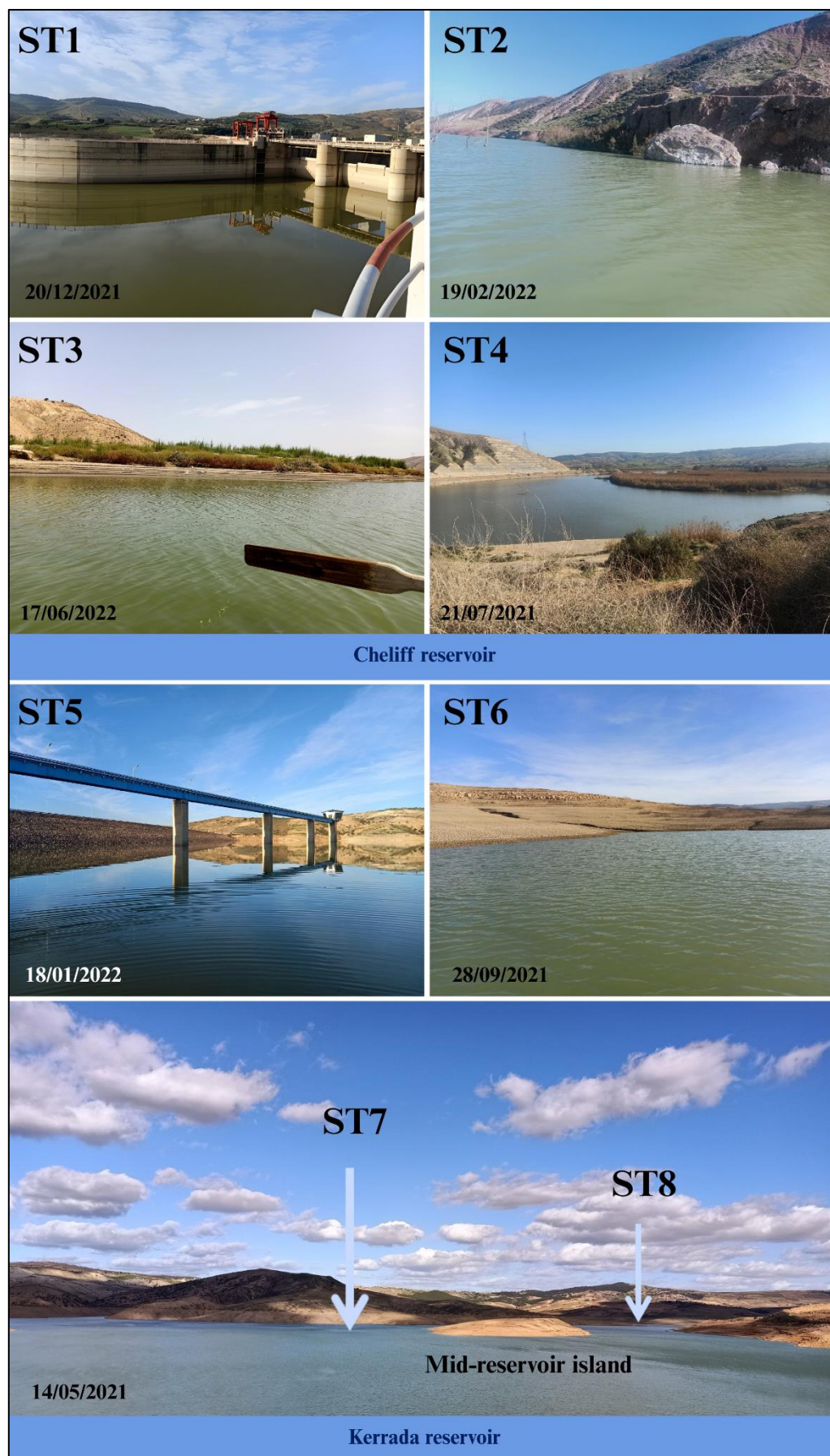


Fig. 25 Photographs of sampling sites in Cheliff and Kerrada reservoirs.



III.1.2. Physicochemical analyses

For limnological characterization, *in situ* measurements were performed at each sampling site, and 15 environmental parameters were measured. Reservoir depth was determined using a sonar-based DIVE-SCANT gauge in the deeper Kerrada reservoir and a graduated weighted line in the shallower Cheliff reservoir (Fig. 26; Fig. S1, Supplementary material). Physicochemical parameters, including water temperature ($^{\circ}\text{C}$), dissolved oxygen (DO ; mg L^{-1}), salinity (Sal), pH, turbidity (Turb; NTU), and total dissolved solids (TDS; mg L^{-1}), were recorded with a multiparametric probe (Hanna GPS HI 9829). Water samples from different depths were collected using a Van Dorn bottle (5 L) and immediately transferred into pre-cleaned 2000 mL polyethylene bottles to avoid contamination. Samples were stored at 4°C in a cooler and transported to the SEAAL laboratory in Kouba-Algiers, where analyses were carried out within 24 hours. Nutrient concentrations, namely phosphate (PO_4 ; mg L^{-1}), nitrate (NO_3 ; mg L^{-1}), nitrite (NO_2 ; mg L^{-1}), and ammonium (NH_4 ; mg L^{-1}), were determined spectrophotometrically following the protocols described by Golterman et al. (1978) and Hansen and Koroleff (1999). Organic matter (OM; mg L^{-1}) was analyzed using the permanganate oxidation method (Rodier, 2009), while particulate matter (PM; mg L^{-1}) was quantified gravimetrically according to the standard method of APHA (2005).

Further, water transparency (SD; m) was assessed using Secchi depth measurements, and chlorophyll-a (CHL; $\mu\text{g L}^{-1}$) concentrations were determined as indicators of phytoplankton biomass. Chlorophyll-a was determined following the standard spectrophotometric protocol described by APHA (2005), and concentrations were calculated using the equations of Jeffrey and Humphrey (1975). However, these variables are not presented in this chapter and are addressed in subsequent chapters.

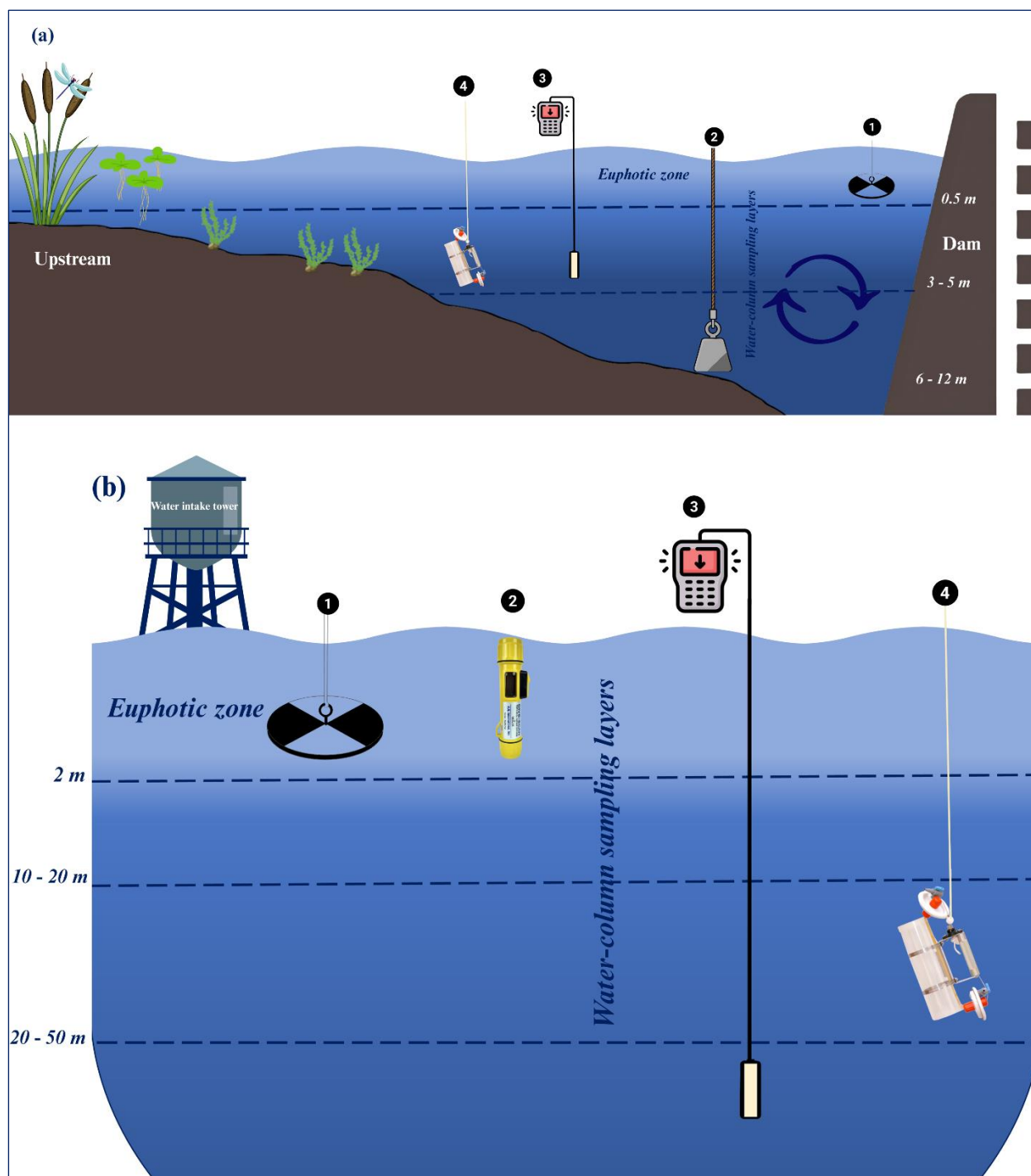


Fig. 26 Schematic representation of the sampling strategy for physicochemical measurements and water collection. **(a)** Cheliff shallow reservoir; **(b)** Kerrada deep reservoir. **(1–4)** Ordered steps of the sampling procedure.



III.1.3. Data analysis

Statistical analyses were performed on datasets of physicochemical parameters collected throughout the study period at all sampling stations and depth levels in each reservoir. Data were grouped by seasons and expressed as mean values.

To assess spatio-temporal differences in physicochemical variables, a Mann–Whitney U (Wilcoxon rank-sum) test was used to compare the two reservoirs (Cheliff and Kerrada), while Kruskal–Wallis test was applied to evaluate seasonal variations and differences among sampling stations and depth layers within each reservoir. These nonparametric tests were selected because the data did not meet the assumptions of normality and homogeneity of variances. All statistical tests were performed at a 5% significance level.

In addition, abiotic data were analyzed using Principal Component Analysis (PCA) to identify the most influential environmental variables shaping spatio-temporal patterns and to explore relationships among physicochemical parameters. All statistical analyses and graphical visualizations were carried out in R software (version 4.3.2), employing the *vegan* and *ggplot2* packages.



III.2. Results

III.2.1. Temperature and hydrologic variation

Although comprehensive hydrological and climatic modeling was not within the scope of this study, monthly trends were examined to contextualize the observed interannual variability (Fig. 27). The comparison of air temperature and precipitation between the two sampling years (November 2020 to October 2022) showed no significant interannual differences in air temperature. In contrast, precipitation varied significantly between the two years (Welch's t -test, $p < 0.05$), with higher values observed in the second year.

Mediterranean and semi-arid climates exhibit distinct patterns in temperature and precipitation variability. Temperature remains relatively stable from year to year, primarily influenced by consistent factors such as solar radiation and large-scale atmospheric circulation, which explains why no significant interannual differences were detected in this study. In contrast, precipitation is highly irregular and strongly affected by stochastic atmospheric events, leading to pronounced fluctuations, particularly in semi-arid regions. This variability is evident in studies showing that rainfall can differ markedly from year to year, and that average annual rainfall decreases from northern to southern areas due to altitude and relief effects. The higher precipitation observed in the second year likely reflects this natural interannual variability and the occurrence of more frequent or intense rainfall events during that period (Meddi & Hubert, 2003; Longobardi & Villani, 2010; Ozturk et al., 2015; Elmeddahi et al., 2016; Achour et al., 2020; Trambly et al., 2020; Achite et al., 2021; Zaibak & Meddi, 2022; Tzanis, 2023; Campos et al., 2025).

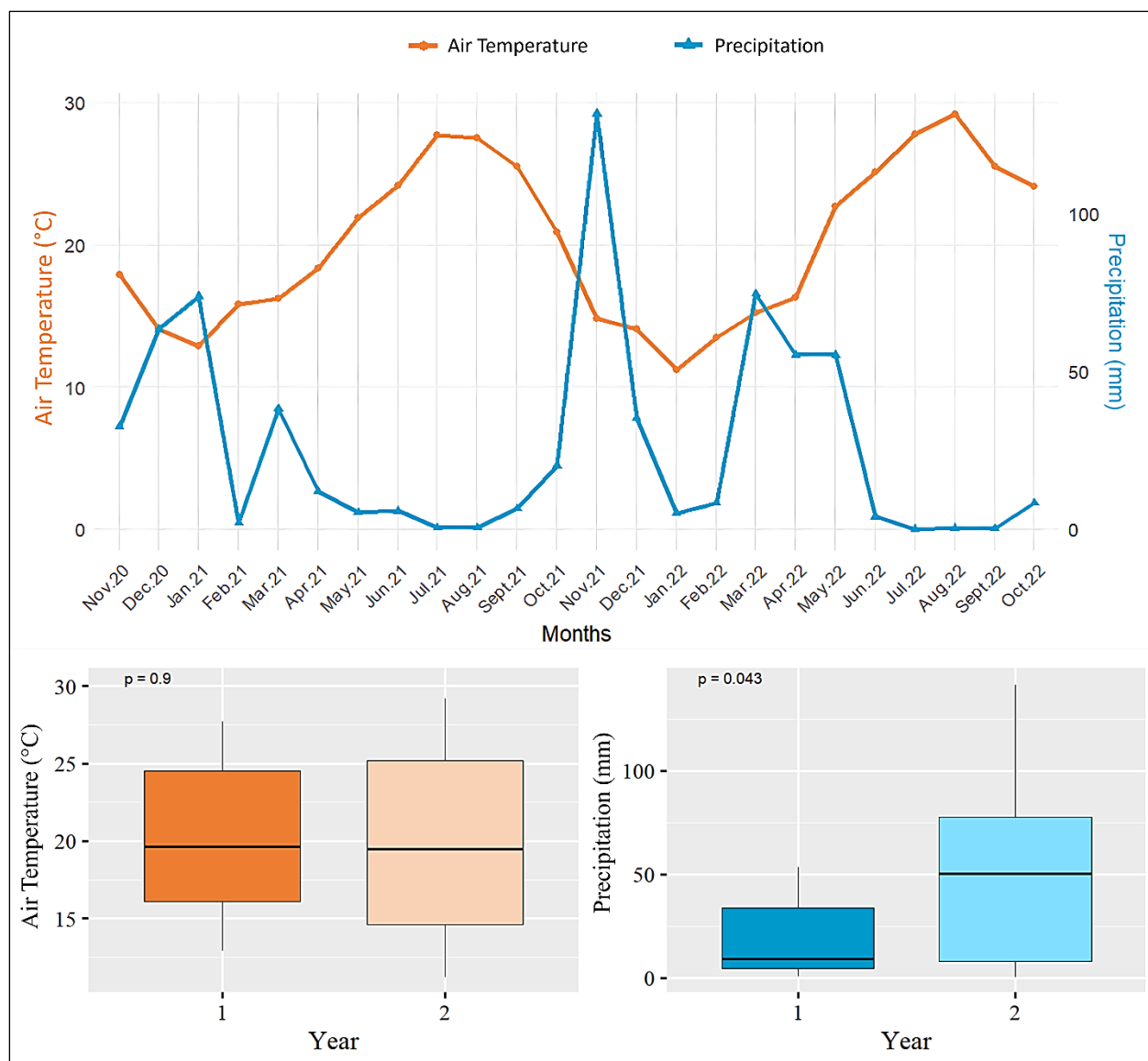


Fig. 27 Monthly trends and interannual differences in air temperature and precipitation (Nov. 2020–Oct. 2022) in the study area. Welch's *t*-Test analysis with 95% Confidence Interval (Baitiche et al., 2025).

III.2.2. Physicochemical parameters assessment

The data (Table 3; Fig. 28) revealed that Cheliff reservoir consistently exhibited higher mean values for all physicochemical parameters except nitrates, for which higher concentrations were recorded in Kerrada reservoir (NO_3^- ; 14.25 mg L^{-1}).

According to Fig. 29 and Fig. 30, temperature showed clear seasonality, with maxima in Summer 1 (36.1°C at Cheliff reservoir; 27.88°C at Kerrada reservoir) and minima in Winter 2 (10.6°C and 12.1°C , respectively). pH also varied significantly (Kruskal–Wallis test, $p < 0.05$), ranging from 7.15 at Kerrada reservoir in Autumn 1 to 13.23 at Cheliff reservoir in Summer 1. DO



fluctuated widely, with Cheliff reservoir showing both the lowest (2.36 mg L⁻¹, Summer 2) and highest values (20.03 mg L⁻¹, Spring 1). Salinity and TDS were generally higher in the dry season and throughout the first sampling year, with higher values in Cheliff reservoir (1.37 PSU; 1284 mg L⁻¹) compared to Kerrada reservoir (1.05 PSU; 1018 mg L⁻¹). PM, turbidity, and OM also varied significantly (Kruskal–Wallis test, $p < 0.05$). Peak values were recorded in Cheliff reservoir (PM: 185.6 mg L⁻¹ in Winter 1; turbidity: 96.5 NTU in Spring 1; OM: 24.1 mg L⁻¹ in Summer 1), whereas the lowest values were observed in Kerrada reservoir (PM: 1.4 mg L⁻¹, Spring/Summer 2; turbidity: 2.2 NTU, Winter 2; OM: 2.3 mg L⁻¹, Spring 1).

Nutrient levels (NO₂⁻, NH₄⁺, PO₄³⁻) were significantly higher in Cheliff reservoir, although seasonal differences were less pronounced (Kruskal–Wallis test, $p \geq 0.01$) for NO₂⁻ and NH₄⁺. Maximum concentrations occurred in Summer 1 for NO₂⁻ (5.63 mg L⁻¹) and NH₄⁺ (9.04 mg L⁻¹), and in Autumn 1 for PO₄³⁻ (2.19 mg L⁻¹). In Kerrada reservoir, nutrients levels remained low, with maxima of NO₂⁻ (0.75 mg L⁻¹, Spring 2) and NH₄⁺ (0.88 mg L⁻¹, Summer 2). PO₄³⁻ showed no significant seasonal variation (Kruskal–Wallis test, $p > 0.05$), peaking at 0.54 mg L⁻¹ in the first sampling year and in Autumn 2, and approaching null values during Summer 2. For nitrates, no significant seasonal variation was detected in Cheliff reservoir (Kruskal–Wallis test, $p > 0.05$), with low mean concentrations (10.38 mg L⁻¹). Conversely, Kerrada reservoir exhibited significant seasonal variability, with maximum concentrations reaching 33.2 mg L⁻¹ in Summer 1.

Table 3 Average values of physicochemical parameters measured in Cheliff and Kerrada reservoirs (2021 - 2022) (Baitiche et al., 2025).

Environmental parameters	Abbreviation	Cheliff reservoir			Kerrada reservoir		
		Min.	Max.	Average $\pm$ Stand. dev	Min.	Max.	Average $\pm$ Stand. dev
Temperature (°C)	Temperature	10.60	36.10	20.73 $\pm$ 6.16	12.10	27.88	18.52 $\pm$ 4.51
pH	pH	7.76	13.23	8.80 $\pm$ 0.72	7.15	9.53	8.15 $\pm$ 0.38
Salinity (PSU)	Sal	0.67	2.19	1.37 $\pm$ 0.39	0.80	1.43	1.05 $\pm$ 0.22
Total Dissolved Solids (mg L ⁻¹)	TDS	666	2103	1284 $\pm$ 359	773	1378	1018 $\pm$ 206
Dissolved Oxygen (mg L ⁻¹)	DO	2.36	20.03	9.71 $\pm$ 3.98	2.37	15.48	8.00 $\pm$ 2.61
Ammonium (mg L ⁻¹)	NH ₄	0.10	9.04	1.80 $\pm$ 1.89	0.01	0.88	0.22 $\pm$ 0.17
Nitrite (mg L ⁻¹)	NO ₂	0.03	5.63	0.86 $\pm$ 1.36	0.02	0.75	0.15 $\pm$ 0.14
Nitrate (mg L ⁻¹)	NO ₃	1.45	41.10	10.38 $\pm$ 8.08	1.23	33.2	14.25 $\pm$ 7.84
Orthophosphate (mg L ⁻¹)	PO ₄	0.01	2.19	0.39 $\pm$ 0.57	0.00	0.54	0.07 $\pm$ 0.10
Organic Matter (mg L ⁻¹)	OM	3.4	24.1	10.0 $\pm$ 4.8	2.3	7.3	4.3 $\pm$ 1.2
Particulate Matter (mg L ⁻¹)	PM	28.2	185.6	86.3 $\pm$ 35.2	1.4	40.1	7.4 $\pm$ 8.3
Turbidity (NTU)	Turb	7.1	96.5	27.2 $\pm$ 17.1	2.2	30.4	6.5 $\pm$ 4.1

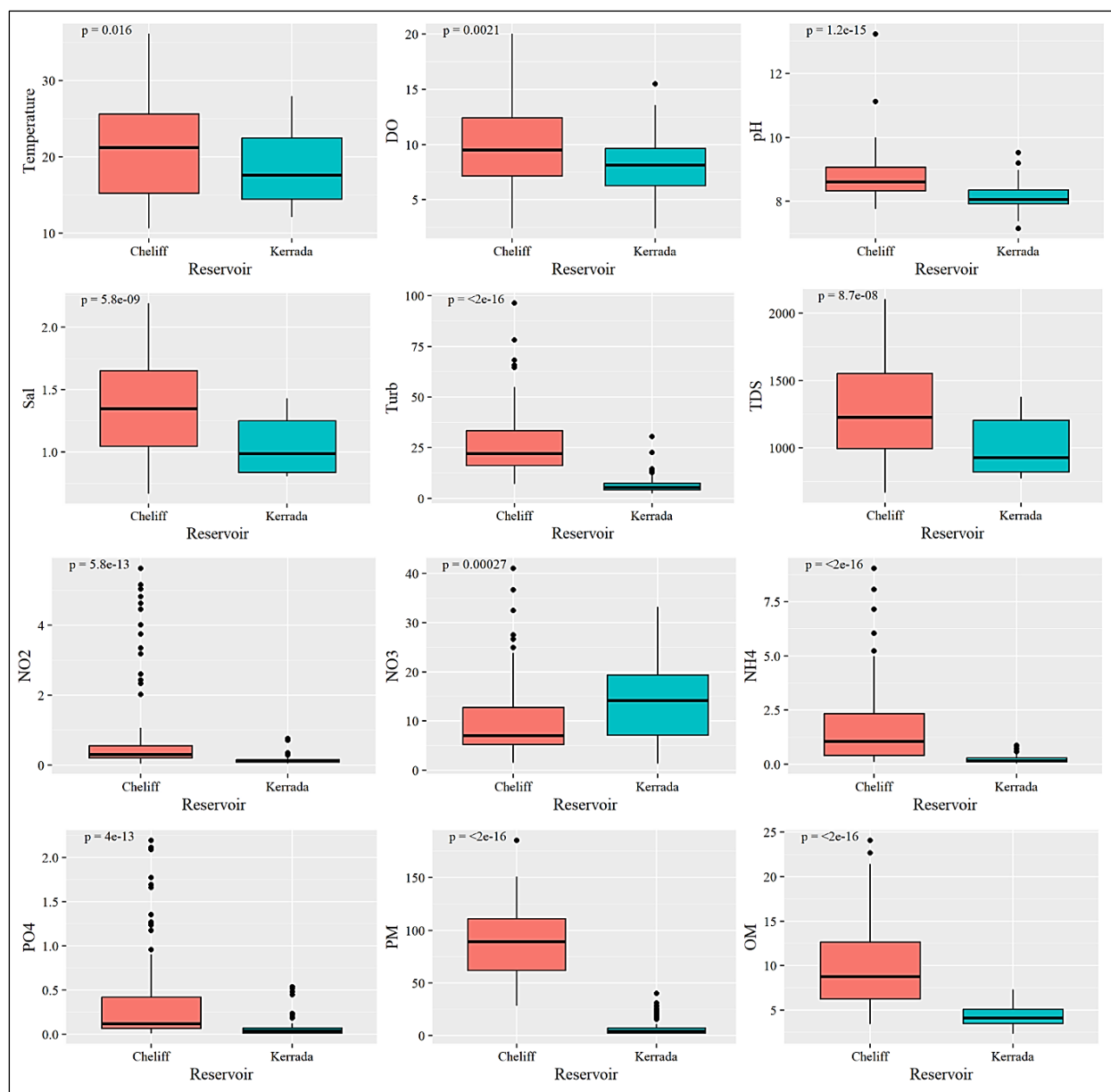


Fig. 28 Comparison of physicochemical variables between Cheliff and Kerrada reservoirs (Wilcoxon test, 95% confidence interval).

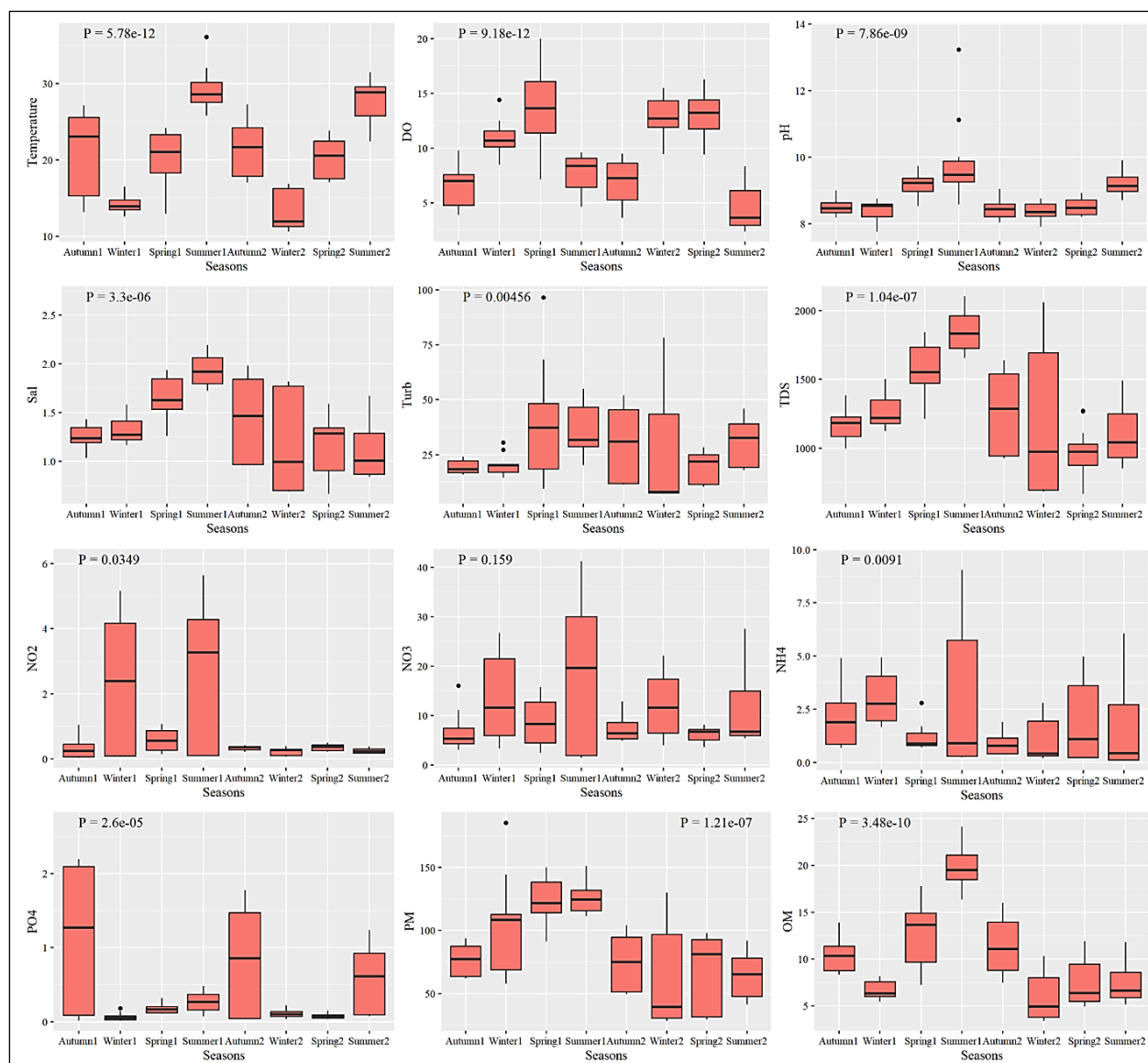


Fig. 29 Seasonal variation of physicochemical parameters in Cheliff reservoir (Kruskal–Wallis test, 95% confidence interval). Seasonal grouping of sampling periods: Autumn1: Nov. 2020, Sept. 2021, and Oct. 2021; Winter1: Dec. 2020, Jan. 2021, and Feb. 2021; Spring1: Mar. 2021, Apr. 2021, and May 2021; Summer1: Jun. 2021, Jul. 2021, and Aug. 2021; Autumn2: Nov. 2021, Sept. 2022, and Oct. 2022; Winter2: Dec. 2021, Jan. 2022, and Feb. 2022; Spring2: Mar. 2022, Apr. 2022, and May 2022; Summer2: Jun. 2022, Jul. 2022, and Aug. 2022.

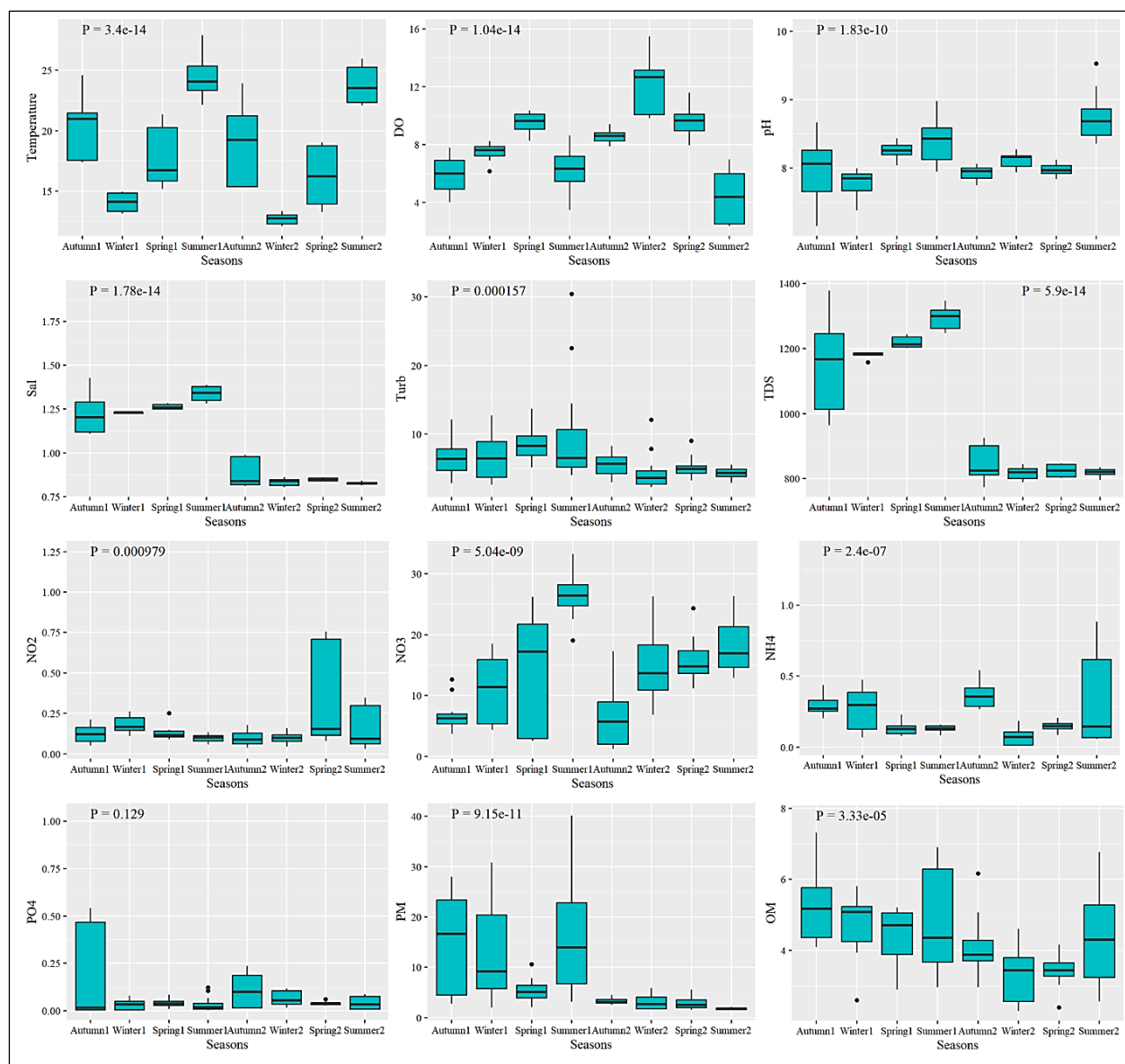


Fig. 30 Seasonal variation of physicochemical parameters in Kerrada reservoir (Kruskal–Wallis test, 95% confidence interval). Seasonal grouping of sampling periods: Autumn1: Nov. 2020, Sept. 2021, and Oct. 2021; Winter1: Dec. 2020, Jan. 2021, and Feb. 2021; Spring1: Mar. 2021, Apr. 2021, and May 2021; Summer1: Jun. 2021, Jul. 2021, and Aug. 2021; Autumn2: Nov. 2021, Sept. 2022, and Oct. 2022; Winter2: Dec. 2021, Jan. 2022, and Feb. 2022; Spring2: Mar. 2022, Apr. 2022, and May 2022; Summer2: Jun. 2022, Jul. 2022, and Aug. 2022.

III.2.3. Spatial and vertical physicochemical patterns across sampling stations and depths

Most physicochemical parameters did not differ significantly among stations (Kruskal–Wallis test; $p > 0.05$), indicating relatively homogeneous environmental conditions across stations in both Cheliff and Kerrada reservoirs (Fig. S2, and Fig. S3; Supplementary material). Despite the absence of statistically significant spatial variation, descriptive analyses revealed distinct station-specific tendencies.



In Cheliff reservoir, ST2 consistently exhibited the highest concentrations for most physicochemical parameters, likely reflecting its proximity to agricultural runoff and sewage effluents. Nutrient levels were also markedly elevated at this station. ST3 generally followed with comparatively high values, except for PM and turbidity, which reached their maximum levels at ST4. An exceptional event was recorded in Summer 1 at ST3, when pH values exceeded 13, coinciding with a critically low DO concentration (2.36 mg L^{-1}). In contrast, this station exhibited high DO values in Spring 1, indicating pronounced temporal variability driven by episodic environmental conditions.

In Kerrada reservoir, the lack of significant differences among stations was consistent across all physicochemical variables except for turbidity. Elevated turbidity values were recorded at ST8, the shallowest site in the reservoir, which is surrounded by agricultural land and receives substantial surface runoff. NO_2 displayed a borderline pattern but did not reach statistical significance (p -value slightly above 0.05); values at ST8 were marginally higher, though variability prevented a significant outcome. NH_4^+ also showed no significant spatial differences and remained low across stations, suggesting limited inputs from organic decomposition or anthropogenic pollution. Overall, these patterns may reflect slight nutrient enrichment at ST8, although not at a level detected by the statistical tests.

Comparisons of physicochemical conditions across depth layers revealed contrasting vertical patterns between Cheliff and Kerrada reservoirs. In Cheliff reservoir, which functions as a shallow, polymictic diversion ecosystem, continuous vertical mixing prevents the establishment of stable thermal or chemical stratification, as is typical of shallow polymictic ecosystems. Consequently, most physicochemical parameters did not differ significantly among depth layers (Kruskal–Wallis test; $p > 0.05$), with the exception of pH and OM (Fig. 31). Extremely high pH values were recorded at the surface (13.23), while the lowest values occurred at depth (7.48). OM concentrations also differed significantly, reaching their maximum values at the surface (24.1 mg L^{-1}) and minimum values at depth (2.4 mg L^{-1}). For the remaining parameters, concentrations generally increased from the surface toward deeper layers, except for temperature and DO, which exhibited their highest values at the surface, reflecting direct atmospheric exchange and solar heating.

In contrast, Kerrada reservoir exhibited a clear vertical gradient, with significant differences among depth layers for most physicochemical variables (Kruskal–Wallis test; $p < 0.05$) (Fig. 32).



Temperature, DO, pH, turbidity, and OM decreased progressively from the surface toward deeper layers, indicating the development of thermal stratification and reduced vertical mixing. Conversely, nutrient compounds (NO_2^- , NO_3^- , NH_4^+ , PO_4^{3-}) and particulate matter PM increased with depth, suggesting nutrient accumulation in bottom waters driven by sediment release, organic matter degradation, and reduced oxygen levels.

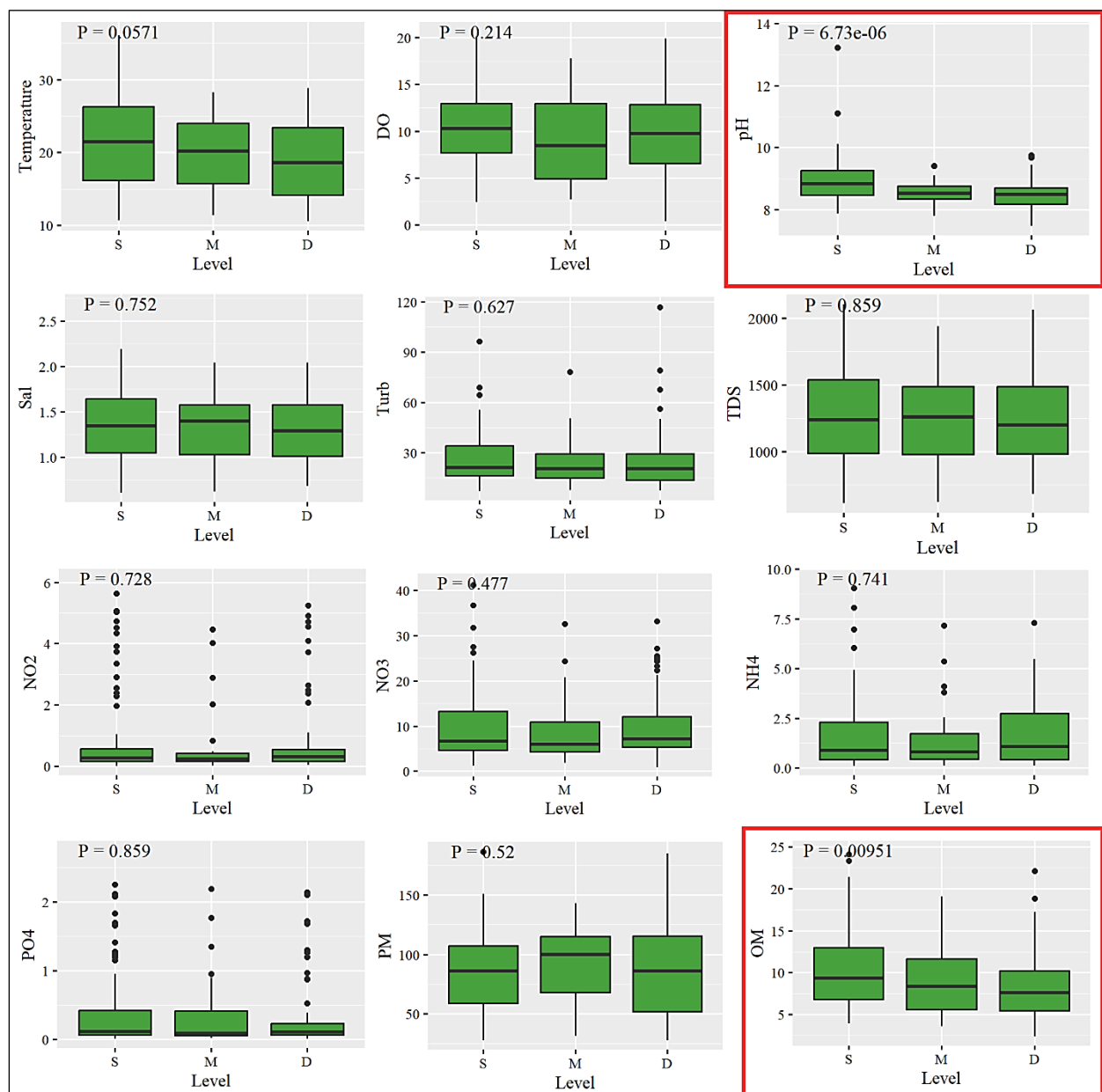


Fig. 31 Comparison of physicochemical parameters among depth layers in Cheliff reservoir (Kruskal–Wallis test; 95% confidence interval).

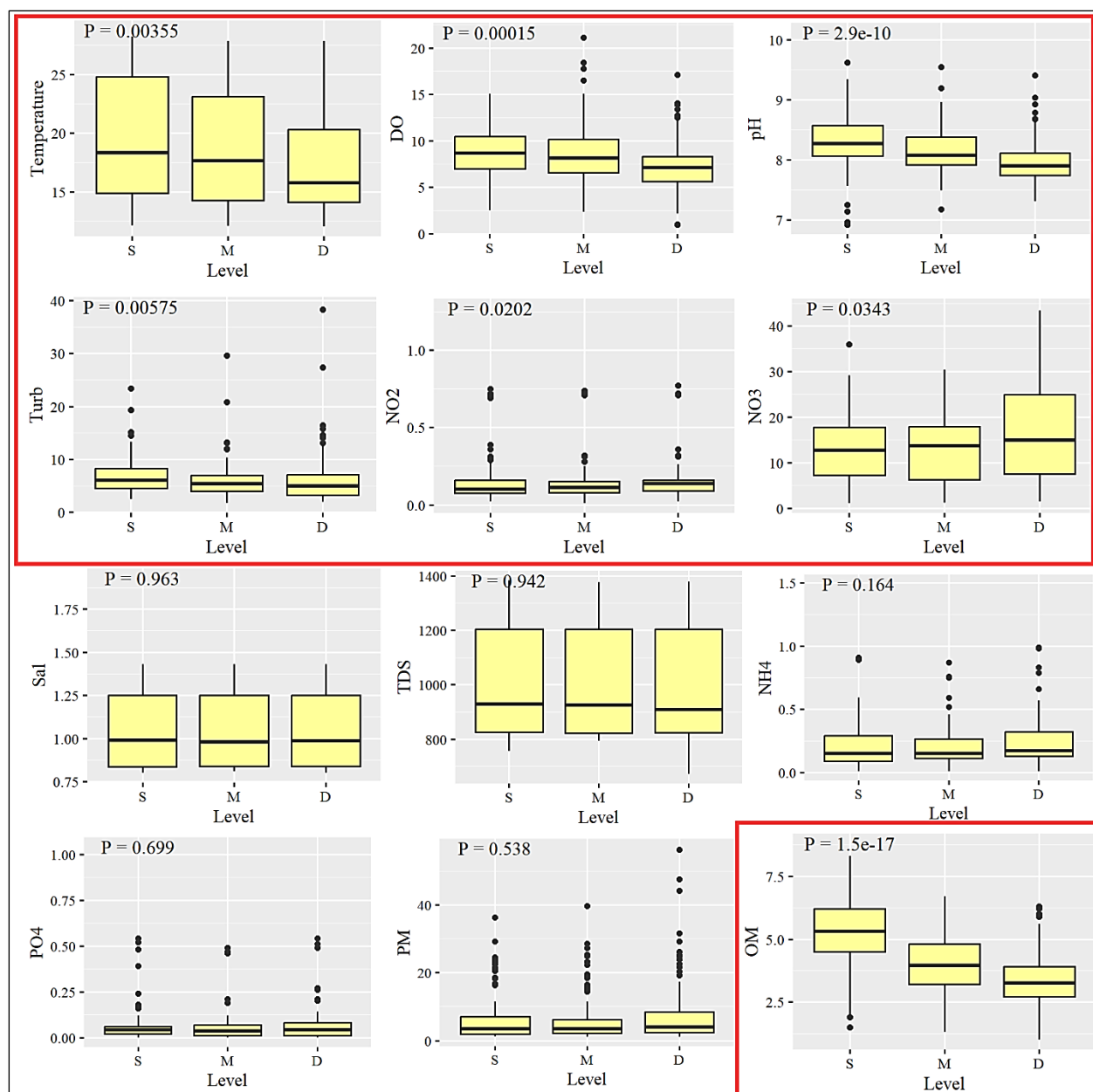


Fig. 32 Comparison of physicochemical parameters among depth layers in Kerrada reservoir (Kruskal–Wallis test; 95% confidence interval).

III.2.4. Thermal stratification and vertical dissolved oxygen patterns in Kerrada reservoir

Thermal stratification was observed only in Kerrada reservoir, whereas Cheliff reservoir, being a shallow diversion ecosystem with a polymictic regime, remained continuously mixed. In Cheliff reservoir, the limited depth prevents the development of stable density gradients, resulting in vertically homogeneous temperatures and DO concentrations throughout the year. Therefore, the following analysis focuses on Kerrada, a deep storage reservoir (maximum depth ~115 m).



Vertical profiles at Kerrada reservoir reveal distinct seasonal patterns in the thermal regime. A well-developed thermocline forms during spring and summer, typically initiating at 10–15 m depth and extending to 40–45 m, where the steepest thermal gradient occurs (Fig. 33). Summer heating raises surface temperatures above 25°C, while deeper layers (>40–45 m) remain between 12–14°C, reflecting strong stratification. During autumn, surface cooling progressively erodes the thermocline as the mixed layer deepens and vertical temperature gradients weaken. By late autumn to early winter, complete homogenization of the water column occurs and the thermocline disappears. Winter profiles indicate full mixing, with temperatures becoming uniform from the surface to ~50 m (11–15°C), and gradually decreasing to ~8–10°C below 50 m down to 100 m. Overall, thermocline formation typically begins in early spring, intensifies during summer, and progressively weakens throughout autumn, with complete destratification in winter. This seasonal cycle confirms a warm monomictic mixing pattern.

Seasonal DO profiles closely follow the seasonal thermal structure. In spring, high DO concentrations ($\approx 9\text{--}11\text{ mg L}^{-1}$) were recorded in the epilimnion (0–15 m), with peak values between 1–5 m within the euphotic zone. A moderate decline appeared in the metalimnion ($\approx 15\text{--}40\text{ m}$), associated with the onset of stratification and increased biological oxygen demand. Deeper layers exhibited the first signs of DO depletion as vertical mixing weakened. In summer, intensified stratification produced pronounced vertical DO gradients. Epilimnetic DO declined relative to spring ($\approx 4\text{--}6\text{ mg L}^{-1}$), likely due to increased respiration and reduced oxygen solubility at higher temperatures. A distinct DO minimum zone developed below 40–50 m ($\approx 3\text{--}5\text{ mg L}^{-1}$), resulting from restricted oxygen replenishment combined with sustained mineralization and heterotrophic activity in deep layers. The close correspondence between the steep thermal gradients and the DO decline highlights a classic summer stratified structure. During autumn, surface cooling deepened the mixed layer and reduced vertical stratification. DO concentrations became more uniform above ~40 m, while deep-water DO remained comparatively low due to cumulative oxygen consumption during stratified months. In winter, complete overturn re-oxygenated the entire water column, producing homogeneous DO values between $\approx 6\text{--}10\text{ mg L}^{-1}$ and fully aligning DO and temperature profiles.

The annual mean profile synthesizes these seasonal patterns. Surface temperatures ($\approx 19\text{--}20^\circ\text{C}$) gradually decline to 10–14°C between 40 and 100 m, indicating a moderate residual thermal gradient. DO concentrations decline progressively with depth, with a distinct inflection at 40–50 m corresponding to the typical seasonal DO depletion zone. The annual profile provides a



smoothed representation of the stratification–mixing cycle and corroborates the warm monomictic character of Kerrada reservoir, which exhibits pronounced stratification from spring through summer and complete mixing during winter.

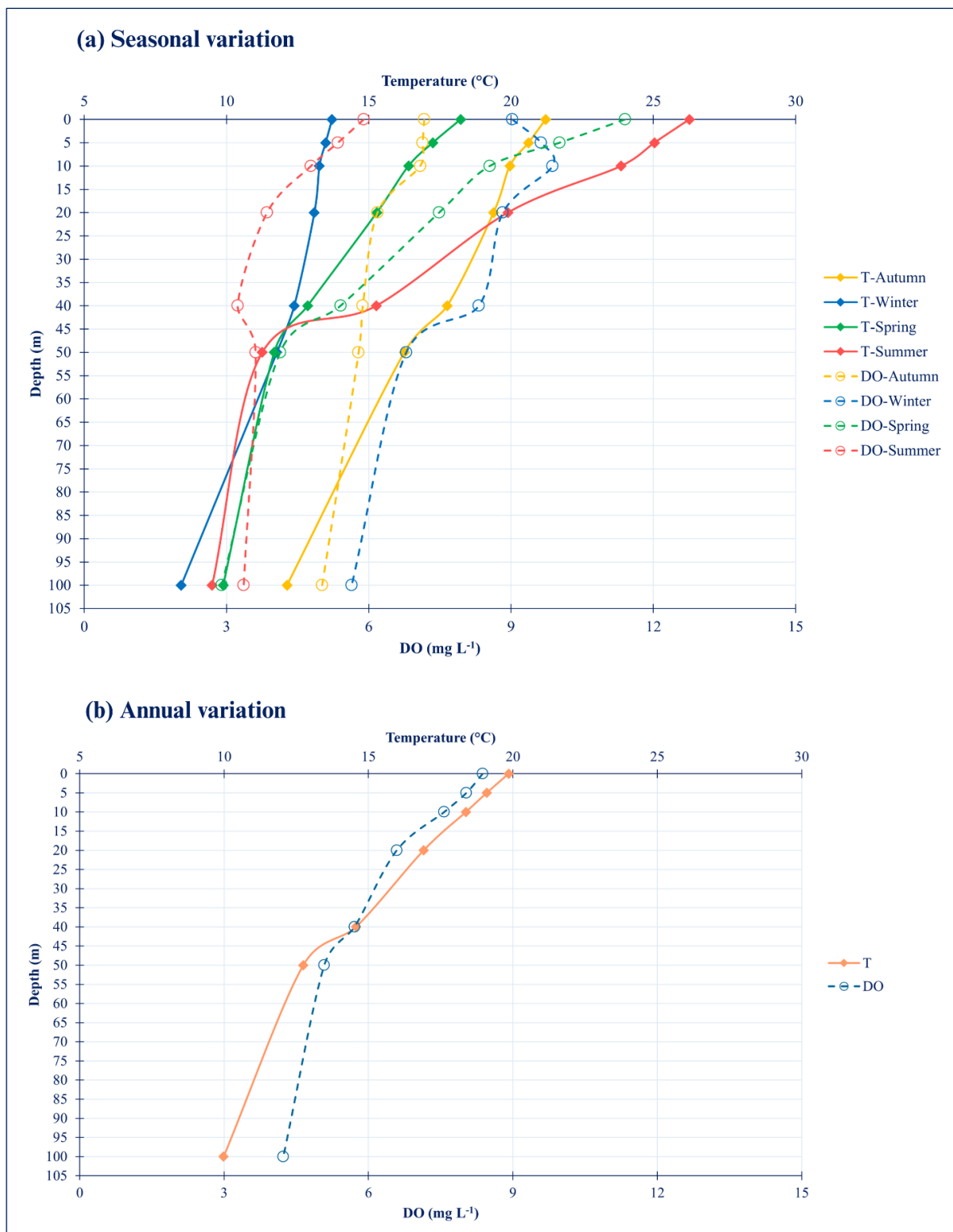


Fig. 33 Seasonal and annual variation of thermal stratification and vertical dissolved oxygen patterns in Kerrada reservoir during two years of sampling (Nov. 2020–Oct. 2022).



III.2.5. Seasonal correlation between physicochemical variables

To assess correlations among environmental parameters in each reservoir, a Principal Component Analysis (PCA) was performed on 12 variables plus depth (Fig. 34). In both reservoirs, two principal components with eigenvalues > 1 were retained, explaining 58.6% of the total variance in Cheliff reservoir and 43.6% in Kerrada reservoir. In Cheliff reservoir, the first component (PC1) accounted for 41.8% of the total variance, whereas in Kerrada reservoir it explained 28.4%. The second component (PC2) accounted for 16.8% and 15.2% of the variance in Cheliff and Kerrada reservoirs, respectively.

Seasonal variability was evident in both ecosystems. In Cheliff reservoir, summer samples were associated with high temperature, PO_4^{3-} , OM, pH, salinity, turbidity, and nutrient enrichment, whereas cooler seasons aligned with higher DO and depth. In Kerrada reservoir, seasonal variability was less pronounced: summer was linked to elevated temperature, NO_3^- , OM, PO_4^{3-} , and pH, while winter samples clustered with higher DO and depth. This distribution showed a tighter grouping compared to Cheliff reservoir, indicating greater seasonal stability. Autumn samples represented intermediate conditions along both axes, while spring samples clustered near the origin, mainly influenced by salinity, TDS, and PM.

The main contributors to Dimension 1 (PC1 / Dim-1) in Cheliff reservoir were salinity, TDS, OM, and PM (each $> 13\%$), while DO and PO_4^{3-} dominated Dimension 2 (PC2 / Dim-2), with DO contributing $> 30\%$. In Kerrada reservoir, Dim-1 was similarly driven by salinity, TDS, and PM (each $> 15\%$), whereas Dim-2 was shaped primarily by pH and temperature ($> 25\%$ each), followed by DO, salinity, and TDS.

These patterns indicate that Dim-1 in both reservoirs reflects gradients in water composition (salinity, dissolved solids, and particulates), whereas Dim-2 highlights oxygen and nutrient dynamics, particularly during seasonal transitions. Overall, Cheliff reservoir exhibited more pronounced seasonal fluctuations, while Kerrada reservoir maintained relatively stable conditions.

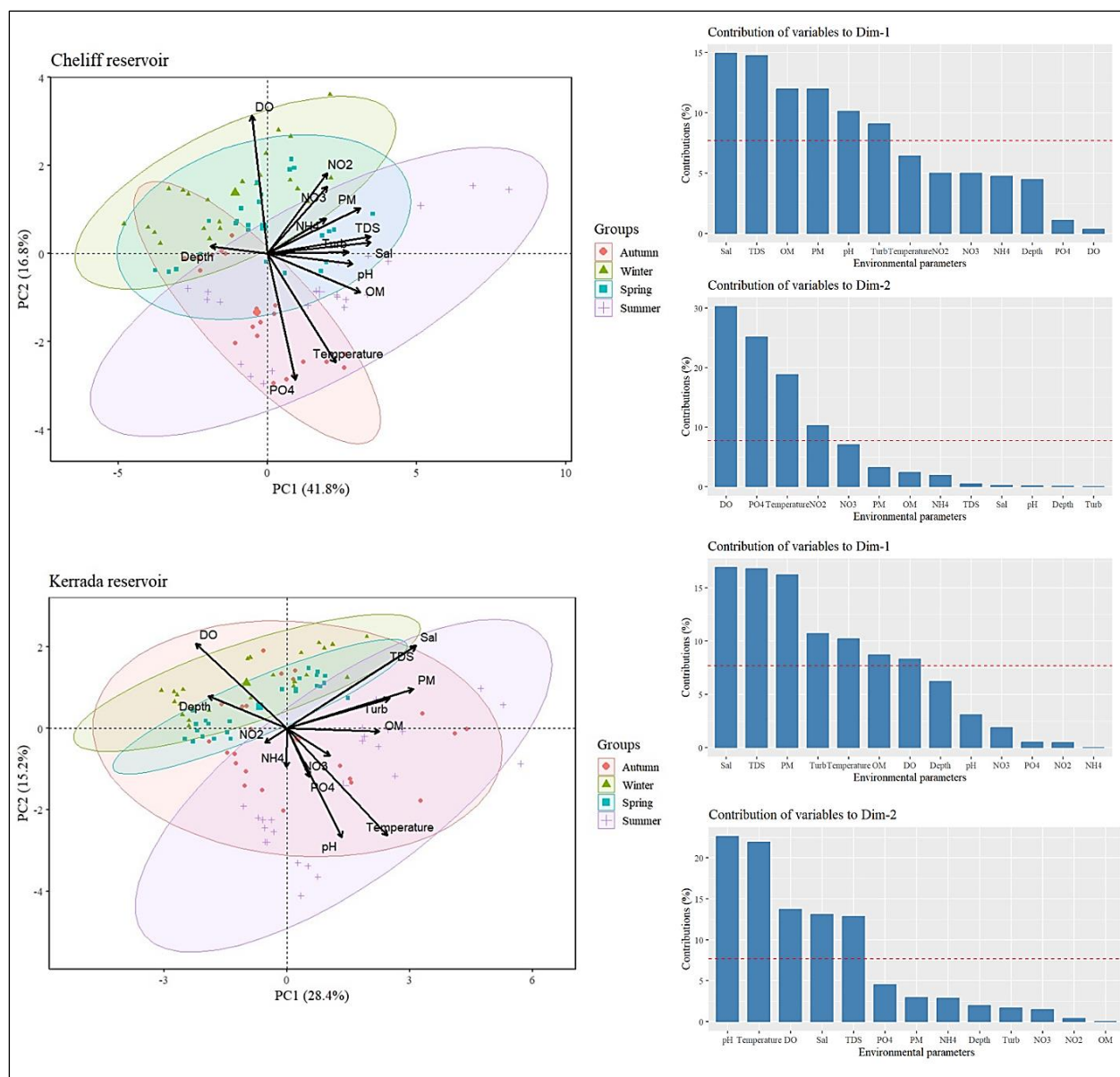


Fig. 34 Principal Component Analysis (PCA) of environmental factors in Cheliff and Kerrada reservoirs, showing the contribution of variables to each dimension. Physicochemical parameter codes are provided in Table 3.



III.3. Discussion

The present study revealed pronounced spatial, vertical, and seasonal contrasts in physicochemical parameters between the shallow Cheliff reservoir and the deeper Kerrada reservoir. These differences patterns were strongly shaped by morphometry, hydrological regime, and climatic variability, consistent with earlier findings from North African and Mediterranean reservoirs (Mhamdi et al., 1994; Cherbi et al., 2008; Nasri et al., 2016; Schilling et al., 2020; Soltani et al., 2020).

Cheliff reservoir consistently exhibited higher mean values of temperature, pH, salinity, TDS, PM, turbidity, and OM compared to Kerrada reservoir. These patterns can be attributed to its shallow depth and greater surface exposure, which enhance evaporation, sediment resuspension, and organic matter accumulation (Dokulil & Teubner, 2000; Naselli-Flores, 2000). Shallow reservoirs are particularly influenced by solar radiation and warm air temperatures, leading to unique thermal dynamics that differ from those in deeper reservoirs (Bu et al., 2022; Piccolroaz et al., 2024). In contrast, the greater depth of Kerrada reservoir provided a stabilizing effect, reducing seasonal variability and supporting predictable stratification patterns (Margalef, 1983; Tundisi & Matsumura-Tundisi, 2008).

In Cheliff reservoir, ST2's high physicochemical values are attributed to agricultural runoff and domestic wastewater inputs, consistent with previous studies (Harkat et al., 2011; Soltani et al., 2020; Guermazi et al., 2023). The extreme pH recorded at ST3 in July 2021, suggests an acute contamination event, potentially linked to algal bloom collapse under nutrient overload and subsequent oxygen depletion (Kies & Kerkouf, 2014). High PM and turbidity at ST4 can be attributed to sediment resuspension facilitated by its shallow depth, a common feature of polymictic ecosystems (Wetzel, 2001; Navarro et al., 2009; Benkaddour et al., 2019; Orduna et al., 2024). Kerrada reservoir, in contrast, exhibited strong spatial homogeneity supported by its large volume and deep-water buffering capacity (Naselli-Flores, 2000; Wang H. et al., 2024). Slightly elevated turbidity and nutrients (NO_2^- , NH_4^+) at ST8 likely originate from agricultural surface runoff, suggest mild littoral nutrient enrichment (Simões et al., 2015; Benkaddour, 2019).

Vertical patterns further underscore the role of depth and mixing dynamics. Cheliff, as a shallow polymictic reservoir, experiences frequent wind-driven mixing, resulting in a lack of persistent thermal or chemical stratification (Soullignac et al., 2014; Banerjee et al., 2017; de Souza Santos et al., 2022). This mixing maintains relatively uniform temperature and dissolved oxygen



profiles, although surface pH and OM levels can increase due to photosynthesis and sediment resuspension (Dokulil & Teubner, 2000; Naselli-Flores, 2000; Bu et al., 2022). In contrast, Kerrada, a deep warm monomictic ecosystem, exhibits significant vertical gradients in temperature, dissolved oxygen, and nutrient concentrations. Nitrogen and phosphorus compounds accumulation in the hypolimnion is attributed to limited mixing and hypolimnetic oxygen depletion during stratification (Margalef, 1983; Tundisi & Matsumura-Tundisi, 2008; Merino-Ibarra et al., 2021).

Thermal stratification, typically occurring during warm seasons, limits vertical mixing and establishes strong thermal and chemical gradients, thereby shaping oxygen distribution, light penetration, and nutrient fluxes (Wetzel, 2000; Lampert & Sommer, 2007). Prolonged summer stratification restricts oxygen replenishment to deeper waters, leading to progressive hypoxia below 40–50 m and promoting the release of ammonium and phosphorus from sediments. Winter mixing restores oxygen and redistributes nutrients, resetting the ecosystem for the following growing season. These dynamics align with stratification patterns typical of Mediterranean deep reservoirs (Naselli-Flores, 2008; Soltani et al., 2020). The annual mean thermal and dissolved oxygen profiles confirm that Kerrada reservoir functions as a warm monomictic reservoir, with extended summer stratification and full winter mixing.

Seasonal fluctuations were most evident for temperature, DO, turbidity, and nutrient concentrations. Temperature seasonality followed patterns typical of arid and semi-arid regions (Mhamdi et al., 1994; Baitiche et al., 2025). In Cheliff reservoir, elevated summer concentrations of PO_4^{3-} , NH_4^+ , and OM reflected nutrient enrichment and organic loading, likely linked to agricultural runoff and enhanced mineralization under elevated temperatures (Dojlido & Best, 1993; Das et al., 2006; Yap, 2013; García-Chicote, 2019; Pinto et al., 2023; Baitiche et al., 2025). Nutrient inputs, especially phosphorus, have been widely linked to anthropogenic pressures (Simões et al., 2015; Guermazi et al., 2023). Precipitation-driven runoff also contributes to the transport and redistribution of nutrients within reservoirs (Navarro et al., 2009; Orduña et al., 2024). Similar summer nutrient peaks have been reported in Algerian reservoirs, where external inputs are amplified by low water levels and high evaporation rates (Ounissi & Bouchareb, 2013; Kachi et al., 2015).



Winter conditions were instead characterized by higher DO concentrations, due to enhanced mixing and atmospheric exchange. Elevated DO levels also suggest higher primary productivity, likely supported by photosynthetic activity (Alayat et al., 2013). Likewise, turbidity in Cheliff reservoir often indicated high productivity, as suspended matter can carry nutrients and organic material that support aquatic life (Enawgaw et al., 2023; Baitiche et al., 2025). By contrast, Kerrada reservoir exhibited lower turbidity and PM, consistent with its function as a storage reservoir receiving fewer external inputs. Although overall nutrient concentrations were lower in Kerrada reservoir, NO_3^- levels were significantly higher, likely reflecting catchment-specific inputs and nitrogen cycling processes. The greater nitrate retention in Kerrada reservoir may also be linked to its depth. Elevated nitrate levels in Mediterranean reservoirs have previously been associated with agricultural practices in surrounding catchments (Naselli-Flores, 2008; Ferreira et al., 2016; Almeida et al., 2020; Soltani et al., 2020).

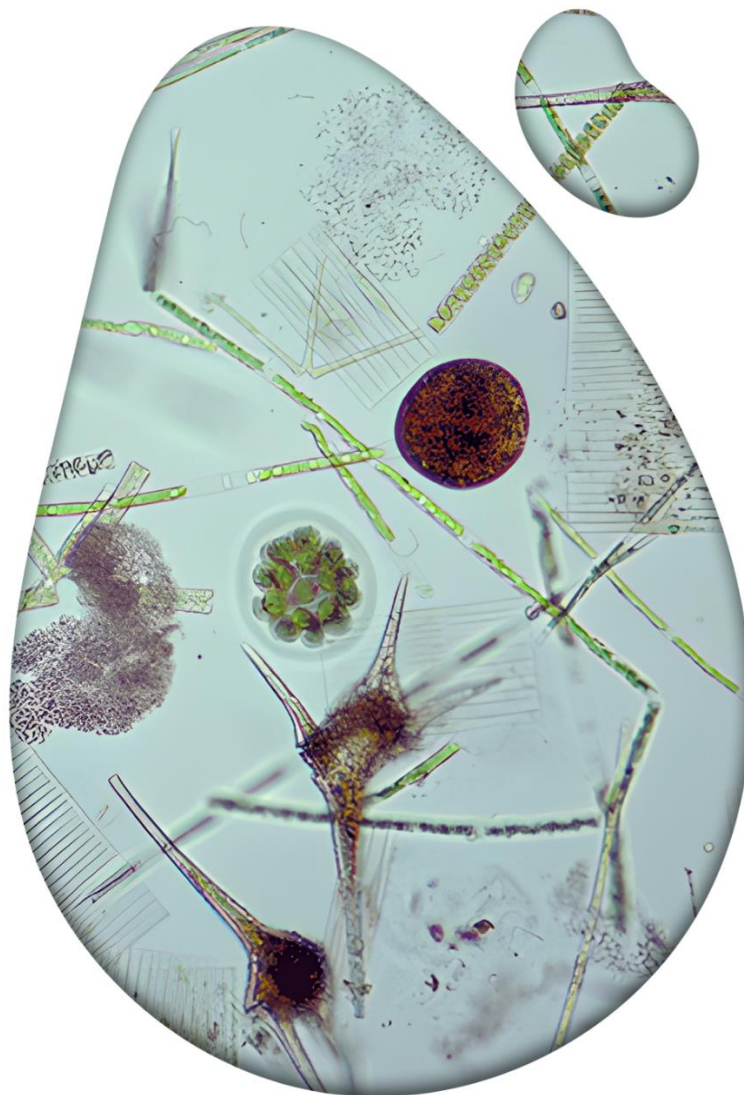
Multivariate approaches, such as Principal Component Analysis (PCA), provide powerful tools to disentangle the influence of multiple environmental drivers on reservoir water quality (Sheela et al., 2012; Parmar & Bhardwaj, 2014; Bouguerne et al., 2017; Baitiche et al., 2025). In both reservoirs, the first component (Dim-1) was strongly associated with salinity, TDS, and PM, emphasizing mineralization and sediment inputs as dominant structuring factors. Elevated temperatures can further accelerate mineralization within the water column, thereby increasing nutrient concentrations (Farrell et al., 2020). This aligns with findings from other Mediterranean and North African reservoirs, where ionic content and suspended solids dominate environmental gradients (Nasri et al., 2016). The second component (Dim-2) reflected seasonal dynamics, being associated with DO and nutrient-related parameters. In Cheliff reservoir, DO and PO_4^{3-} were dominant contributors, consistent with enrichment during warmer seasons. In Kerrada reservoir, pH and temperature explained most of the variance, reflecting its relatively stable but seasonally modulated environment.

Morphometry, particularly depth, is a key determinant of thermal and chemical dynamics in reservoirs. Recent hydrogeomorphometric analyses confirm that morphometric and catchment characteristics explain a large fraction of observed differences in water-quality patterns across reservoirs (Park, et al., 2014a). Seasonal variability, particularly in temperature, dissolved oxygen, and nutrient concentrations, further drives ecosystem functioning, with warm seasons enhancing nutrient mineralization and organic matter decomposition, while cooler periods promote mixing and higher oxygenation (Dojlido & Best, 1993; Farrell et al., 2020).



Overall, these results demonstrate that reservoir morphometry and hydrology are key determinants of physicochemical variability, modulating the response of aquatic ecosystems to both external forcing and seasonal dynamics. Shallow reservoirs such as Cheliff are more vulnerable to short-term climatic fluctuations and anthropogenic pressures, whereas deeper ecosystems like Kerrada exhibit greater resilience but remain sensitive to nutrient inputs, particularly nitrates. These findings reinforce the view that water management strategies must account for both reservoir morphometry and watershed characteristics to maintain water quality in Algerian and Mediterranean regions ([Naselli-Flores & Barone, 2005](#); [Baitiche et al., 2025](#)).

Chapter IV: Phytoplankton community structure





This chapter investigates the phytoplankton community structure and dynamics in the two study reservoirs over two full annual cycles using monthly sampling. Building on the physicochemical context established in the preceding chapter, the analysis examines how environmental variability, reservoir morphometry, and hydrological fluctuations influence phytoplankton diversity, abundance, and species composition in Cheliff and Kerrada reservoirs. The chapter describes the temporal and spatial patterns of phytoplankton diversity and community composition, highlighting contrast between the two reservoirs, and evaluates their relationships with key environmental parameters. Understanding the phytoplankton community structure is essential for evaluating how environmental variability influences overall ecosystem functioning, for informing reservoir management, and for providing critical context for subsequent assessments of trophic status and eutrophication mitigation.

IV. 1. Material and methods

IV. 1. 1. Sampling procedure

Phytoplankton samples were collected monthly over a two-year period, from November 2020 to October 2022. In each reservoir, four sampling sites were selected to capture spatial variability (Fig. 24; Table 3, Chapter III). Samples were taken along three depth layers, following the same procedure used for the physicochemical parameters. Quantitative sampling was performed by filtering 5 L of water through a 20 μm mesh conical plankton net (SDMO QUINIOU Model F131P20) and by using a Van Dorn bottle (5 L), following the method described by Miracle et al. (1992) (Fig. S4, Supplementary material). In addition, qualitative horizontal sampling was conducted by towing the plankton net in the euphotic zone. All samples were preserved in formaldehyde (4%) and transported to the laboratory for identification and enumeration.

Simultaneously, additional water samples were collected for chlorophyll-a, and nutrients analyses, as detailed in Chapter III. Phytoplankton abundance (ind. L^{-1}) was estimated by counting individuals per species in multiple aliquots and calculating mean values. Quantification followed Utermöhl sedimentation method (Utermöhl, 1958) (Fig. S5, Supplementary material), using 10 mL sedimentation chambers and an inverted microscope (ZEISS-Winkel) equipped with differential interference contrast (DIC), as described by Wetzel & Likens (2000). For very small taxa, such as Crococcales and *Chlorella* spp., counting was carried out using a Malassez counting chamber (0.02 mm^3 volume) (Fig. S6, Supplementary material) under a light microscope at 400 $\times$ magnification, with cell densities calculated from the average of multiple counted fields.



For taxonomic identification, 10 mL sub-samples were taken from each homogenized sample and examined under a stereomicroscope and light microscope (400× and 1000× magnification), as well as an inverted microscope (Nikon Eclipse Ti-U) equipped with a digital camera (Fig. S7, Supplementary material). Species identification followed standard and contemporary taxonomic references: Huber-Pestalozzi (1968), and Chapman & Chapman (1973) for **Chlorophyceae**; Förster (1982), Komárek (1983), and (Coesel, 1996) for **desmids** and **green algae**; Krammer & Lange-Bertalot (1986, 1988, 1991a, 1991b) for **diatoms**; Popovský & Pfiester (1990) for **Dinophyceae**; Komárek (1988), Komárek & Anagnostidis (2005), Komárek et al. (2003; 2014), Komárek (2010), and Komárek & Johansen (2015) for **Cyanophyceae**; Tell (1986), Wołoski & Walne (2007), Wołoski & Hindák (2004), and Wołoski (2011) for **Euglenophyceae**; John et al. (2002) and Wehr et al. (2015) for comprehensive freshwater phytoplankton identification and ecology; and Guiry & Guiry (2024, [AlgaeBase](#), continuously updated) for taxonomic validation.

IV. 1. 2. Phytoplankton counting and density calculation

a) Utermöhl method

Phytoplankton density (D , ind. L^{-1}) was calculated following Basset et al. (2009):

$$D = \frac{N_i \times R \times 1000}{V}$$

N_i: number of individuals counted (cells, filaments, aggregates, or colonies); **R**: ratio between the chamber surface area (530.66 mm²; chamber radius = 13 mm) and the total area of the counted microscope fields (surface area of one microscope field = $0.112 \times 13 \times 2 = 2.912$ mm² × number of fields counted; band width = 0.112 mm); **V**: sedimented volume (10 mL); and 1000 is the conversion factor used to express density per liter.

A total of 50 fields were counted for rare species and low-density samples, whereas 20 fields were counted for abundant species and high-density samples.

b) Cell counting with Malassez chamber

For very small phytoplankton taxa (Crococcales, *Chlorella* spp.), cell density was determined using a Malassez counting chamber. The chamber consists of a thick glass slide



with a grid of 100 equal rectangles (total volume = 1 mm³ = 1 µL; dimensions: 2.5 mm × 2 mm × 0.20 mm). Each rectangle represents 0.01 mm³ (0.01 µL) of sample.

Cell density (D , cells L⁻¹) was calculated as:

$$D = \frac{N_i}{a \times v} \times Fd$$

N_i : number of cells counted; a : number of rectangles counted; v : volume of one rectangle (0.01 mm³); and Fd : dilution factor. Results were converted to individual per liter.

IV. 1. 3. Statistical analyses

Statistical analyses were performed on physicochemical and biological (phytoplankton) datasets obtained throughout the study period from all stations in both reservoirs. Data were grouped by season and reported as average values. Environmental parameters are presented in Chapter III.

Biodiversity was assessed using:

- Shannon diversity index (H') (Shannon & Weaver, 1949):

$$H' = - \sum_{i=1}^S p_i \log_2 p_i$$

- Pielou's Evenness index (J') (Pielou, 1966):

$$J' = \frac{H'}{\ln S}$$

$p_i = \frac{n_i}{N}$, n_i is the average number of individuals of species i ; p_i is the ratio of the individual number of the species i and total individual number N , and S is the total number of species.

Ecological attributes including species richness, density (ind. L⁻¹), relative abundance, Shannon index, and evenness were computed using PAST software (Hammer et al., 2001).

To evaluate spatio-temporal differences in diversity indices, a Mann–Whitney U test (Wilcoxon rank-sum test) was applied to compare Cheliff and Kerrada reservoirs. Seasonal differences were visualized using boxplots. These non-parametric tests were chosen due to



violations of normality and homogeneity of variances. All tests were conducted at a 5% significance level.

Redundancy Analysis (RDA) was applied to identify environmental variables significantly influencing phytoplankton composition, with forward selection of predictors. Prior to analysis, biotic data were Hellinger-transformed and environmental variables were $\log(x+1)$ -transformed, except for pH, to meet homoscedasticity and normality requirements (Legendre & Gallagher, 2001).

Only the **30 most abundant phytoplankton taxa**, each contributing $> 0.5\%$ of total density, were included in the RDA. The significance of canonical axes and selected variables was tested using Monte Carlo permutation tests under the reduced model.

All statistical analyses and graphical visualizations were performed in R software (version 4.3.2) using the *vegan* and *ggplot2* packages.



IV. 2. Results

IV.2.1. Evolution of phytoplankton diversity in Cheliff and Kerrada reservoirs

A total of **140 phytoplankton species** were identified across the two reservoirs, with significantly higher species richness in Cheliff reservoir (**131 species**) compared to Kerrada (**117 species**) (Table S3, Supplementary material). **Chlorophyceae** represented the most diverse class (45 species), followed by **Diatomophyceae** (37 species), **Cyanophyceae** (29 species), and **Euglenophyceae** and **Zygnematophyceae** (11 species each). **Dinophyceae** (4 species), **Chrysophyceae** (2 species), and **Cryptophyceae** (1 species) exhibited comparatively low diversity. These taxonomic classes were distributed heterogeneously between the two reservoirs.

Species richness (S) and Shannon diversity indices (H') differed significantly between the two reservoirs (Wilcoxon test, $p < .05$), with Cheliff reservoir showing higher mean species richness (95 species) and H' (3.6). No significant difference in Pielou's Evenness Index (J') was observed (Wilcoxon test, $p > .05$), indicating a more even species distribution in both reservoirs (mean $J' = 0.8$) (Fig. 35). Seasonal variation in phytoplankton diversity and evenness was significant in both reservoirs, with consistently higher diversity during the second sampling year, particularly in summer and Autumn 2022 (Fig. 36). In Cheliff reservoir, species richness ranged from a minimum of 75 species in Spring 1 to a maximum of 122 species in Spring 2. In Kerrada reservoir, species richness varied from 24 species (Spring 2) to 89 species (Autumn 2). Shannon diversity values ranged from 2.8 (Cheliff, Spring 1) to 4.3 (Cheliff, Autumn 2), and from 2.6 (Kerrada, Spring 2) to 4.0 (Kerrada, Autumn 2). Pielou's evenness varied between 0.6 (Cheliff, Spring 1) and 0.9 (Cheliff, Summer 2), and between 0.7 (Kerrada, Winter 2) and 0.9 (Kerrada, Summer 2).

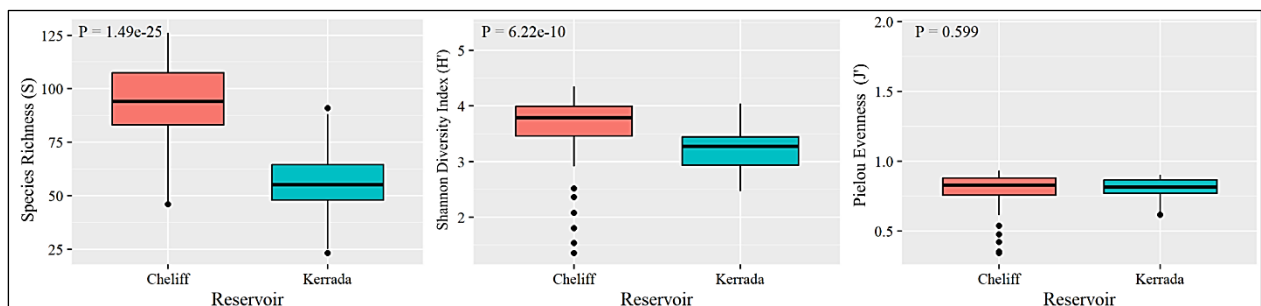


Fig. 35 Comparison of diversity indices (Shannon index H', Pielou evenness J', and Species richness S) between Cheliff and Kerrada reservoirs (Wilcoxon test, 95% Confidence Interval).

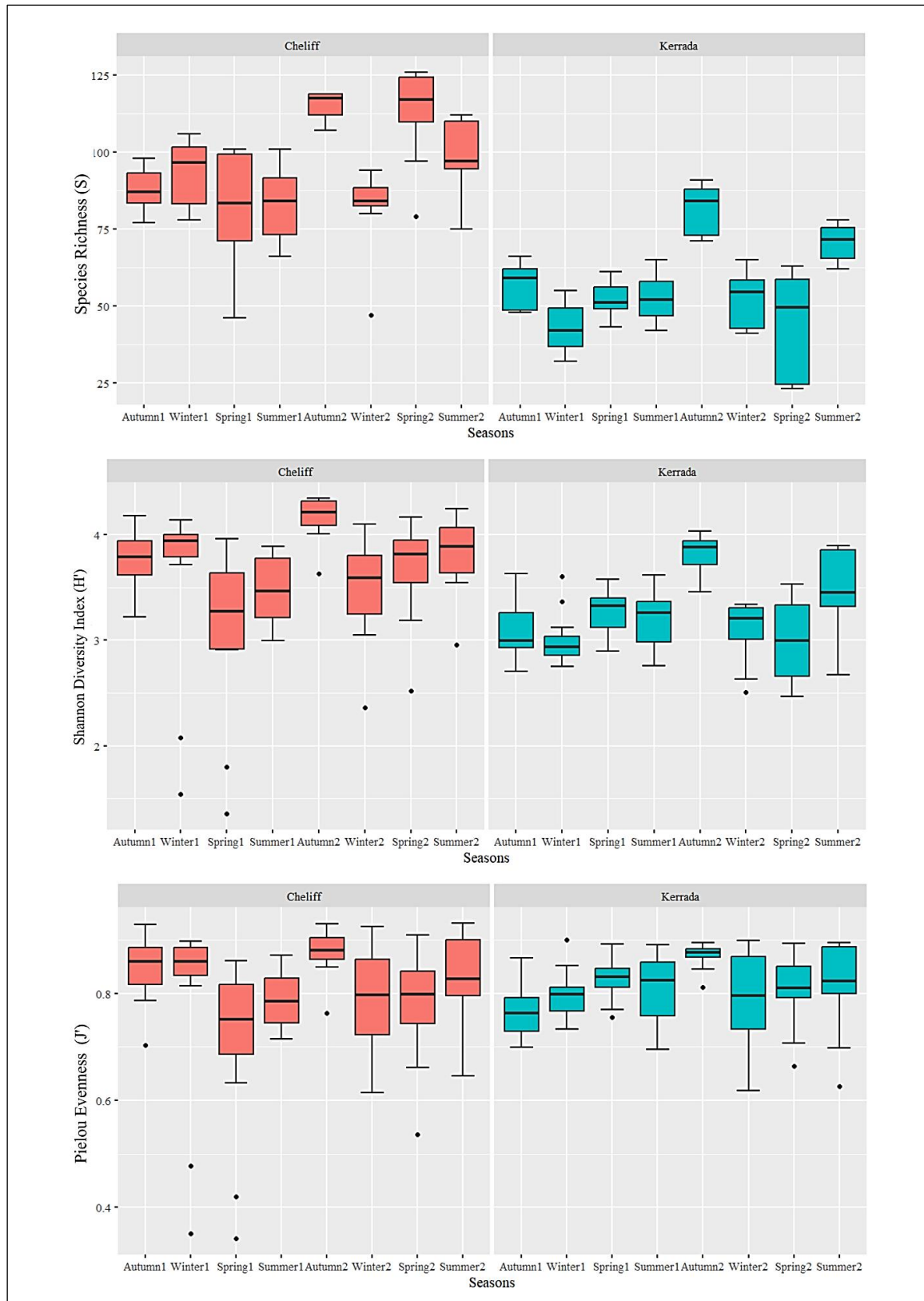
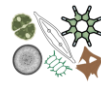


Fig. 36 Seasonal variation of diversity indices Species Richness (S), Shannon Index (H'), and Pielou's Evenness (J') in Cheliff and Kerrada reservoirs. The boxplots display the distribution of values for each season, where the central line represents the median, the box limits indicate the interquartile range (IQR), and the whiskers extend to the minimum and maximum values (excluding outliers).



IV.2.2. Phytoplankton groups diversity and dominance

A. Chlorophyceae

Chlorophyceae were dominated by *Chlorella vulgaris* in both reservoirs (Cheliff: 25.5%; Kerrada: 27.8%). *C. vulgaris* plays a crucial ecological role, contributing to water quality improvement and nutrient cycling. Its diverse ecotypes allow adaptation to a broad range of environmental conditions, enabling persistence in both wastewater and natural freshwater ecosystems (Faruque et al., 2024). Ecologically, *C. vulgaris* enhances dissolved oxygen concentrations and reduces biochemical oxygen demand (BOD), thereby improving overall water quality (Dollah et al., 2024) (Fig. 37).

Among the most dominant taxa, *Planctonema lauterbornii* was highly abundant in Kerrada reservoir (18.3%) but occurred at very low densities in Cheliff reservoir (0.3%). This planktonic species contains terminal oil droplets that increase buoyancy, allowing it to remain within well-lit epilimnetic layers of stratified water bodies, which may explain its higher abundance in Kerrada reservoir at depth (~0 – 10 m) in summer (Padisák et al., 2003; Molina-Navarro et al., 2014; Schabetsberger et al., 2017). Studies on deep reservoirs report prolonged dominance of this species in surface assemblages during stratification (Leitão et al., 2003; Mackey et al., 2013).

In Cheliff reservoir, other Chlorophyceae contributed moderately to overall abundance, including *Actinastrum hantzschii* (D = 7.1%), *Monoraphidium contortum* (D = 5.8%), *Schizochlamys gelatinosa* (D = 4.4%), *Tetradismus lagerheimii* (D = 4.3%), and *Tetrastrum heteracanthum* (D = 4.0%) (Fig. 30). These taxa are typical components of eutrophic reservoir communities and are well documented in Algerian freshwater ecosystems. Ecologically, *A. hantzschii* and *T. lagerheimii* are small, colony-forming chlorophytes indicative of meso- to eutrophic conditions, often thriving under stable, nutrient-rich conditions (Deyab et al., 2021). *M. contortum* is a crescent-shaped species tolerant of fluctuating light and nutrient conditions, frequently becoming abundant in late summer when grazing pressure declines (Ruggeri et al., 2021). *S. gelatinosa* forms gelatinous colonies that provide resistance to zooplankton grazing and are often associated with organic-rich waters. *T. heteracanthum* forms spiny coenobia that enhance buoyancy and deter grazers, supporting persistence during stratification. Collectively, these species reflect a nutrient-enriched, productive environment that favors small, colony-forming green algae capable of exploiting epilimnetic light and resisting herbivory. ST2 and ST3 were characterized by



high Chlorophyceae abundance, while the lowest values were recorded at the shallowest station (ST4) in Cheliff reservoir.

In Kerrada reservoir, phytoplankton community was characterized by filamentous and planktonic Chlorophyceae, notably *Parietochloris pseudoalveolaris* ($D = 6.6\%$) and *Oedogonium sexangulare* ($D = 5.6\%$). *P. pseudoalveolaris* (syn. *Neochloris pseudoalveolaris*) is known from culture studies to accumulate intracellular lipid globules under specific light–nutrient regimes, a trait that favors persistence and buoyancy in well-lit epilimnetic layers of stratified reservoirs. Its moderate dominance in the upper layers (euphotic zone $\sim 0\text{--}5\text{m}$) suggests efficient exploitation of stable thermal layers and available nutrients in the photic zone (Chi et al., 2021). *O. sexangulare* is a filamentous, primarily benthic alga that often becomes free-floating through detachment or resuspension. Its appearance in the plankton during autumn (September) and winter, with higher abundances, likely indicates littoral–pelagic coupling or mixing events (Rattanasasensri, 2020; Chi et al., 2021). The highest values were recorded at ST5, near the intake tower, which may indicate that opening the valves promotes vertical mixing of the water column, allowing this benthic species to appear in surface layers. These findings align with studies showing that filamentous green algae exploit stable thermal layers and resuspension processes to dominate surface assemblages in deep reservoirs (Boero et al., 2008). Chlorophyceae reached minimum relative abundance at ST5 and maximum at ST7, located in the mid-reservoir zone.

Additionally, *Neglectella solitaria* and *Oocystis lacustris* exhibited higher dominance in Kerrada compared to Cheliff reservoir. *N. solitaria* forms small colonies or paired cells that tolerate low light and are adapted to the stable epilimnion of deep reservoirs, maintaining positive buoyancy and efficient nutrient uptake (Wang H. et al., 2024). *O. lacustris* is a cosmopolitan coccoid alga with thick-walled, mucilaginous coenobia that confer grazing resistance, allowing it to dominate during periods of low turbulence (Oliva et al., 2001). The higher abundance of these taxa in Kerrada likely reflects the reservoir's deeper, thermally stratified conditions, favoring small, colony-forming Chlorophyceae with high nutrient uptake efficiency and grazer defense strategies (Moreno-Ostos et al., 2006; Wang H. et al., 2024).

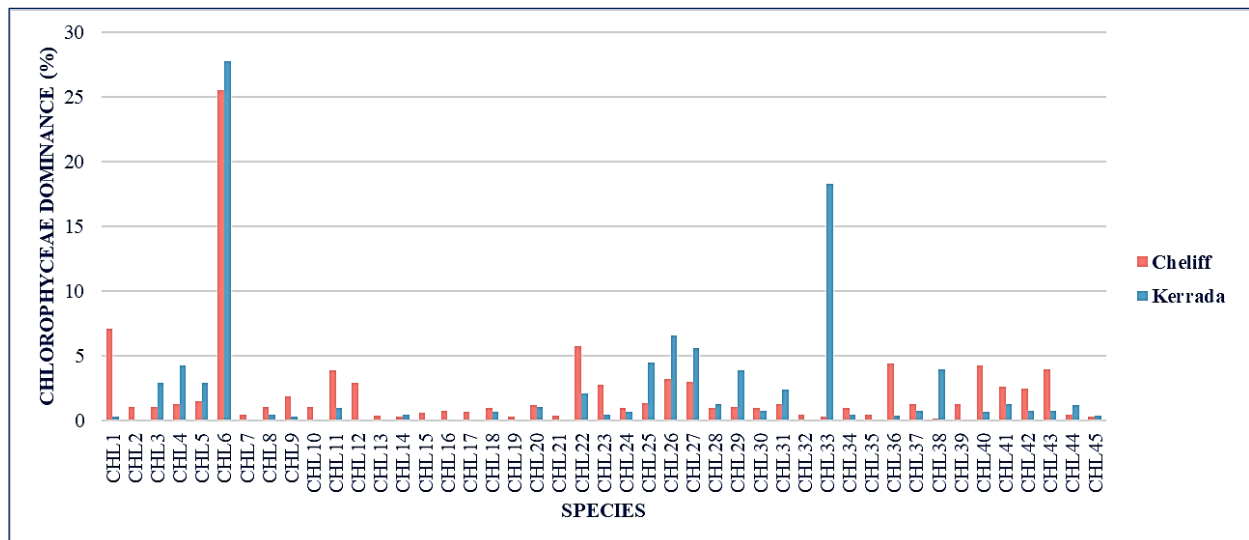


Fig. 37 Chlorophyceae species dominance in Cheliff and Kerrada reservoirs (species codes are provided in Supplementary material, Table S3).

B. Bacillariophyceae

Bacillariophyceae ranked second after Chlorophyceae in species diversity, with 37 taxa identified across both reservoirs (Cheliff: 34 species; Kerrada: 29 species). Cheliff, the shallow reservoir, showed higher diatom abundance, consistent with the dominance of benthic and tychoplanktonic forms in nutrient-rich environments (Berdalet et al., 1996; Carlsson & Granéli, 1999; Nehar et al., 2015).

Among diatoms, centric species were most dominant. *Stephanocyclus meneghinianus* was the leading taxon in both reservoirs, with higher dominance in Cheliff reservoir ($D = 23.7\%$) compared to Kerrada reservoir ($D = 15\%$). Other common centric diatoms included *Cyclotella atomus* (Cheliff: $D = 8.2\%$; Kerrada: $D = 6.9\%$), *Stephanodiscus neoastraea* (Cheliff: $D = 7.6\%$; Kerrada: $D = 7.2\%$), *Pantocsekiella ocellata* (Cheliff: $D = 6.9\%$; Kerrada: $D = 7.0\%$), and *Discostella stelligera* (Cheliff: $D = 4.5\%$; Kerrada: $D = 5.0\%$) (Fig. 38). *S. meneghinianus* is highly tolerant of pollution and low oxygen (Germain, 1981; Dam et al., 1994; Ludes & Coste, 1996; El Haouati et al., 2015) and is widespread in Algerian reservoirs. *C. atomus* is a planktonic diatom frequently associated with hypereutrophic conditions (Germain, 1981), whereas *S. neoastraea* is an alkalibiontic species typically found in eutrophic, well-oxygenated waters (El Haouati et al., 2015). *P. ocellata* is an alkaliphilous taxon reported in Algerian lakes with variable salinity and oxygen ranges (Schlegel & Scheffler, 1999; Fawzi et al., 2002; Chaïb & Tison-Rosebery, 2012; Adaouri et al., 2024). *D. stelligera*, a small centric diatom, was more abundant in Cheliff reservoir,



reflecting its preference for shallow, polymictic ecosystems with frequent resuspension. Its lower abundance in Kerrada reservoir likely results from stratification limiting nutrient recycling.

Pennate diatoms were also well represented. The genus *Nitzschia* was the most diverse (six species), with *Nitzschia palea* being most dominant (Kerrada: D = 8.5%; Cheliff: D = 7.9%). *N. palea* is known for tolerance to organic pollution and elevated H₂S levels (Baudrimont, 1974; Lange-Bertalot, 1979; Foged, 1987; Hiramza et al., 2021) and is common in Algerian waters (Chaïb et al., 2011; El Haouati et al., 2015; Nehar et al., 2015; Chabaca et al., 2020; Negadi et al., 2021). *Nitzschia acicularis* showed higher dominance in Cheliff reservoir (D = 7.3%) than in Kerrada reservoir (D = 4.4%), likely due to enhanced mixing and nutrient resuspension in the shallow reservoir, favoring its persistence in suspension (Tang & Cao, 2019). Its lower abundance in Kerrada reservoir may reflect nutrient depletion in stratified epilimnetic waters. Tychoplanktonic *Actinella punctata* commonly found in oligotrophic to mesotrophic freshwater ecosystems, exhibited notable dominance in Kerrada reservoir (D = 8.2%), suggesting favorable conditions for littoral–pelagic coupling.

Several benthic and epiphytic species (*Achnantheidium catenatum*, *Cocconeis placentula*, *Craticula ambigua*, *Diploneis ovalis*, *Gomphonema parvulum*, *Nitzschia dissipata*, *Surirella ovalis*, *Surirella librile*) were exclusive to Cheliff reservoir, consistent with its shallow, polymictic nature and strong river inflow that facilitates continuous resuspension of benthic forms (Germain, 1981; Fawzi et al., 2002; Chaïb et al., 2011; Kerimoglu & Rinke, 2013; Nehar et al., 2015). These results are reinforced by the higher abundances and greater diatom diversity at ST4, upstream of the reservoir at the river inflow, followed by ST3. Both sites are characterized by the presence of riparian vegetation, which promotes diatom growth. Conversely, *Asterionella formosa*, *Fragilaria capucina*, and *Hyalodiscus* sp. were exclusive to Kerrada reservoir, reflecting their adaptation to stable, stratified, and oligotrophic-to-mesotrophic conditions typical of deep storage reservoirs.

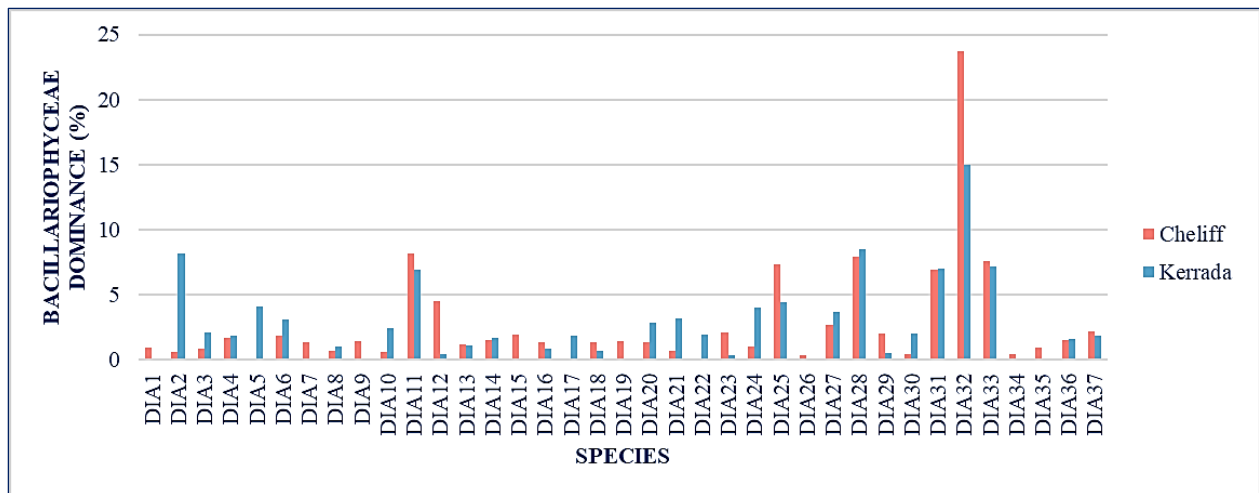


Fig. 38 Bacillariophyceae species dominance in Cheliff and Kerrada reservoirs (species codes are provided in Supplementary material, Table S3).

C. Cyanophyceae

Several studies have reported on Cyanophyceae proliferation in North African freshwater ecosystems, highlighting its ecological and toxicological relevance (Oudra et al., 2001; Sabour et al., 2002; Bensouilah et al., 2003; Nasri et al., 2004; El Herry et al., 2008; Bidi-Akli et al., 2017; Benayache et al., 2022; Guellati et al., 2024). These studies demonstrated that bloom-forming cyanobacteria, particularly *Microcystis*, release microcystins into natural waters.

In the present study, Cyanophyceae were represented by 29 taxa, with slightly higher richness in Kerrada reservoir (29 taxa) than in Cheliff reservoir (26 taxa). *Microcystis aeruginosa* was the most dominant species in both reservoirs (Cheliff: D = 17.1%; Kerrada: D = 13.1%). This species is a globally widespread bloom-forming cyanobacterium typical of eutrophic and hypereutrophic ecosystems, thriving under warm, stratified, nutrient-enriched conditions (Reynolds et al. 2000; Sharma & Tiwari, 2018; Ochieng et al., 2022; Sennour et al., 2025). Its mucilaginous colonies confer buoyancy, enabling efficient light acquisition and competitive exclusion of other phytoplankton. *M. aeruginosa* is also a major microcystin producer, and its proliferation is frequently linked to reduced biodiversity, oxygen depletion following bloom collapse, and water quality degradation (Paerl & Otten, 2013; Radji et al., 2013).

Pseudanabaena catenata was the second most abundant taxon (Cheliff: D = 11.6%; Kerrada: D = 8.6%), followed by *Aphanocapsa elachista*, which dominated Kerrada reservoir (D = 14.6%) and was also abundant in Cheliff reservoir (D = 11%) (Fig. 39). *A. elachista* forms small mucilaginous colonies and is capable of nitrogen fixation through non-heterocystous mechanisms, typically occurring in microoxic zones or at night to protect the nitrogenase enzyme from oxygen



(Bergman et al., 1997; Kulasooriya & Magana-Arachchi, 2016). This confers a competitive advantage under nitrogen-limited conditions, particularly during summer stratification in deep reservoirs (Horváth et al., 2013; Paerl, 2017). Its high abundance recorded in deeper layers of the metalimnion (~10–50 m) further reflects its ability to thrive under low-oxygen conditions. The substantial contribution *A. elachista* to total cyanobacterial abundance suggests a central role in primary production and a potential involvement in bloom-related oxygen dynamics (Ferber et al., 2004; Bhardwaj et al., 2024).

Cheliff reservoir exhibited also high dominance of filamentous cyanobacteria, including *Pseudanabaena mucicola* (D = 8.4%), *Planktolyngbya limnetica* (D = 8.2%), *Raphidiopsis raciborskii* (D = 4.3%), *Limnothrix redekei* (D = 4.3%), *Planktothrix agardhii*, and *Raphidiopsis curvata* (both D = 4%). These taxa are characteristic of eutrophic to hypertrophic conditions, and their filamentous morphology confers grazing resistance and tolerance to low-light environments. (Pagni et al., 2020; Meriggi et al., 2024). *R. raciborskii* is particularly notable as an invasive, bloom-forming species with broad environmental tolerance (Padisák, 1997; Kokociński et al., 2010; Pagni et al., 2020), whereas *P. agardhii* is often associated with turbid, hypertrophic reservoirs (Kokociński et al., 2010). Its proliferation has been reported as the cause of massive fish mortality in the two basins of Zeralda hunting reserve (Bidi-Akli et al., 2017). The co-occurrence of these species suggests that Cheliff reservoir supports a cyanobacterial community adapted to stable, nutrient-rich, shallow systems prone to bloom formation (Stević et al., 2025).

In Kerrada reservoir, *Limnothrix redekei* (D = 9.3%), *Chlorogloea microcystoides* (D = 4.8%), and *Aphanothece variabilis* (D = 4.7%) were prominent. Dominance of *L. redekei* indicates a stable, productive ecosystem where vertical mixing is limited, allowing this species to outcompete other phytoplankton under low-light conditions (Reynolds & Bellinger, 1992; Rojo & Cobelas, 1994; Moustaka-Gouni et al., 2007). The presence of mucilaginous colonial taxa such as *C. microcystoides* and *A. variabilis* further indicates environmental conditions that favor grazing resistance and allow for prolonged persistence within the water column (Ray et al., 2021).

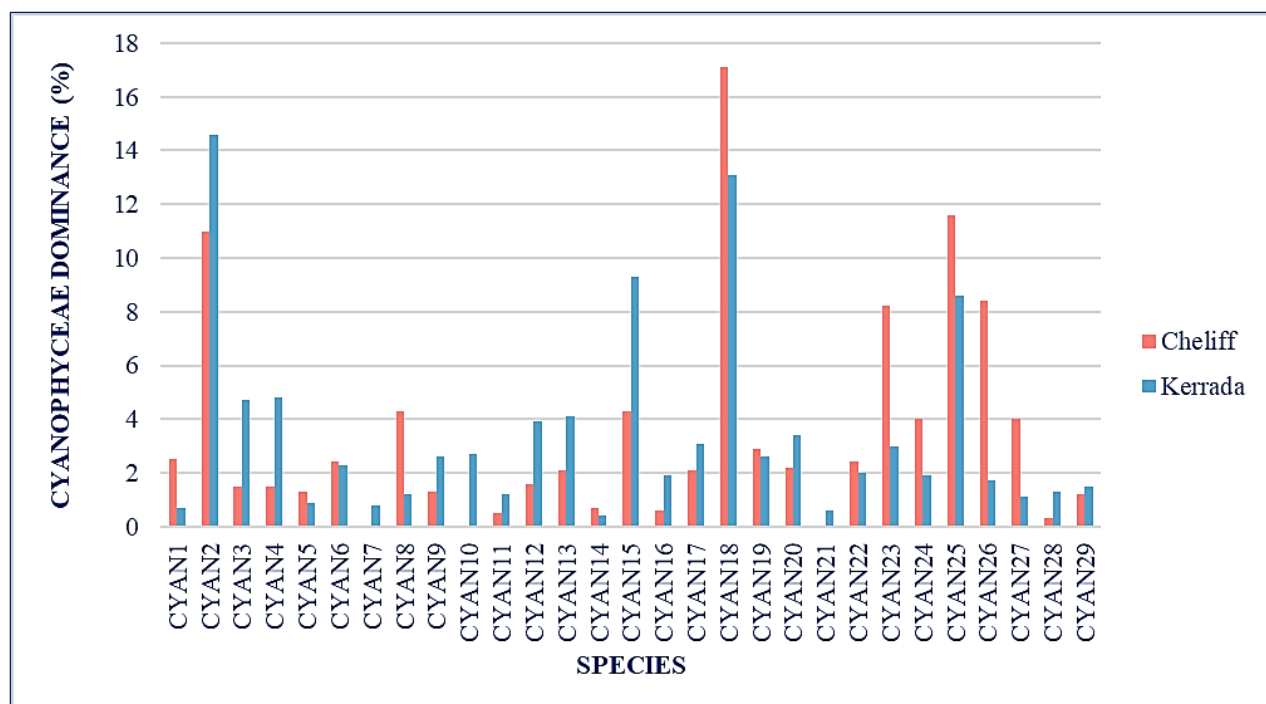
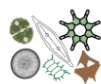


Fig. 39 Cyanophyceae species dominance in Cheliff and Kerrada reservoirs (species codes are provided in Supplementary material, Table S3).

D. Euglenophyceae

Euglenophyceae were represented by 11 species, with higher diversity recorded in Cheliff reservoir (11 species) than in Kerrada reservoir (9 species). In Cheliff reservoir, *Euglenaria caudata* was the overwhelmingly dominant species ($D = 51.5\%$), followed by *Euglena oblonga* ($D = 17.3\%$) and *Lepocinclis acus* ($D = 12.3\%$) (Fig. 40). Notably, members of the genus *Phacus* were present exclusively in Cheliff reservoir. As a shallow diversion reservoir with higher nutrient inputs and stronger sediment–water interactions, Cheliff provides favorable conditions for *Phacus* proliferation. *Phacus acuminatus* and *Phacus curvicauda* proliferated during autumn (September 2022), exhibiting high dominance across all stations, particularly at ST3 where the water level was very low (<1 m), while ST4 was completely dry. Under these conditions of overall low water level, *P. acuminatus* was especially abundant. Cheliff reservoir also recorded *Trachelomonas* species, but with much lower dominance compared to Kerrada: *Trachelomonas volvocina* ($D = 4.2\%$), *Trachelomonas caudata* ($D = 2.7\%$), and *Trachelomonas hispida* ($D = 1.6\%$). *Trachelomonas armata* occurred only in Cheliff reservoir, with very low dominance ($D = 0.5\%$). The strong dominance of *Euglena* species reflects the shallow, nutrient-enriched, and organically loaded conditions of Cheliff reservoir, as Euglenophyceae are known to proliferate in environments with elevated organic matter and fluctuating oxygen concentrations (Wołowski & Hindák, 2005; Ghosh & Keshri, 2010; Wołowski, 2011; Kim et al., 2016; Arab et al., 2019; Afaf et al., 2022).



In Kerrada reservoir, euglenophycean community was dominated by the genus *Trachelomonas*, which was also the most diverse group, with four species recorded in both reservoirs. *T. volvocina* was the most abundant taxon ($D = 63.3\%$), followed by *T. caudata* ($D = 9.5\%$) and *T. hispida* ($D=4.7\%$). The prevalence of loricate *Trachelomonas* species in Kerrada reservoir likely reflects the influence of its deeper, thermally stratified water column and relatively lower organic load, conditions that favor loricate euglenoids adapted to mesotrophic ecosystems and capable of mixotrophic nutrition (Wołowski & Grabowska 2007; Coesel & Krienitz, 2008; Wang H. et al., 2024). *Phacus acuminatus* was recorded only sporadically in Kerrada reservoir: once in summer (June 2021) with very low densities (364 ind. L^{-1}) across all stations, and again in July 2021 at ST4 (1456 ind. L^{-1}). However, the species was absent during the remainder of the two-year sampling period. These occasional occurrences were likely driven by temporary increases in organic matter and localized low-water conditions.

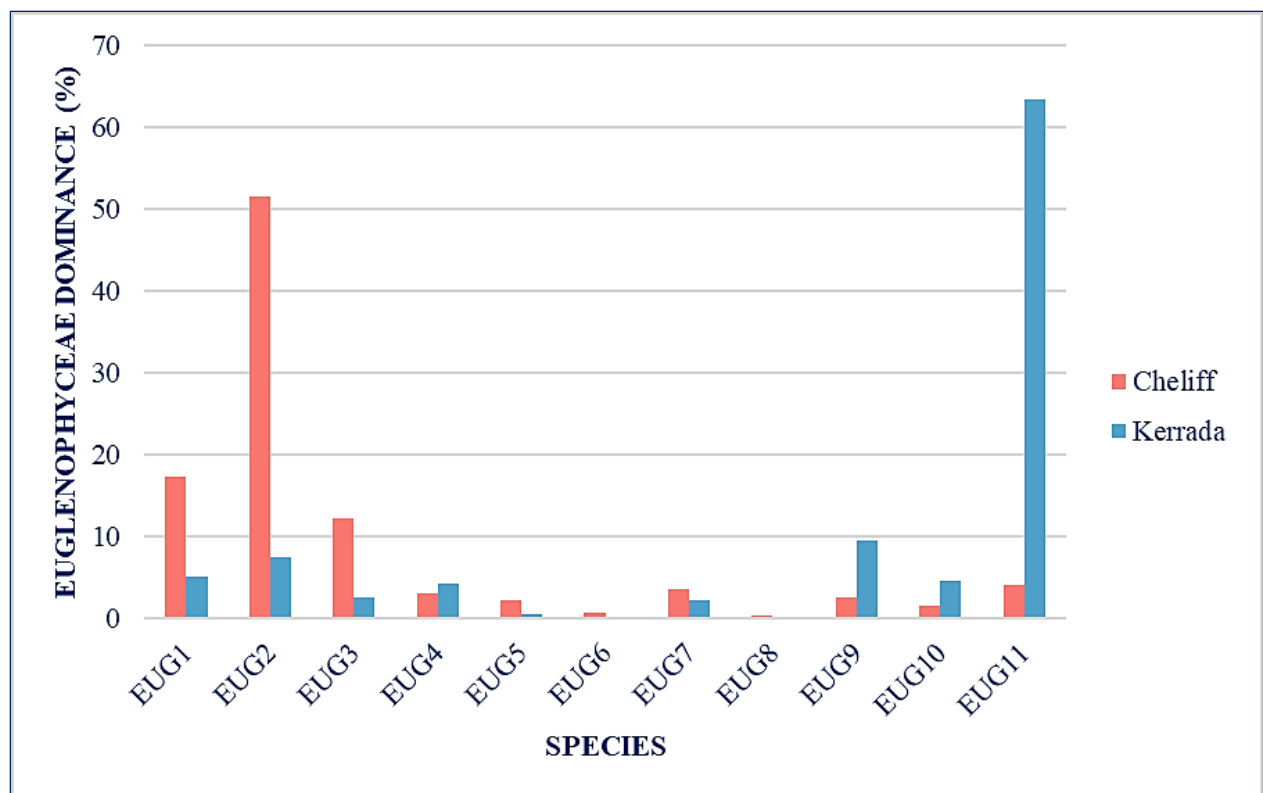


Fig. 40 Euglenophyceae species dominance in Cheliff and Kerrada reservoirs (species codes are provided in Supplementary material, Table S3).



E. Zygnematophyceae

Zygnematophyceae were represented by 11 species, with greater diversity and overall abundance recorded in Kerrada reservoir (10 species) compared to Cheliff reservoir (9 species). *Closterium aciculare* was the dominant taxon in both reservoirs, showing slightly higher dominance in Cheliff reservoir ($D = 36.2\%$) than in Kerrada reservoir ($D = 27.3\%$) (Fig. 41). The prevalence of *C. aciculare* reflects its affinity for nutrient-enriched freshwater ecosystems and tolerance to broad light and oxygen gradients (Brook, 1981; Meesters & Coesel, 2007; Afaf et al., 2022). Its higher relative abundance in Cheliff reservoir likely results from the reservoir's shallow depth, which enhances sediment–water interactions and nutrient availability, favoring desmid proliferation. In contrast, the greater species richness in Kerrada reservoir may be facilitated by its stable water column and reduced turbidity, which provide optimal light conditions for a wider spectrum of desmid taxa sensitive to mixing and light limitation (Happey-Wood, 1988; Coesel & Kooijman-Van-blokland, 1991; Reynolds, 2006).

In Cheliff reservoir, *Closterium acutum* ($D = 20.6\%$), *C. acutum* var. *variabile* ($D = 19.4\%$), and *Cosmarium granatum* ($D = 7.1\%$) were among the dominant Zygnematophyceae but were absent from Kerrada reservoir. These desmids are commonly associated with well-lit, slightly alkaline, and nutrient-enriched shallow waters (Brook, 1981; McKnight, 2000). Their dominance in Cheliff reservoir underscores the influence of its polymictic regime, higher nutrient flux from sediments, and shallow morphometry in promoting desmid growth.

In Kerrada reservoir, *Cosmarium depressum* ($D = 16.3\%$) and *C. punctulatum* ($D = 13.1\%$) were the most abundant Zygnematophyceae, together with *C. acutum* var. *variabile* ($D = 10.8\%$). Dominance of these species reflects Kerrada's more stable hydrological and physicochemical conditions, with lower turbidity and well-developed stratification during the warm season. *Cosmarium* species are known to prefer clear, well-illuminated, and moderately nutrient-enriched environments, avoiding turbulent waters (Brook, 1981; Meesters & Coesel 2007; Sendra et al., 2019). The occurrence of *C. acutum* var. *variabile* in both reservoirs, though with lower dominance in Kerrada reservoir, suggests this taxon exhibits ecological plasticity, tolerating both shallow and deep ecosystems but responding differently to light and nutrient regimes. Overall, the Zygnematophyceae composition in Kerrada reservoir indicates a community adapted to stable, well-lit conditions that support high desmid diversity and biomass.

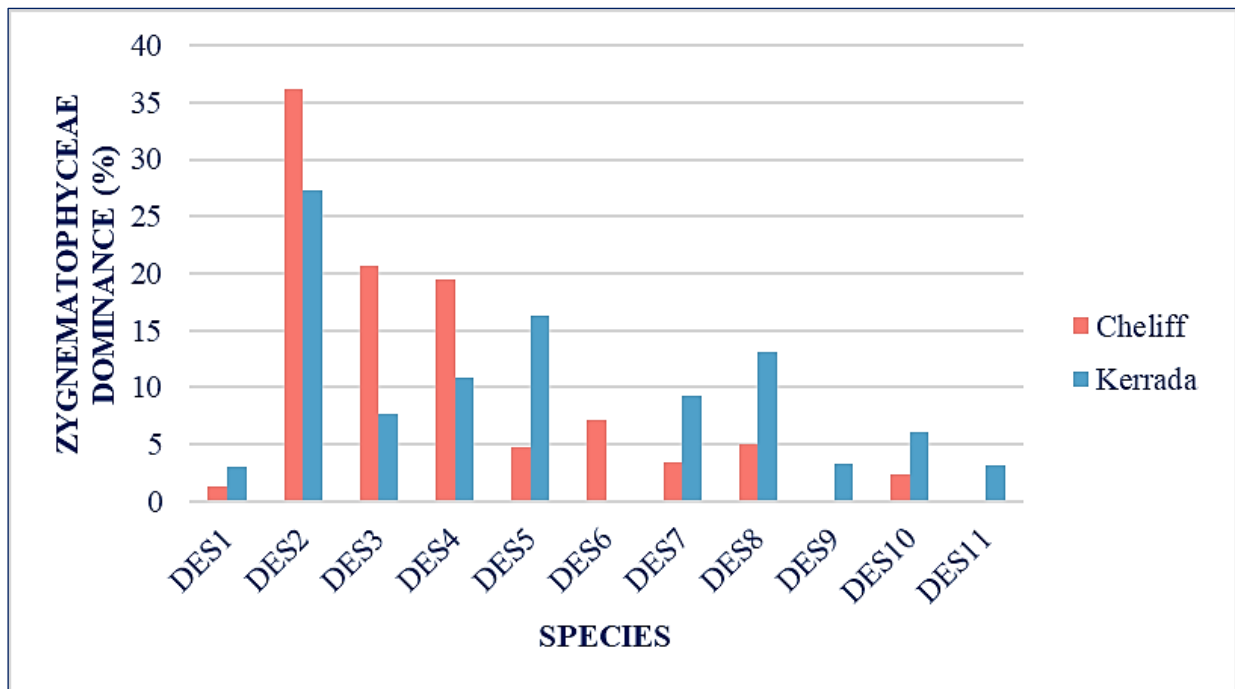


Fig. 41 Zygmatophyceae species dominance in Cheliff and Kerrada reservoirs (species codes are provided in Supplementary material, Table S3).

F. Dinophyceae

Dinophyceae (Pyrrophyceae) were relatively scarce in both reservoirs, with only four species recorded in Kerrada reservoir and three in Cheliff reservoir. *Gymnodinium* sp. was absent from Cheliff reservoir, likely due to its sensitivity to the shallow, turbid, and unstable conditions that characterize this diversion reservoir. In Cheliff reservoir, *Peridinium cinctum* was the dominant dinoflagellate ($D = 61.6\%$), reflecting its preference for nutrient-rich, well-mixed waters where it can successfully compete with other phytoplankton. In contrast, Kerrada reservoir was dominated by *Ceratium hirundinella* ($D = 50.5\%$), a large dinoflagellate well adapted to deep, thermally stratified ecosystems (Sendra et al., 2019; Adaouri et al., 2024) (Fig. 42). Its dominance across a broad depth range (0–40 m), with higher abundances recorded at 20 m particularly during summer and autumn, may be linked to its capacity for diel vertical migration. This ability allows *C. hirundinella* to optimize light harvesting and nutrient acquisition throughout the water column, providing a competitive advantage in stable, deep reservoirs (Whittington et al., 2000; Grigorszky et al., 2019).

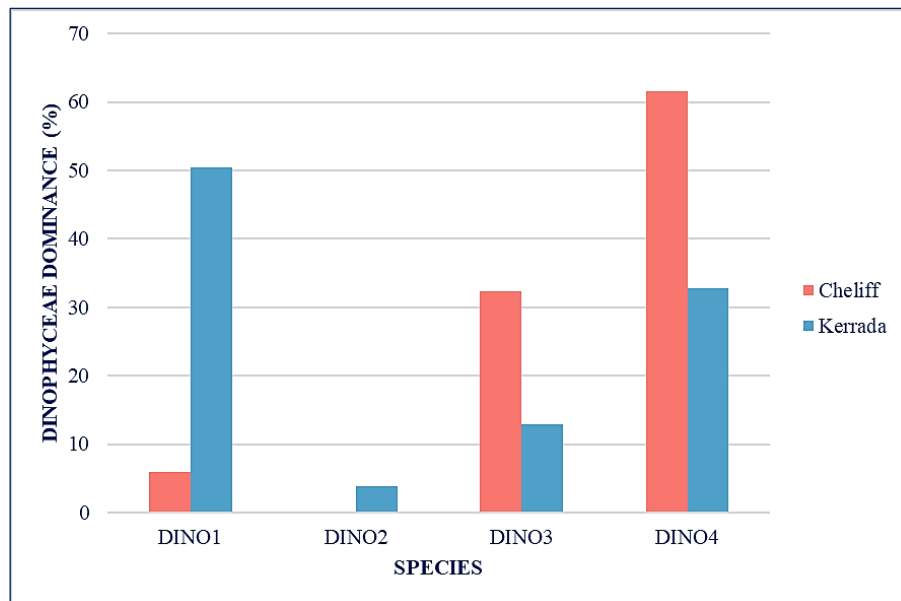


Fig. 42 Dinophyceae species dominance in Cheliff and Kerrada reservoirs (species codes are provided in Supplementary material, Table S3).

G. Chrysophyceae

Chrysophyceae (golden algae) were represented by two common species in the studied reservoirs, exhibiting clear differences in dominance patterns between sites. In Kerrada reservoir, *Dinobryon pediforme* was the dominant chrysophyte ($D = 69.6\%$), consistent with its preference for deep, well-oxygenated, and relatively transparent waters (Wang H. et al., 2024) (Table S3, Supplementary material). As a mixotroph, this species can supplement photosynthesis with bacterivory under nutrient-limited conditions, a strategy well suited to Kerrada's stable, stratified environment during the warm seasons. In contrast, *Mallomonas caudata* dominated in Cheliff reservoir ($D = 69.2\%$), a shallow and nutrient-enriched reservoir. *Mallomonas* species are often abundant in eutrophic waters and can tolerate higher turbidity and nutrient fluctuations, conditions characteristic of Cheliff reservoir (Nicholls & Wujek, 2015).

H. Cryptophyceae

Cryptomonas pyrenoidifera was the only recorded species of Cryptophyceae in both reservoirs, showing markedly higher dominance in Kerrada reservoir and being nearly absent in Cheliff reservoir (Table S3, Supplementary material). Cryptophytes are motile, mixotrophic flagellates that thrive in deep, well-stratified, and moderately nutrient-rich waters, where they can vertically migrate to optimize light and nutrient acquisition (Klaveness, 1988; Becker et al., 2010). The higher dominance of *C. pyrenoidifera* particularly during the warm seasons in Kerrada reflects



the reservoir's stable thermal stratification and clearer conditions, which favor its growth and persistence (Camacho et al., 2001; Metfies et al., 2010). In contrast, the shallow, turbid, and frequently mixed environment of Cheliff reservoir likely constrains its competitive advantage, resulting in its near absence. This pattern underscores the sensitivity of cryptophytes to light availability and mixing depth, making them valuable indicators of stratification and water column stability in reservoir ecosystems.

IV.2.3. Comparative patterns of phytoplankton relative abundance

According to the heatmap of the 30 most dominant phytoplankton species (Fig. 43), Kerrada reservoir exhibited a generally higher relative abundance of most species, with many taxa appearing more evenly distributed across the community (warmer colors – yellow/orange). In contrast, Cheliff reservoir was characterized by the strong dominance of one or two species, most notably *Euglenaria caudata*, indicating a less even community structure. *E. caudata* showed very high dominance in Cheliff reservoir ($D = 10.2\%$) but was nearly absent in Kerrada reservoir ($D = 0.2\%$). This pronounced dominance in Cheliff reservoir may indicate organic enrichment or eutrophic conditions, as many euglenoids are known to proliferate in nutrient-rich, organically loaded environments (Tian et al., 2017; Arab et al., 2019; Ouattara et al., 2023; Varga et al., 2024).

Several species, such as *Golenkinia radiata*, *Microcystis aeruginosa*, and *Aphanocapsa elachista*, displayed similar relative abundances in both reservoirs, suggesting that they tolerate a range of environmental conditions.

The more evenly distributed community structure observed in Kerrada reservoir suggests a stable, thermally stratified ecosystem where interspecific competition is more balanced, preventing the extreme dominance of a single taxon. These differences likely reflect the morphometric and physico-chemical contrasts between the reservoirs: Cheliff reservoir, being shallow and subject to strong mixing, higher turbidity, and greater sediment–water interactions, favors tolerant, opportunistic species such as *E. caudata* (Austin et al., 2017). Whereas Kerrada reservoir, with its deeper, stratified, and clearer waters, supports a more diverse and evenly distributed phytoplankton community (Becker et al., 2010; Bai et al., 2023; Varga et al., 2024).

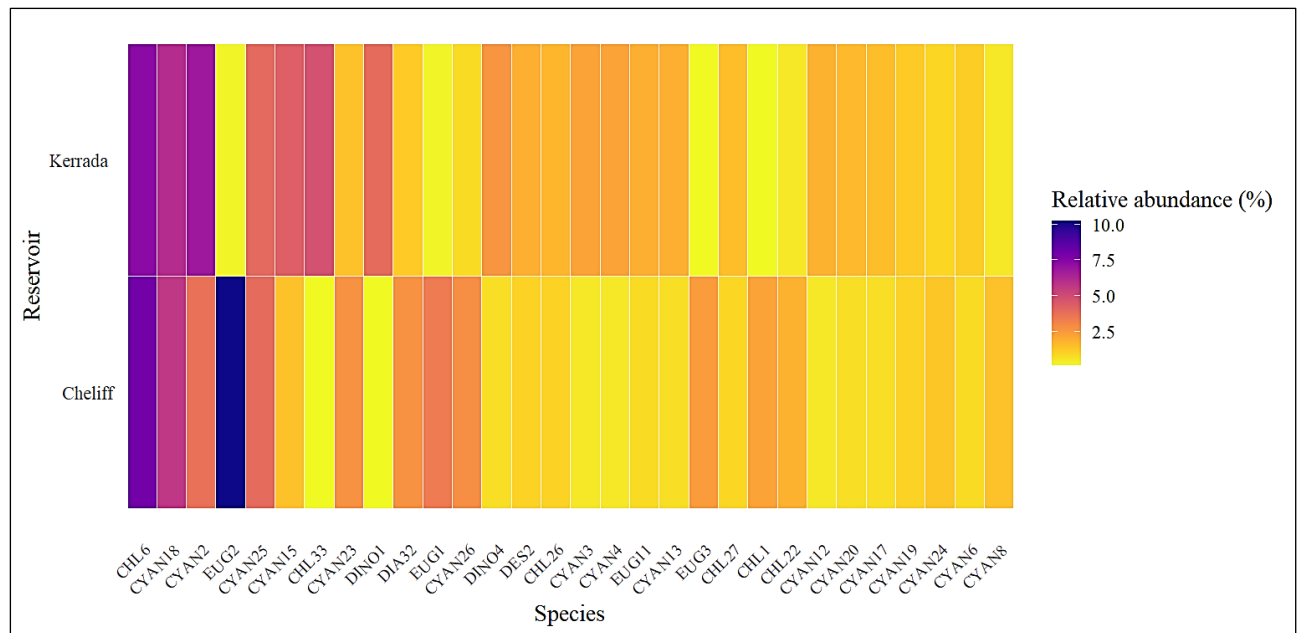


Fig. 43 Heatmap of the 30 most dominant phytoplankton species over two years of sampling in Cheliff and Kerrada reservoirs.

IV. 2.4. Seasonal changes in phytoplankton group densities

Phytoplankton density varied markedly between Cheliff and Kerrada reservoirs (Fig. 44). In Cheliff reservoir, mean total density reached $\sim 8 \times 10^5$ ind. L^{-1} , with a peak of $\sim 14 \times 10^5$ ind. L^{-1} in Spring 1. Kerrada reservoir exhibited a significantly lower mean density ($\sim 16 \times 10^4$ ind. L^{-1}), with a peak of $\sim 3 \times 10^5$ ind. L^{-1} in Autumn 1. The lowest densities occurred in Cheliff reservoir during Autumn 1 ($\sim 4 \times 10^5$ ind. L^{-1}) and in Kerrada reservoir during Spring 2 ($\sim 1 \times 10^5$ ind. L^{-1}).

High densities in Cheliff reservoir were mainly driven by Euglenophyceae. The exceptionally high relative abundance of this group in Spring 1 resulted from peak densities of *Euglenaria caudata*, which reached 5.5×10^6 ind. L^{-1} at ST4. This was followed by *Euglena oblonga* (1.6×10^6 ind. L^{-1}) and *Lepocinclis acus* (1.2×10^6 ind. L^{-1}), also concentrated at ST4. Cyanophyceae represented 33.4% of total phytoplankton abundance, with peaks during warm seasons (19% in Spring 2, $\sim 4 \times 10^5$ ind. L^{-1}). The highest relative abundance in Spring 2 was driven by a bloom of *Microcystis aeruginosa*, which reached 9.1×10^5 ind. L^{-1} at ST4. Other noteworthy cyanobacterial peaks included: *Pseudanabaena catenata* (4.5×10^5 ind. L^{-1}) at ST1 in Winter 2; *Aphanocapsa elachista* (3.6×10^5 ind. L^{-1}) at ST4 in Winter 2; *Planktolyngbya limnetica* (2.5×10^5 ind. L^{-1}) at ST3 in Autumn 2, coinciding with reservoir-level decline and drying of ST4; *Planktothrix agardhii* (2×10^5 ind. L^{-1}) at ST4 in Summer 2; *Pseudanabaena mucicola* and *Raphidiopsis curvata* ($\sim 2 \times 10^5$ ind. L^{-1}) at ST2 in Spring 1.



Chlorophyceae were the second most abundant group, peaking in Spring 2. The dominant taxon was *Chlorella vulgaris* (4.4×10^6 ind. L^{-1}) at ST1 in Winter 1. Other notable records included: *Parietochloris pseudoalveolaris* (3×10^6 ind. L^{-1}) at ST1 in Summer 1; *Actinastrum hantzschii* (2.9×10^6 ind. L^{-1}) at ST1 in Spring 1; *Schizochlamys gelatinosa* (2.2×10^6 ind. L^{-1}) at ST2 in Winter.

Diatomophyceae were abundant during winter and spring. Key species included: *Stephanocyclus meneghinianus* (2.5×10^5 ind. L^{-1}) at ST2; *Stephanodiscus neoastraea* (1×10^5 ind. L^{-1}) at ST2; *Nitzschia acicularis* (1×10^5 ind. L^{-1}) at ST2; *Nitzschia palea* (8×10^4 ind. L^{-1}) at ST2.

In Kerrada reservoir, Cyanophyceae and Chlorophyceae were the main contributors. Cyanophyceae dominated total density (46.6%), with maximum values in Autumn 1 (27%, $\sim 1 \times 10^5$ cells L^{-1}). The high relative abundance was primarily due to the proliferation of *Limnothrix redekei*, which reached 1×10^5 ind. L^{-1} at ST2. Chlorophyceae peaked in Winter 2, driven mainly by: *Chlorella vulgaris* (1.5×10^4 ind. L^{-1}) at ST5 and ST6; and *Stigeoclonium tenue* (3.6×10^4 ind. L^{-1}) at ST5. Additional important contributions included: *Parietochloris pseudoalveolaris* (1×10^5 ind. L^{-1}) at ST1 (Autumn 1); *Planctonema lauterbornii* (1×10^5 ind. L^{-1}) at ST4 (Summer 1); *Oedogonium sexangulare* (5.5×10^4 ind. L^{-1}) at ST1 (Autumn 1).

Diatomophyceae in Kerrada peaked in autumn, driven by *Stephanodiscus neoastraea* (1.3×10^4 ind. L^{-1}) at ST8, and *Stephanocyclus meneghinianus* (1×10^4 ind. L^{-1}) at ST7.

Dinophyceae were markedly more abundant in Kerrada reservoir, especially in summer and autumn ($\sim 3 \times 10^4$ ind. L^{-1}). Key contributor species included *Ceratium hirundinella* (4×10^4 ind. L^{-1}) at ST8, and *Peridinium cinctum* (2×10^4 ind. L^{-1}) at ST6.

In Cheliff reservoir, ST4 exhibited the highest relative abundances of Euglenophyceae, Cyanophyceae, and Chlorophyceae, making it the most productive and diverse station, followed by ST3. Diatomophyceae were more abundant at ST2, although taxonomic diversity remained highest at ST3 and ST4 due to dense riparian vegetation supporting benthic and tychoplanktonic diatoms. In Kerrada reservoir, ST8 the shallowest station displayed the highest relative abundances and diversity, especially for Cyanophyceae and Diatomophyceae. ST7, located near the mid-reservoir island, exhibited the second-highest diversity, reflecting its intermediate depth and stable environmental conditions.

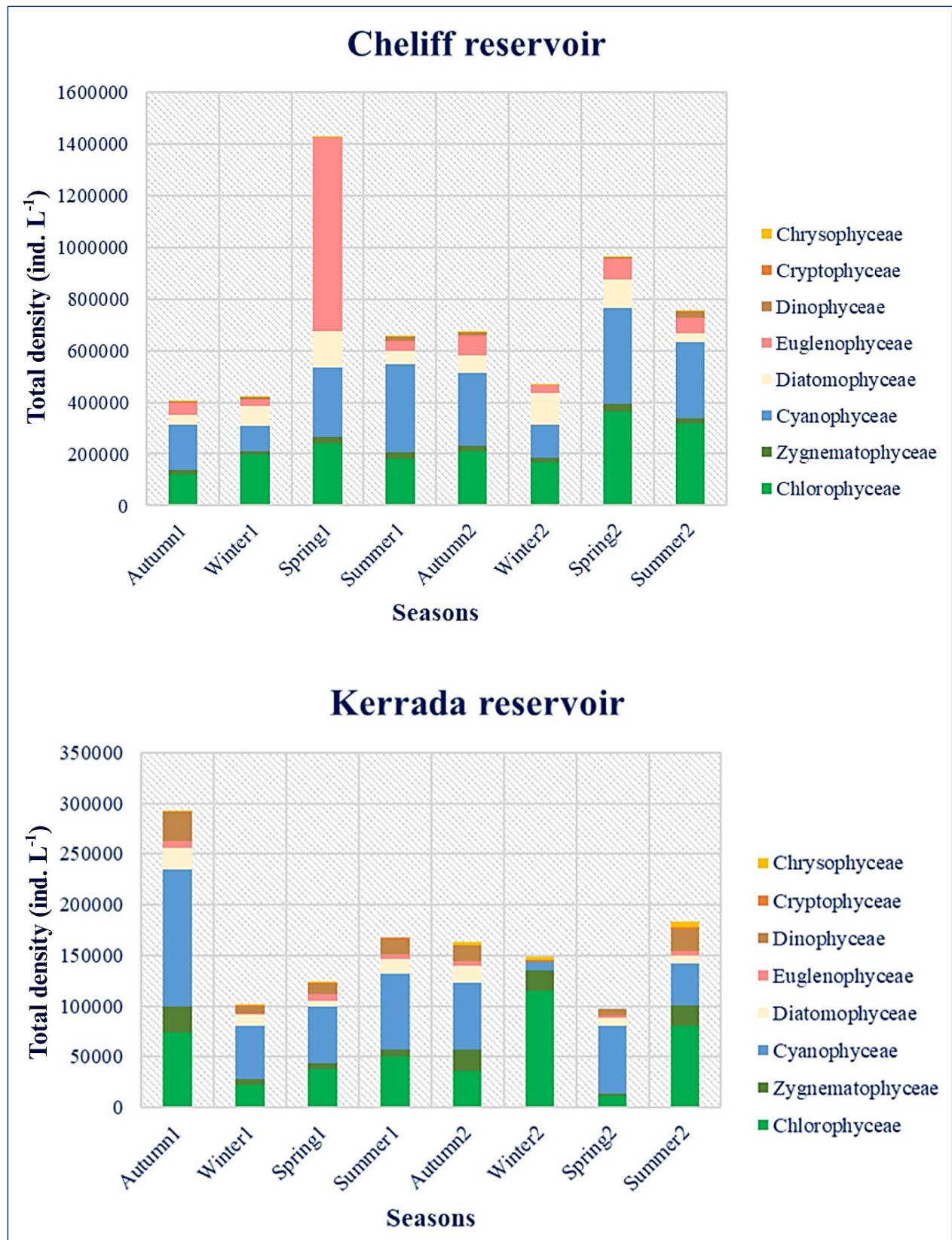


Fig. 44 Seasonal variation of phytoplankton density of different groups in Cheliff and Kerrada reservoirs.



IV.2.5. Environmental factors driving phytoplankton

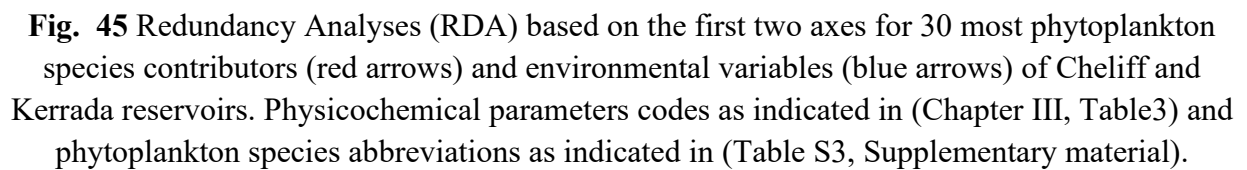
To visualize the relationships between forward-selected environmental variables (blue vectors) and phytoplankton species (red vectors), a redundancy analysis (RDA) was performed with reservoirs grouped (Fig. 45). The first two axes explained 34.91 % of the total variance, with Axis 1 accounting for 28 %. This axis represents the main environmental gradient separating Cheliff from Kerrada reservoir and is strongly driven by depth, which was positively correlated with Kerrada reservoir samples. This separation indicates distinct community compositions shaped by different environmental conditions.

Cheliff reservoir was associated with higher PM, CHL, turbidity, OM, NH_4^+ , TDS, NO_2^- , and salinity, favoring taxa tolerant to suspended solids such as Euglenophyceae and Chlorophyceae. Kerrada reservoir was mainly influenced by depth and NO_3^- , reflecting a more stratified and less turbid environment, favoring dinoflagellates (*Ceratium hirundinella*, *Peridinium cinctum*, and *Peridinium bipes*) and other taxa adapted to stable water columns (Fig. S8, Fig. S9, Table S4; Supplementary material).

Axis 2 explained 6.91 % of the variance and was positively correlated with DO. High-DO samples were associated with diatoms (*Stephanocylus meneghinianus*, *Discostella stelligera*, and *Nitzschia acicularis*), Euglenophyceae (*Euglenaria caudata*, *Lepocinclis acus*, *L. oxyuris*), and Chlorophyceae (*Schizochlamys gelatinosa*), suggesting that oxygen availability supports diatom dominance. Temperature was negatively aligned with Axis 2, indicating seasonal effects, with several Cyanophyceae (*Aphanocapsa elachista*, *Aphanothece variabilis*, *Chlorogloea microcystoides*, *Entophysalis arboriformis*, and *Microcystis aeruginosa*) associated with warmer periods. *C. microcystoides*, *A. variabilis*, and *E. arboriformis* were also linked to NO_3^- .

Chlorophyceae occurred across both reservoirs but were mainly linked to Cheliff reservoir, including *Desmodesmus communis*, *D. protuberans*, *Tetrademus lagerheimii*, *Monoraphidium griffithii*, and *M. contortum*, indicating tolerance to turbidity and nutrient enrichment. *Planctonema lauterbornii* was highly associated with Kerrada reservoir (depth, temperature), while *Strombomonas fluviatilis* was linked to OM in Cheliff reservoir.

Overall, Cheliff reservoir (shallow, turbid, nutrient-rich) is dominated by green algae and Cyanophycean adapted to eutrophic, mixed conditions, while Kerrada reservoir (deep, stratified) favors dinoflagellates and Cyanophycean species typical of thermally stable systems. These results support previous findings that reservoir morphometry controls light climate, nutrient cycling, and stratification patterns (Padisák et al., 2003; Reynolds, 2006). Seasonal DO and temperature





IV.3. Discussion

This study provides a comparative assessment of phytoplankton community structure and environmental drivers in two contrasting reservoirs: Cheliff, a shallow polymictic diversion reservoir, and Kerrada, a deep stratified storage reservoir. Our findings emphasize the central role of morphometry, mixing dynamics, and nutrient regimes in shaping phytoplankton structure, diversity, and seasonal succession. Spatial–temporal heterogeneity in physicochemical parameters was the main determinant of species composition, consistent with previous studies linking environmental variability to phytoplankton dynamics (Reynolds, 1984; Romo & Miracle, 1994; Winder & Hunter, 2008; Yang et al., 2016; Ochieng et al., 2022).

Phytoplankton species richness was significantly higher in Cheliff reservoir compared to Kerrada reservoir, supporting the hypothesis that shallow, polymictic ecosystems maintain higher diversity through continuous sediment–water nutrient exchange and frequent mixing events that prevent competitive exclusion (Naselli-Flores, 1999; Reynolds, 2006; Tang et al., 2025). High Shannon diversity ($H' = 3.6$) and evenness ($J' \approx 0.8$) further indicate a productive but balanced community. Kerrada reservoir, by contrast, exhibited slightly lower richness but a more evenly distributed community, consistent with the stabilizing effects of stratification and reduced disturbance (Moreno-Ostos et al., 2006; Bai et al., 2023). These patterns align with the intermediate disturbance hypothesis, whereby frequent mixing in Cheliff reservoir sustains high diversity by reducing competitive dominance (Townsend et al., 1997; Flöder & Sommer, 2000; Znachor et al., 2020; Peng et al., 2021).

Chlorophyceae were the most diverse group (45 species), reflecting their ecological plasticity across light and nutrient gradients (Reynolds, 2006; Barros et al., 2013). Diatoms (37 species) followed, strongly shaped by sediment resuspension in Cheliff reservoir and by pelagic, stratified, well-oxygenated conditions in Kerrada reservoir (Soininen, 2007; Chaïb & Tison-Rosebery, 2012). Minor groups such as Dinophyceae, Chrysophyceae, and Cryptophyceae, although species-poor, remained ecologically important due to mixotrophy and nutrient cycling (Jones, 1999; Mitra, 2024; Schenone et al., 2024).

Chlorophytes such as *Chlorella vulgaris* dominated both reservoirs, underscoring their high ecological tolerance, and contribution to oxygenation and nutrient cycling (Sendra et al., 2019; Dollah et al., 2024; Faruque et al., 2024). Strong presence of small, colony-forming green algae in Cheliff reservoir (*Actinastrum hantzschii*, *Monoraphidium contortum*, *Tetrademus lagerheimii*),



indicative of eutrophic, nutrient-enriched conditions (Deyab et al., 2021; Ruggeri et al., 2021; Sennour et al., 2025), which reflects the productive status of this shallow reservoir. Kerrada reservoir supported taxa adapted to stratification, such as *Planctonema lauterbornii* and *Parietochloris pseudoalveolaris* (Molina-Navarro et al., 2014; Chi et al., 2021), along with cryptophytes (*Cryptomonas pyrenoidifera*), highlighting the role of mixotrophy in deep, nutrient-poor waters (Becker et al., 2010; Wang H. et al., 2024).

Diatom patterns reflected morphometry-driven differences: Cheliff reservoir supported benthic and tychoplanktonic taxa (e.g., *Achnantheidium catenatum*, *Cocconeis placentula*), promoted by sediment resuspension (Berdalet et al., 1996; Carlsson & Granéli, 1999; Nehar et al., 2015). This is reflected by the dominance of *Stephanocyclus meneghinianus*, a tolerant bioindicator of eutrophic, low-oxygen conditions, and by the presence of *Cyclotella atomus* (hypereutrophic waters) and *Stephanodiscus neoastraea* (eutrophic, well-oxygenated waters) (Germain, 1981; El Haouati et al., 2015). *Pantocsekiella ocellata* and *Discostella stelligera* indicate variable salinity and oxygen conditions, with *D. stelligera* characteristic of shallow, well-mixed ecosystems (Schlegel & Scheffler, 1999; Fawzi et al., 2002; Chaïb & Tison-Rosebery, 2012; Adaouri et al., 2024). Its lower abundance in Kerrada reservoir likely reflects stratification limiting nutrient recycling. While Kerrada reservoir harbored pelagic, stratification-adapted forms (*Asterionella formosa*, *Fragilaria capucina*) characteristic of mesotrophic to oligotrophic conditions (Morais et al., 2018).

Cyanophyceae were abundant in both reservoirs but more dominant in Kerrada (46.6% of density). The prevalence of *Microcystis aeruginosa* is concerning due to its capacity for harmful bloom formation and toxin release (Paerl & Otten, 2013; Sharma & Tiwari, 2018; Arab et al., 2019; Ochieng et al., 2022; Guellati et al., 2024; Wang L. et al., 2024). The co-occurrence of nitrogen-fixing taxa (*Aphanocapsa elachista*) suggests nitrogen limitation during stratified periods, favoring diazotrophic cyanobacteria (Horváth et al., 2013; Paerl, 2017; Bhardwaj et al., 2024). Cheliff reservoir exhibited a more diverse cyanobacterial community, including filamentous forms (*Planktolyngbya limnetica*, *Raphidiopsis raciborskii*, *Planktothrix agardhii*), which are characteristic of hypertrophic, turbid ecosystems and often resistant to grazing (Padisák, 1997; Bidi-Akli et al., 2017; Pagni et al., 2020; Meriggi et al., 2024). In Kerrada, *Limnothrix redekei* and mucilaginous colonies (*Chlorogloea microcystoides* and *Aphanothece variabilis*) dominated, reflecting adaptations to low light and grazing pressure (Rojo & Cobelas, 1994; Moustaka-Gouni et al., 2007; Ray et al., 2021).



Euglenophyceae were more diverse in Cheliff reservoir, dominated by *Euglenaria caudata*, reflecting its shallow, nutrient- and organic-rich conditions (Alves-da-Silva & Tamanaha, 2008; Wołowski, 2011; Kouassi et al., 2013), whereas Kerrada reservoir was dominated by loricate *Trachelomonas* species, indicating adaptation to stratified, mesotrophic waters (Coesel & Krienitz, 2008). Zygnematophyceae diversity was higher in Kerrada reservoir, favored by stable, clear waters, while Cheliff reservoir supported higher biomass of *Closterium aciculare* and other desmids due to enhanced nutrient flux from sediments (Brook, 1981; Reynolds, 2006). Dinophyceae showed habitat partitioning, with *Peridinium cinctum* dominating Cheliff's well-mixed waters and *Ceratium hirundinella* thriving in Kerrada's stratified column (Sendra et al., 2019; Adaouri et al., 2024). Chrysophyceae were represented by *Dinobryon pediforme* in Kerrada reservoir (stable, oligotrophic conditions) and *Mallomonas caudata* in Cheliff reservoir (turbid, eutrophic waters) (Wang L. et al., 2024). The higher dominance of *Cryptomonas pyrenoidifera* in Kerrada reservoir highlights its sensitivity to water column stability and light conditions (Klaveness, 1988; Camacho et al., 2001; Becker et al., 2010). Overall, these patterns underline the strong influence of morphometry, stratification, and nutrient regimes in shaping phytoplankton functional group composition between the two reservoirs.

Phytoplankton density peaked in Cheliff reservoir during spring, promoted by mixing-driven nutrient fluxes and favoring chlorophytes and euglenoids (Wołowski, 2011; Benmatouk et al., 2024). The higher densities of Euglenophyceae in Cheliff reservoir during spring and summer contrast with the findings of Arab et al. (2019), who reported a decline in Euglenophyceae during these seasons. Kerrada reservoir exhibited autumn peaks after stratification breakdown, benefiting motile taxa such as dinoflagellates (*Ceratium hirundinella*) and cryptophytes (Reynolds, 2006; Becker et al., 2010; Bai et al., 2023; Varga et al., 2024).

Functional groups followed clear seasonal trends. In Cheliff reservoir Chlorophyceae and Euglenophyceae dominated in spring, diatoms in winter–spring, and filamentous cyanobacteria in summer. In Kerrada reservoir, cyanophyceae and chlorophyceae dominated in summer stratification, diatoms peaked in autumn overturn, and dinoflagellates thrived in summer–autumn, reflecting adaptation to stable, stratified conditions and low grazing pressure (Salmaso, 2000; Djabourabi, 2017; Benmatouk et al., 2024). These patterns reflect the replacement of diatoms by cyanobacteria under low turbulence and warm conditions, consistent with classical succession models (Sommer et al., 1986; Camacho et al., 2003; Moustaka-Gouni et al., 2007; Sendra et al., 2019).



RDA results confirmed that reservoir morphometry, depth, and DO–temperature gradients were the main drivers of phytoplankton community structure. Cheliff reservoir, with high turbidity and OM, favored tolerant taxa such as Euglenophyceae and Chlorophyceae. In contrast, Kerrada reservoir, defined by greater depth and higher nitrate availability, supported dinoflagellate and cyanobacterial dominance. These findings agree that morphometry controls light climate, mixing, and nutrient availability, shaping phytoplankton composition (Padisák et al., 2003; Kerimoglu & Rinke, 2013; Wang J.-Y. et al., 2022). High DO concentrations promoted diatoms (*Stephanocyclus meneghinianus*, *Discostella stelligera*, *Nitzschia acicularis*), confirming that oxygen availability promotes diatom dominance during well-mixed periods (Rolland et al., 2009; El Haouati et al., 2015; Nehar et al., 2015). Several Euglenophyceae (*Euglenaria caudata*, *Lepocinclis* spp.), and the chlorophyte *Schizochlamys gelatinosa*, also exhibited positive associations with DO.

In contrast, higher temperatures, negatively correlated with DO, favored cyanobacteria (*Aphanocapsa elachista*, *Aphanothece variabilis*, *Chlorogloea microcystoides*, *Entophysalis arboriformis*, *Microcystis aeruginosa*), consistent with spring–summer blooms under nutrient-rich conditions (Djabourabi, 2017; Arab et al., 2019; Manamani & Bensouilah, 2023; Belhaoues et al., 2024; Benmatouk et al., 2024; Sennour et al., 2025). Diatoms dominated cooler, phosphorus-rich periods (Wei et al., 2024), while high nitrate favored dinoflagellates, and phosphorus limitation shifted communities toward cyanobacteria (Kwon et al., 2009; Dashkova et al., 2022). Chlorophyceae were broadly distributed, strongly associated with Cheliff reservoir (*Desmodesmus communis*, *Tetradescmus lagerheimii*, *Monoraphidium* spp.) under eutrophic, turbid conditions (Sendra et al., 2019; Molner et al., 2025), whereas *Planctonema lauterbornii* characterized Kerrada’s deeper, stratified environment. Overall, these results confirm that Cheliff’s shallow polymictic regime supports green algae and cyanobacteria, while Kerrada favors dinoflagellates and bloom-forming cyanobacteria, with succession strongly regulated by seasonal DO–temperature dynamics (Padisák et al., 2003; Reynolds, 2006). The ability of cyanobacteria to regulate buoyancy through gas vesicles enables vertical migration and facilitates bloom formation (Bidi-Akli et al., 2017).

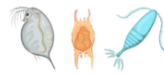
Overall, our findings underscore the role of morphometry, mixing dynamics, and nutrient regimes in structuring phytoplankton communities. Cheliff reservoir represents a polymictic, eutrophic ecosystem dominated by opportunistic taxa, whereas Kerrada reservoir reflects a more stable, stratified ecosystem where cyanobacteria and dinoflagellates thrive. These patterns highlight the need for targeted monitoring and management strategies that account for



hydrodynamics and morphometric differences to prevent harmful blooms and maintain ecological balance.

Chapter V: Zooplankton community structure





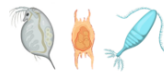
This chapter examines zooplankton community structure and dynamics in the two study reservoirs, over two consecutive years. Building upon the physicochemical and phytoplankton context established in the preceding chapters, this analysis focuses on the major freshwater zooplankton groups: Rotifera, Copepoda, and Cladocera. A preliminary overview of this study was presented in Baitiche et al. (2025); here, we expand that work through a detailed assessment of temporal and spatial patterns in diversity, abundance, and community composition. The main objectives are to characterize differences in zooplankton assemblages between the two reservoirs and to evaluate how morphometry, depth gradients, and water-level fluctuations structure these communities. Particular attention is given to identifying which groups exhibit the highest sensitivity to environmental variability, especially in the contrasting contexts of shallow polymictic versus deep monomictic stratified ecosystems. The findings contribute to a deeper understanding of how zooplankton respond to environmental forcing in contrasting reservoir types and provide valuable insights for reservoir management, long-term ecosystem monitoring, and biodiversity conservation. Moreover, the assessment of zooplankton community structure offers critical context for the following chapter, which evaluates the water quality and trophic status of these freshwater ecosystems. Understanding zooplankton dynamics is therefore essential for linking environmental variability to ecosystem functioning and for informing sustainable reservoir management.

V. 1. Material and methods

V. 1. 1. Sampling procedure

Zooplankton samples were collected monthly over a two-year period, from November 2020 to October 2022. In each reservoir, four sampling sites were selected to capture spatial variability (Fig. 24; Table 3, Chapter III). Samples were taken across three depth layers, following the same procedure applied for physicochemical parameters and phytoplankton. Quantitative sampling was performed by filtering 5 L of water through a conical plankton net with a 50 μm mesh size (SDMO QUINIOU, Model F131P50) at each site and by collecting water a Van Dorn bottle (5 L), following the method described by Miracle et al. (1992) (Fig. S4, Supplementary material). Additionally, qualitative vertical samples were obtained by towing the plankton net from near the bottom to the surface. All samples were preserved in formaldehyde (4%) and transported to the laboratory for identification and counting.

Zooplankton density (ind. L^{-1}) was estimated by averaging the number of individuals per species counted in multiple aliquots examined under an inverted microscope (ZEISS-Winkel)



using a sedimentation chamber (10 mL), following the methods of Wetzel and Likens (1991) and Bellinger and Sigee (2010), with differential interference contrast (DIC) microscopy. Additional observations were conducted with a stereomicroscope (dissecting microscope) (Fig. S10, Supplementary material), which is particularly useful for identifying microcrustaceans. Enumeration was facilitated using zooplankton counting platforms, Dollfus chambers, or Petri dishes subdivided into equal squares, with manual counters often used to assist in species counting.

For taxonomic identification, sub-samples (10 mL) were taken from homogenized samples and examined under a stereomicroscope (40× magnification) and an inverted microscope (Nikon Eclipse Ti-U) equipped with a digital camera (Fig. S7, Supplementary material).

Species identification was based on: Dussart (1967, 1989), Akli (1992), Dussart and Defaye (2002, 2006), Błędzki and Rybak (2016), and Alekseev et al. (2021) for **Copepoda**; Koste (1978), Nogrady and Segers (2002) for **Rotifera**; Ould Rouis (1995), Alonso (1996), and Błędzki and Rybak (2016) for **Cladocera**.

V. 1. 2. Zooplankton counting and density calculation

a) Sedimentation chamber

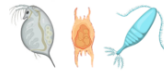
After collection and concentration, zooplankton samples were analyzed using the sedimentation chamber technique. When organism density was low, the entire sample was allowed to settle and all individuals were counted under an inverted microscope. Zooplankton density (D , ind. L^{-1}) was calculated following the methods of Wetzel and Likens (1991) and Bellinger and Sigee (2010):

$$D = \frac{N_i \times 1000}{V}$$

N_i : number of individuals counted for each species; V : sedimented volume (10 mL); and 1000 is the conversion factor used to express density per liter.

b) Sedgewick–Rafter chamber

Quantitative enumeration of zooplankton was carried out using a Sedgewick–Rafter (SR) counting chamber (1 mL capacity; 50 mm × 20 mm × 1 mm) (Fig. S11, Supplementary material). When the sample contained a high number of organisms, subsamples were taken



to facilitate counting (Mack et al., 2012). Prior to counting, each preserved sample was gently homogenized by thorough mixing to ensure even distribution of organisms. A subsample (1mL) was then carefully pipetted into the chamber, avoiding the introduction of air bubbles. The chamber was covered with a clean glass coverslip to prevent evaporation and to maintain a uniform liquid surface.

Counting was performed under a compound microscope at magnifications appropriate to organism size. The chamber contains a grid of 1,000 fields (each 1 mm²). Either the entire chamber was examined, or a representative number of randomly selected fields were counted when organism density was very high. The number of individuals per liter was calculated according to the following formula:

$$D = \frac{N_i \times V_c}{a \times d \times V_s}$$

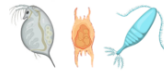
D: zooplankton density (ind. L⁻¹); **N_i:** number of individuals counted; **V_c:** total volume of the concentrated sample (mL); **a:** area of the fields counted (mm²); **d:** depth of the chamber (mm = 1); **V_s:** volume of the subsample (mL).

At least **three replicates** were counted per sample to minimize subsampling error, and results were expressed as mean values. Subsampling introduces error, so care must be taken to ensure proper mixing and pipetting (Błędzki & Rybak, 2016).

V. 1. 3. Statistical analyses

Statistical analyses were performed on physicochemical and biological (zooplankton) datasets obtained throughout the study period from all stations across different levels of each reservoir. Data were grouped by season and reported as average values. Environmental parameters are presented in Chapter III.

To assess biodiversity patterns, we calculated Shannon's diversity index (H') (Shannon & Weaver, 1949) and Pielou's evenness index (J') (Pielou, 1966). Additional ecological attributes, including species richness, density (ind. L⁻¹), relative abundance, Shannon index, and evenness index, were computed using PAST software (Hammer et al., 2001).

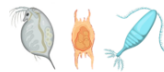


To evaluate spatio-temporal differences in diversity indices, a Mann–Whitney U test (Wilcoxon rank-sum) was applied to compare Cheliff and Kerrada reservoirs. Seasonal differences were visualized with boxplots. These non-parametric tests were chosen due to violations of normality and homogeneity of variances.

Redundancy Analysis (RDA) was then applied to examine the influence of environmental variables on zooplankton community composition, using a forward selection procedure to identify significant predictors. Biotic data were transformed using the Hellinger method, and environmental variables were log-transformed $\log(x + 1)$, except for pH, to satisfy assumptions of homoscedasticity and normality (Legendre & Gallagher, 2001). To emphasize dominant taxa, only zooplankton species contributing more than 0.5% of the total density were included in the analysis. The significance of ordination axes and explanatory variables was tested with Monte Carlo permutations under the reduced model.

To further assess relative contributions of phytoplankton groups and environmental variables to zooplankton community structure, variance partitioning and RDA were performed (Oksanen et al., 2022). This analysis used to quantify the unique and shared fractions of explained variance attributable to each predictor set, with significance tested by permutation.

All statistical tests were conducted at a 5% significance level. Data analysis and graphical visualization were conducted in R software (version 4.3.2), with plots generated using the *vegan* and *ggplot2* packages.



V. 2. Results

V.2.1. Evolution of zooplankton diversity in Cheliff and Kerrada reservoirs

A total of **29** zooplankton species were identified across the two reservoirs, with significantly higher species richness in Cheliff reservoir (**28 species**) compared to Kerrada reservoir (**20 species**) (Table S5, Supplementary material). Rotifers were the most diverse group (19 species), followed by cladocerans (7 species) and copepods (3 species). Notably, six rotifer species: *Brachionus durgae*, *Brachionus plicatilis*, *Colurella cohurus*, *Keratella hiemalis*, *Pompholyx sulcata*, and *Synchaeta pectinata* were absent in Kerrada reservoir. Among the cladocerans, *Ceriodaphnia reticulata* and *Daphnia magna* were found exclusively in Cheliff reservoir, while *Leptodora kindtii* was observed only in Kerrada reservoir. The copepod *Eucyclops serrulatus* was also absent in Kerrada reservoir.

Species richness (S) and Pielou's evenness (J') differed significantly between the two reservoirs (Wilcoxon test, $p < 0.05$) (Fig. 46), with Cheliff reservoir showing higher mean species richness (11 species). No significant difference in Shannon diversity index was observed between the reservoirs (Wilcoxon test, $p > 0.05$) (Cheliff; $H' = 1.3$; Kerrada; $H' = 1.4$); however, Pielou's evenness index was significantly lower in Cheliff reservoir ($J' = 0.6$), indicating a more uneven species distribution dominated by cladocerans (*Bosmina longirostris*, *Moina micrura*) and rotifers (*Keratella valga*), in comparison with Kerrada reservoir ($J' = 0.8$).

Seasonal variation in zooplankton diversity and evenness was significant in both reservoirs, with consistently higher diversity during the second sampling year, particularly in Summer and Autumn 2 (Fig. 47). Shannon diversity maximum values were recorded in Cheliff reservoir in Spring 2 ($H' = 2.6$), and maximum in Kerrada reservoir were recorded in Autumn 2 ($H' = 2.4$). Pielou's evenness varied between 0.2 (Cheliff, Winter 2) and 1 (Cheliff, Winter 1), and between 0.3 (Kerrada, Summer 2) and 1 (Kerrada, Summer 1).

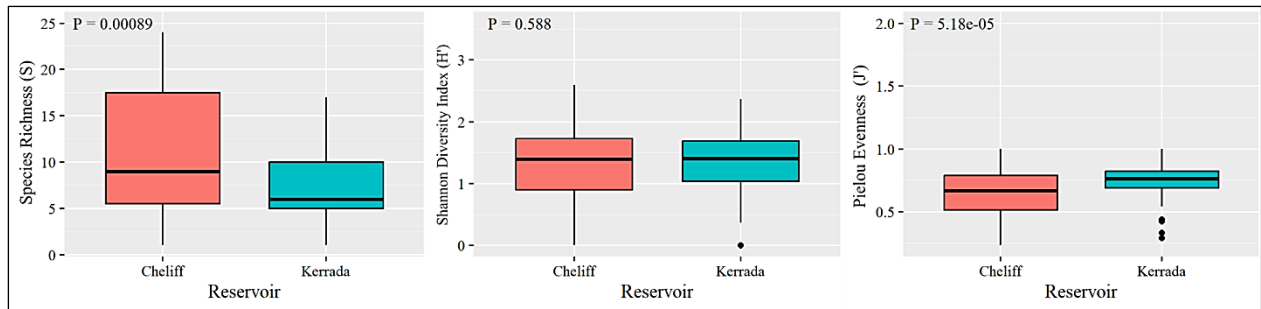
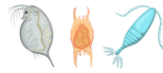


Fig. 46 Comparison of diversity indices (Shannon index H' , Pielou evenness J' , and Species richness S) between Cheliff and Kerrada reservoirs (Wilcoxon test, 95% Confidence Interval).

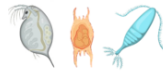
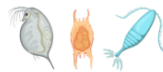


Fig. 47 Seasonal variation of diversity indices Species Richness (S), Shannon Index (H'), and Pielou's Evenness (J') in Cheliff and Kerrada reservoirs. The boxplots display the distribution of values for each season, where the central line represents the median, the box limits indicate the interquartile range (IQR), and the whiskers extend to the minimum and maximum values (excluding outliers).



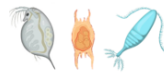
V.2.2. Zooplankton groups diversity and dominance

a) Rotifers

Rotifer assemblages showed contrasting patterns of dominance between the two reservoirs (Fig. 48). Rotifers were dominated by *Keratella valga* in Cheliff reservoir, accounting for 28.1% of the total rotifer dominance (D), and by *Hexarthra mira* in Kerrada reservoir (D = 42.4%). The ecological success of *K. valga* in shallow, nutrient-rich ecosystems has been widely reported, as it tolerates fluctuations in food resources and turbidity (Ejsmont-Karabin, 2012; Umi et al., 2018; Chakrabarti, 2023). Conversely, *H. mira* often thrives in deeper, more stable water bodies with higher transparency, where its fast-swimming ability provides a competitive advantage under reduced predation pressure (Sarma et al., 2005; Picapedra et al., 2021).

Among the most dominant taxa in Cheliff reservoir, *Synchaeta pectinata* (D = 23.8%), *Brachionus angularis* (D = 19.7%), and *Polyarthra dolichoptera* (D = 10.7%) contributed substantially to the community. *S. pectinata* is typically associated with eutrophic conditions and is known as a suspension feeder on algae and detritus, often proliferating in shallow, nutrient-enriched habitats (Sládeček, 1983; Ochocka & Pasztaleniec, 2016; Umi et al., 2018; García-Chicote et al., 2019; Picapedra et al., 2019). *B. angularis* is a cosmopolitan and opportunistic species with high tolerance to organic enrichment and variable food availability, making it a reliable bioindicator of meso- to eutrophic environments (Sládeček, 1983; Muñoz-Colmenares et al., 2021). *P. dolichoptera*, a highly motile, and filter-feeding rotifer, is often dominant in reservoirs with moderate trophic status and fluctuating food resources; its ability to feed on flagellated algae and bacteria gives it a competitive edge in both shallow and deep systems (Gulati, 1983; May & O'Hare, 2005).

In Kerrada reservoir, besides the dominance of *H. mira*, notable contributions were made by *Keratella tropica* (D = 17.5%), *P. dolichoptera* (D = 15.3%), and *Keratella quadrata* (D = 7.4%). *K. tropica* is characteristic of tropical and subtropical water bodies and shows preference for warm, stable environments, often linked to mesotrophic to eutrophic conditions (Segers, 1995; Nogueira, 2001). *K. quadrata*, on the other hand, is a eurytopic species widely distributed across diverse freshwater habitats and is known for its morphological plasticity, which provides defense against predation by cladocerans and copepods (Dumont, 1983; Stemberger & Gilbert, 1985; Sarma & Nandini, 2019).



The presence of *K. tropica* in Kerrada reservoir, contrasted with its low dominance in Cheliff reservoir ($D = 0.5\%$), highlights ecological dynamics influenced by species competition and environmental conditions. This pattern is likely due to the high dominance of *K. valga* in Cheliff reservoir, a competitive species that may limit the establishment of *K. tropica*. The coexistence of these two rotifers reflects their competitive interactions, allowing *K. tropica* to persist in Kerrada reservoir while remaining rare in Cheliff reservoir.

The dominance of *P. dolichoptera* in Kerrada reservoir can be attributed to its adaptability to deep stratified ecosystems, enabling it to efficiently exploit resources throughout the water column and to tolerate relatively lower nutrient concentrations. This ecological flexibility is supported by several studies where *P. dolichoptera* thrives in diverse aquatic environments, often under conditions less favorable for other rotifers (García-Chicote et al., 2019; Muñoz-Colmenares et al., 2021; Picapedra et al., 2021). In tropical lentic ecosystems, its presence is associated with high water transparency and nutrient availability, supporting its ability to utilize a broad range of resources (Umi et al., 2024).

The occurrence of *K. tropica* and *K. quadrata* alongside *P. dolichoptera* reflects the adaptability of Kerrada's rotifer assemblages to deeper and more stable environmental conditions compared with Cheliff reservoir. This adaptability is evident in the species' seasonal dynamics and their responses to various environmental factors.

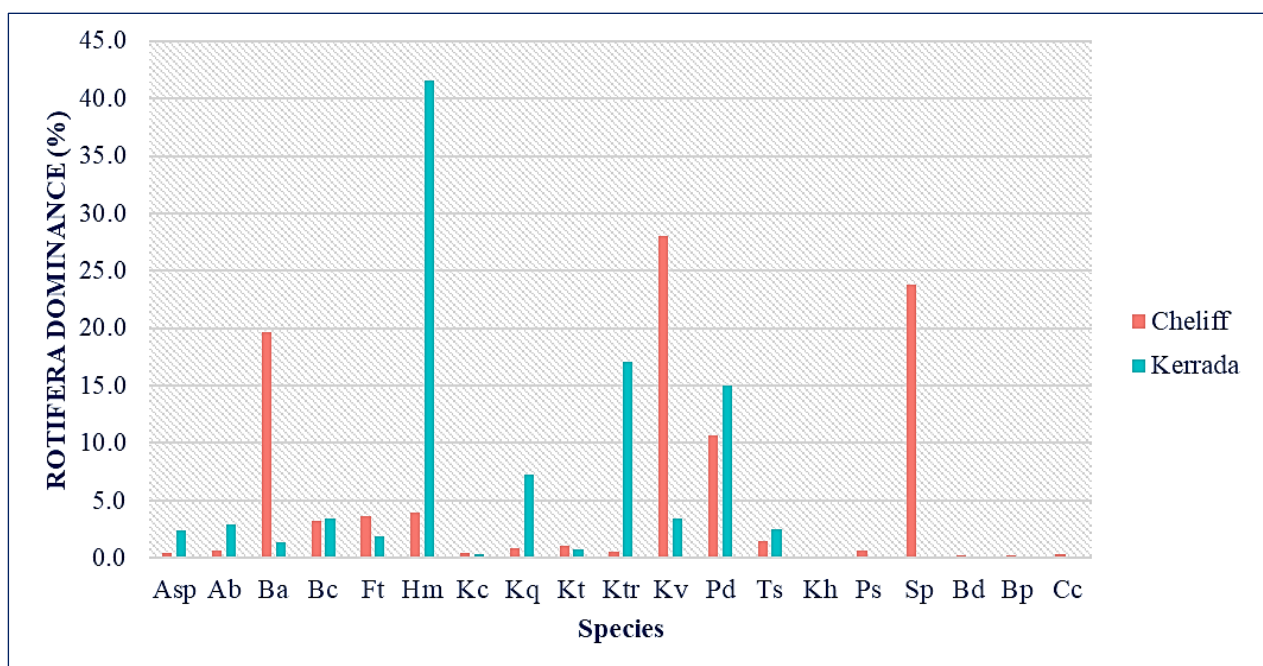
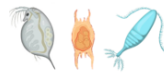


Fig. 48 Rotifera species dominance in Cheliff and Kerrada reservoirs (Species codes; Table S.5, Supplementary material).



b) Cladocerans

Cladocerans were more abundant in Kerrada reservoir (Fig. 49), where the community was strongly dominated by *Diaphanosoma brachyurum* (D = 56.2%). This species is typical of large, deep, and stratified reservoirs, where its high filtering efficiency and ability to exploit nanoplanktonic algae allow it to dominate in mesotrophic to eutrophic ecosystems (Stich, 2004; Choi et al., 2014; Muñoz-Colmenares et al., 2021). *D. brachyurum* has also been reported as a dominant taxon in several Mediterranean reservoirs, where its abundance is linked to water transparency and lower predation pressure from fish (Muñoz-Colmenares et al., 2021).

In Cheliff reservoir, *Bosmina longirostris* was the most dominant cladoceran species (D = 33.7%), followed closely by *Moina micrura* (D = 31.1%). *B. longirostris* is a small-bodied cladoceran often associated with eutrophic and shallow ecosystems, where it exhibits high tolerance to fluctuating environmental conditions and turbidity (Adamczuk, 2016; Nevalainen et al., 2018; Sehili et al., 2020). The predominance of *B. longirostris* in Algeria, particularly in arid and semi-arid regions, has been reported by Bouzidi et al. (2010) and Sehili et al. (2020). Its dominance in both reservoirs (D = 32.9% in Kerrada) indicates its opportunistic feeding strategies and ability to coexist with larger cladocerans under different trophic regimes (Pinto et al., 2021b; Muñoz-Colmenares et al., 2021; Stamou et al., 2021). *M. micrura*, another cosmopolitan species, thrives in nutrient-rich shallow water bodies and is considered a reliable indicator of eutrophication in Mediterranean lakes and reservoirs (Alonso, 1996). Its success in Cheliff is likely related to high food availability and tolerance to warm, turbid conditions typical of shallow reservoirs in North Africa.

Leptodora kindtii was present exclusively in Kerrada reservoir (D = 5%). This predatory cladoceran is usually restricted to deep, clear lakes and reservoirs, where it preys on smaller zooplankton such as rotifers and small cladocerans (Amoros, 1984; Vogt et al., 2013; Bowersox et al., 2014). Its occurrence in Kerrada reservoir suggests the presence of a more structured pelagic food web, consistent with the greater depth and stability of this reservoir compared to Cheliff. In contrast, *Ceriodaphnia reticulata* (D = 16.8%) and *Daphnia magna* (D = 10.6%) were recorded only in Cheliff reservoir. *C. reticulata* tends to occupy deeper layers of the water column, thriving in shallow ecosystems and nutrient-rich environments (Ould Rouis et al., 2022). *D. magna* is a large-bodied cladoceran widely used as a bioindicator due to its sensitivity to eutrophication and pollution, but it can persist in shallow and nutrient-rich reservoirs when fish predation is moderate (Wagner et al., 2005; Bidi-Akli et al., 2014; Almeida et al., 2020).

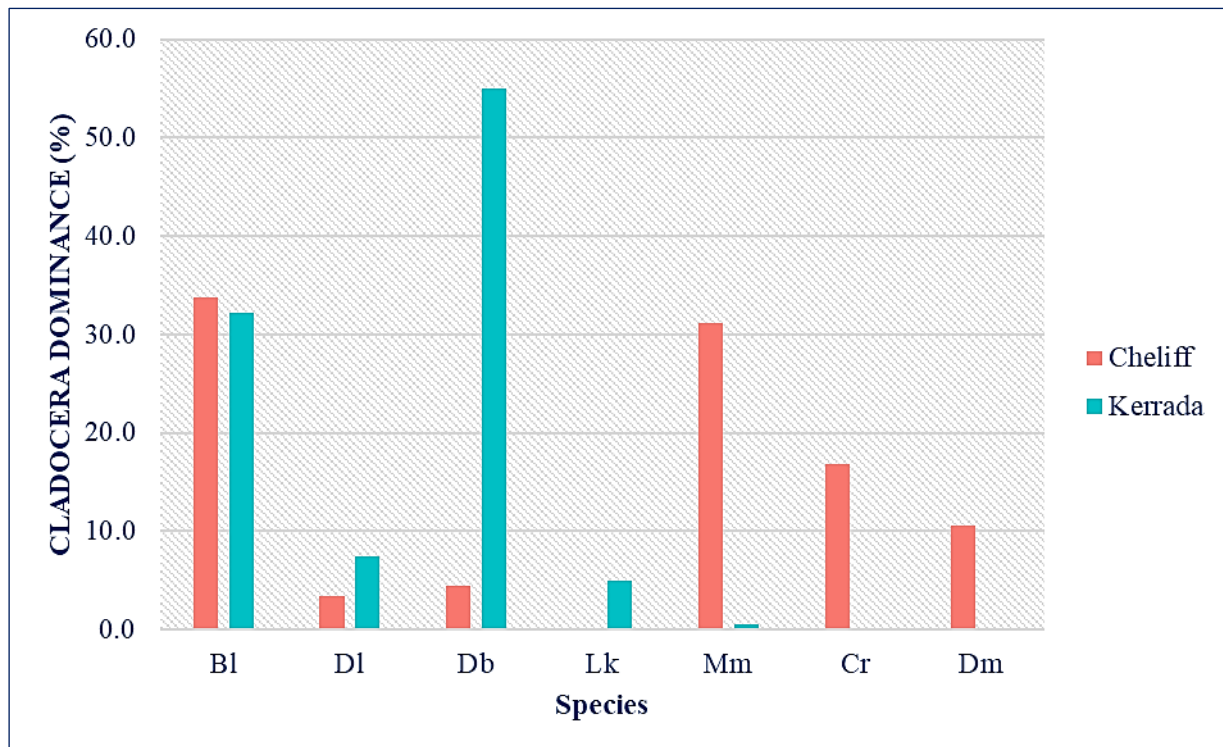
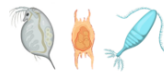


Fig. 49 Cladocera species dominance in Cheliff and Kerrada reservoirs (Species codes; Table S.5, Supplementary material).

c) Copepods

Copepod assemblages exhibited contrasting patterns of dominance between the two reservoirs (Fig. 50). In Cheliff reservoir, cyclopoid copepods were overwhelmingly dominant, with *Acanthocyclops americanus* contributing 93.4% of the assemblage. This species is known to thrive in shallow, eutrophic, and variable freshwater environments (Boxshall & Defaye, 2008; Alekseev & Defaye, 2011; Montagud et al., 2019; Muñoz-Colmenares et al., 2021). In contrast, Kerrada reservoir was dominated by calanoids, particularly *Copidodiaptomus numidicus* (79.7%), a North African endemic species typically associated with deeper, meso- to oligotrophic water bodies (Defaye & Dussart, 1988; Mouelhi et al., 2000; Marrone et al., 2017; García-Chicote et al., 2019; Ould Rouis et al., 2022; Akli-Bidi et al., 2023). *Eucyclops serrulatus*, a cosmopolitan cyclopoid often found in littoral zones and tolerant to fluctuating conditions (Boxshall & Defaye, 2008), occurred exclusively in Cheliff reservoir, albeit at very low dominance (3.3%).

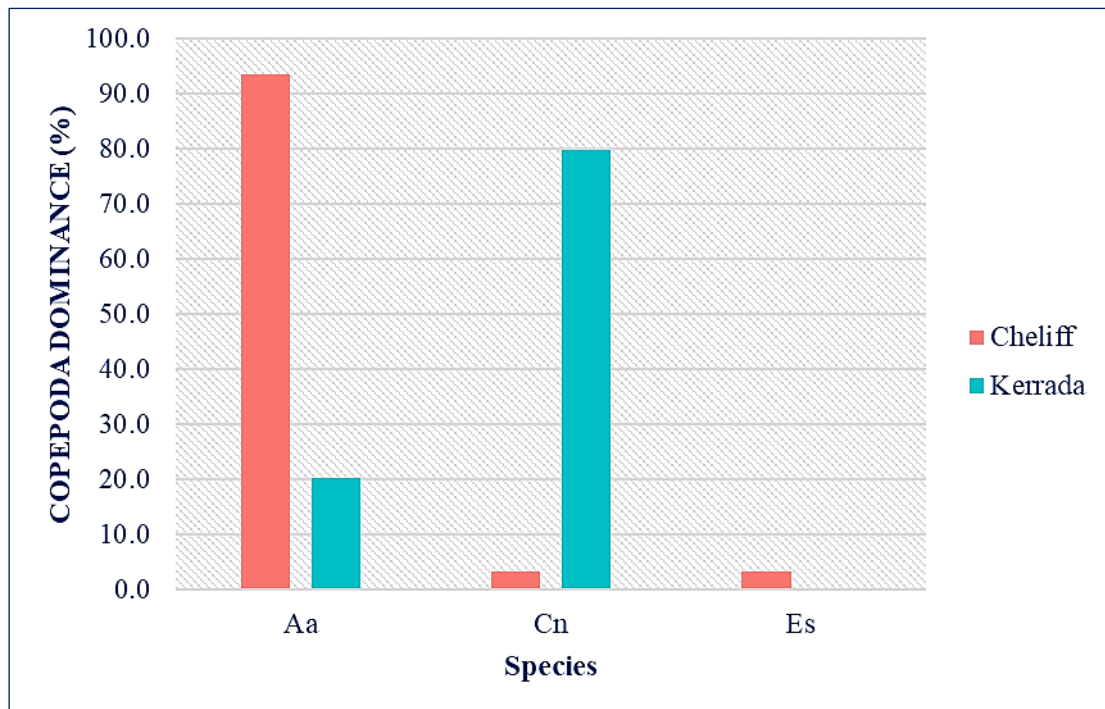
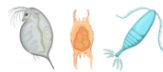


Fig. 50 Copepoda species dominance in Cheliff and Kerrada reservoirs (Species codes; Table S.5, Supplementary material).

V.2.3. Comparative patterns of zooplankton relative abundance

According to the relative abundance heatmap of zooplankton species (Fig. 51), zooplankton community in Cheliff reservoir is strongly dominated by a few taxa, as indicated by the dark-blue bars on the left. *K. valga* exhibited the highest relative abundance (26.3%), followed by *S. pectinata* (22.3%), *B. angularis* (18.4%), and *P. dolichoptera* (10%). Most other species occurred with very low dominance (< 5%), represented in yellow shades, indicating that Cheliff reservoir hosts a community with a pronounced monodominant structure driven by rotifers, characterized by low evenness. This pattern reflects environmental conditions that favor opportunistic and resilient taxa typical of shallow, more eutrophic ecosystems.

In contrast, zooplankton community in Kerrada reservoir was more evenly distributed among several species. The most abundant taxa were *D. brachyurum* (25.1%), *H. mira* (17.7%), *B. longispina* (14.7%), *C. numidicus* (9.5%), *K. tropica* (7.3%), and *P. dolichoptera* (6.4%), while the remaining species contributed less than 5%. This more balanced dominance pattern indicates a higher evenness and community stability compared to Cheliff reservoir, likely linked to the deeper and less eutrophic conditions of Kerrada reservoir, which can sustain multiple coexisting taxa.

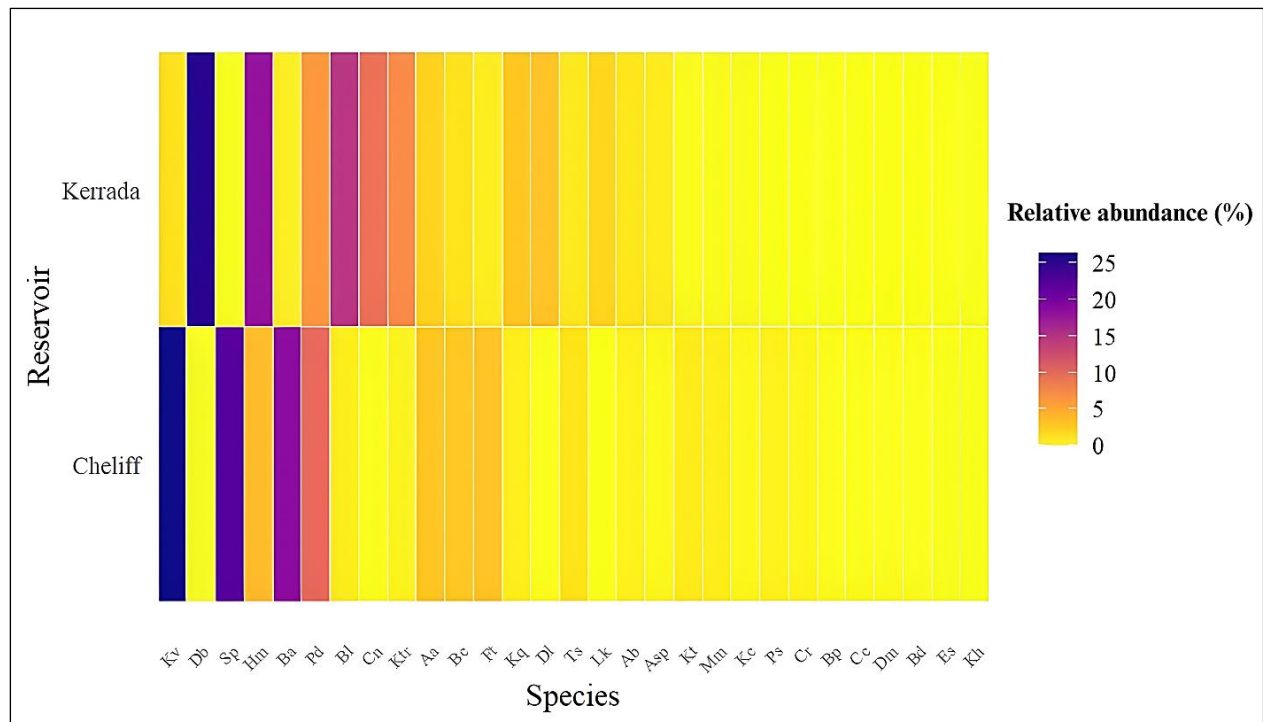
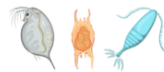
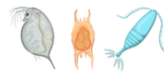


Fig. 51 Dominance Heatmap of zooplankton species over two years of sampling in Cheliff and Kerrada reservoirs.

V. 2.4. Seasonal changes in zooplankton group densities

Zooplankton density exhibited pronounced fluctuations between Cheliff and Kerrada reservoirs (Fig. 52). In Cheliff reservoir, mean total zooplankton density reached 479 individuals per liter (ind. L⁻¹), with marked peaks observed in Winter 2 (917 ind. L⁻¹), Summer 2 (873 ind. L⁻¹), and Autumn 2 (869 ind. L⁻¹). In contrast, Kerrada reservoir supported substantially lower mean densities (~40 ind. L⁻¹), with its maximum recorded in Autumn 1 (90 ind. L⁻¹). Minimum values were recorded in Winter 1 in Cheliff reservoir (39 ind. L⁻¹) and in Winter 2 in Kerrada reservoir (9 ind. L⁻¹). The elevated zooplankton densities in Cheliff reservoir were primarily driven by rotifers, which represented 94% of total abundance, while Kerrada community showed greater variability among taxonomic groups, with cladocerans (46%) slightly surpassing rotifers (42%) (Table S5, Supplementary material). Throughout the study period, rotifer *K. valga* was the most abundant species in Cheliff reservoir, with a mean density of 124.6 ind. L⁻¹.

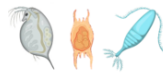
In Cheliff reservoir, the highest seasonal peaks were caused by abrupt proliferations of several rotifer species. *S. pectinata* appeared for the first time in Winter 2 at ST3, with extremely high surface densities (9000 ind. L⁻¹), thereby contributing substantially to the exceptional zooplankton winter peak. Such sudden pulses are typically associated with nutrient enrichment following mixing and runoff events (Guermazi et al., 2023; Orduna et al., 2024). After this initial



outbreak, *S. pectinata* persisted at moderate densities (lower densities ~ 20 ind. L^{-1} in summer), reflecting a shift toward new equilibrium conditions. Other rotifers showed similar seasonal dynamics aligned with temperature and resource availability. *Filinia terminalis* contributed notably in Winter 2 (456 ind. L^{-1} at ST3) and persisted through subsequent seasons at mean densities of ~ 40 ind. L^{-1} . In Summer 2, *P. dolichoptera* reached 1002 ind. L^{-1} at ST3, and *H. mira* appeared for the first time in the same season (876 ind. L^{-1} at ST3), later persisting at lower densities (~ 90 ind. L^{-1}). *B. angularis* peaked in Autumn 2 (1070 ind. L^{-1} at ST3), coinciding with a decline in water level and the drying of ST4, conditions that often increase organic matter concentration and enhance resource availability (Dokulil & Teubner, 2000; García-Chicote, 2019; Pinto et al., 2023). *K. valga* exhibited a distinctive peak in Spring 1 at ST1 (1649 ind. L^{-1} at 4 m depth), with lower densities at the surface (292 ind. L^{-1}) and bottom (666 ind. L^{-1}), suggesting behavioral responses linked to feeding strategies and avoidance of surface turbulence (Umi et al., 2018; Chakrabarti, 2023).

Copepods also contributed to seasonal fluctuations in Cheliff reservoir. *A. americanus* reached ~ 100 ind. L^{-1} in both autumn seasons at ST2 and ST3, a period consistent with the completion of its reproductive cycle following summer warming (Sarma et al., 2019; Alekseev et al., 2021). Adult copepods disappeared in December 2022, although juveniles (~ 50 ind. L^{-1}) and nauplii (~ 5 ind. L^{-1}) remained present, indicating a population bottleneck driven by declining temperatures. The highest naupliar densities (~ 336 ind. L^{-1}) occurred in Spring 1 and Autumn 1 at ST2 and ST3, reflecting synchronized reproductive events (Alekseev, 2021). *E. serrulatus* first appeared in September 2021 (10 ind. L^{-1}) and reappeared intermittently during the second year, always at low densities (< 5 ind. L^{-1}), mainly at ST4. This pattern suggests periodic introductions or the persistence of small littoral populations (Boxshall & Defaye, 2008). Its wide distribution has historically been interpreted as cosmopolitanism, although recent studies indicate that some records may reflect recent invasions (Akli-Bidi, 2022).

Cladoceran peaks in Cheliff reservoir occurred mainly in the autumn seasons and in Summer 2. *B. longirostris* reached 200 ind. L^{-1} at ST2 in Autumn 1, while *M. micrura* attained 55 ind. L^{-1} in Summer 2 (ST2). *C. reticulata* appeared in Spring 2 and reached 35 ind. L^{-1} in Autumn 2 at ST2. *D. magna* peaked in Autumn 2 (26 ind. L^{-1} at ST2), having first been recorded in October 2021 (5 ind. L^{-1} at ST1 and ST4), then disappearing temporarily before reappearing continuously in Summer 2 and Autumn 2 (mean densities ~ 20 ind. L^{-1}). These cladoceran patterns reflect the species' preference for moderate temperatures, increased phytoplankton availability, and reduced

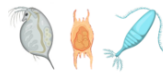


fish predation in late summer and autumn (Bidi-Akli et al., 2014; Sehili et al., 2020; Pinto et al., 2021b; Stamou et al., 2021). The reappearance of *D. magna* supports the role of predation cycles and temperature thresholds in regulating large-bodied cladocerans, which can persist in shallow, nutrient-rich reservoirs when predation pressure is moderate (Wagner et al., 2005; Bidi-Akli et al., 2014; Almeida et al., 2020).

In Kerrada reservoir, the highest zooplankton densities occurred in Autumn 1 and were dominated by cladocerans. *B. longirostris* reached 342 ind. L⁻¹ at ST7 (15–30 m depth) and remained present through both years except in winter, indicating diel or seasonal vertical migration driven by thermal stratification and potential predator avoidance (Bouzidi et al., 2010; Adamczuk, 2016; Stamou et al., 2021). *D. brachyurum* also peaked in Autumn 1 (110 ind. L⁻¹ at ST7 surface) and was omnipresent, with higher densities in surface layers. In October, however, it exhibited greater abundance around 20 m depth, likely reflecting vertical responses to changing thermal structure. Its deeper distribution complements its high filtering efficiency and capacity to exploit nanoplanktonic algae (Stich, 2004; Choi et al., 2014; Muñoz-Colmenares et al., 2021). *Daphnia longispina* reached 30 ind. L⁻¹ in Autumn 1 but disappeared during winter. *L. kindtii* was most abundant at 10–30 m depth in Summer 2 (~30 ind. L⁻¹) and was absent in winter, a pattern typical of mesotrophic ecosystems where transparency favors visually hunting predators. Its absence in winter suggests thermal constraints and limited prey availability (Amoros, 1984; Vogt et al., 2013; Bowersox et al., 2014).

Among rotifers, *H. mira* was a major driver of autumn peaks in Kerrada reservoir, reaching 574 ind. L⁻¹ at ST7 in Autumn 1. This species consistently exhibited autumn maxima and very low winter abundances, indicating favorable conditions following stratification breakdown (Sommer et al., 2012; Picapedra et al., 2021). *P. dolichoptera* peaked in Summer 2 (106 ind. L⁻¹ at ST5) and occurred year-round with minimal winter presence. *K. tropica* reached ~50 ind. L⁻¹ in Summer 2 across all stations, displaying typical warm-season peaks associated with stable epilimnetic layers (Nogueira, 2001; May & O'Hare, 2005; Umi et al., 2024).

Copepods in Kerrada reservoir exhibited their highest densities during Autumn 1, Autumn 2, and Spring 2. *C. numidicus* reached ~100 ind. L⁻¹ in Autumn 1, accompanied by elevated juvenile (1,512 ind. L⁻¹) and naupliar (156 ind. L⁻¹) densities at ST7. This species was omnipresent, with minimal densities in summer and consistently lower values in deep waters (~10 ind. L⁻¹) compared with surface layers. These patterns reflect the reproductive strategy of calanoid copepods



in stratified reservoirs, where reproduction aligns with spring phytoplankton blooms and autumn mixing. Higher surface densities correspond to favorable oxygen and food conditions (Błędzki & Rybak, 2016; Marrone et al., 2017; García-Chicote et al., 2019; Min et al., 2021). *A. americanus* reached 20 ind. L⁻¹ in autumn but maintained very low mean densities (~3 ind. L⁻¹) throughout the year.

Cheliff reservoir displayed marked spatial heterogeneity, with ST3 emerging as the most productive and diverse station, followed by ST2. This pattern likely reflects localized nutrient inputs, higher organic loading, hydromorphological features promoting nutrient cycling, and the presence of riparian vegetation that provides refuge for zooplankton, particularly cladocerans, from fish predation. In contrast, Kerrada reservoir exhibited limited spatial variability except at ST7, where densities and diversity were slightly elevated. The central location of ST7 near a mid-reservoir island likely enhances hydrodynamic retention and promotes zooplankton aggregation.

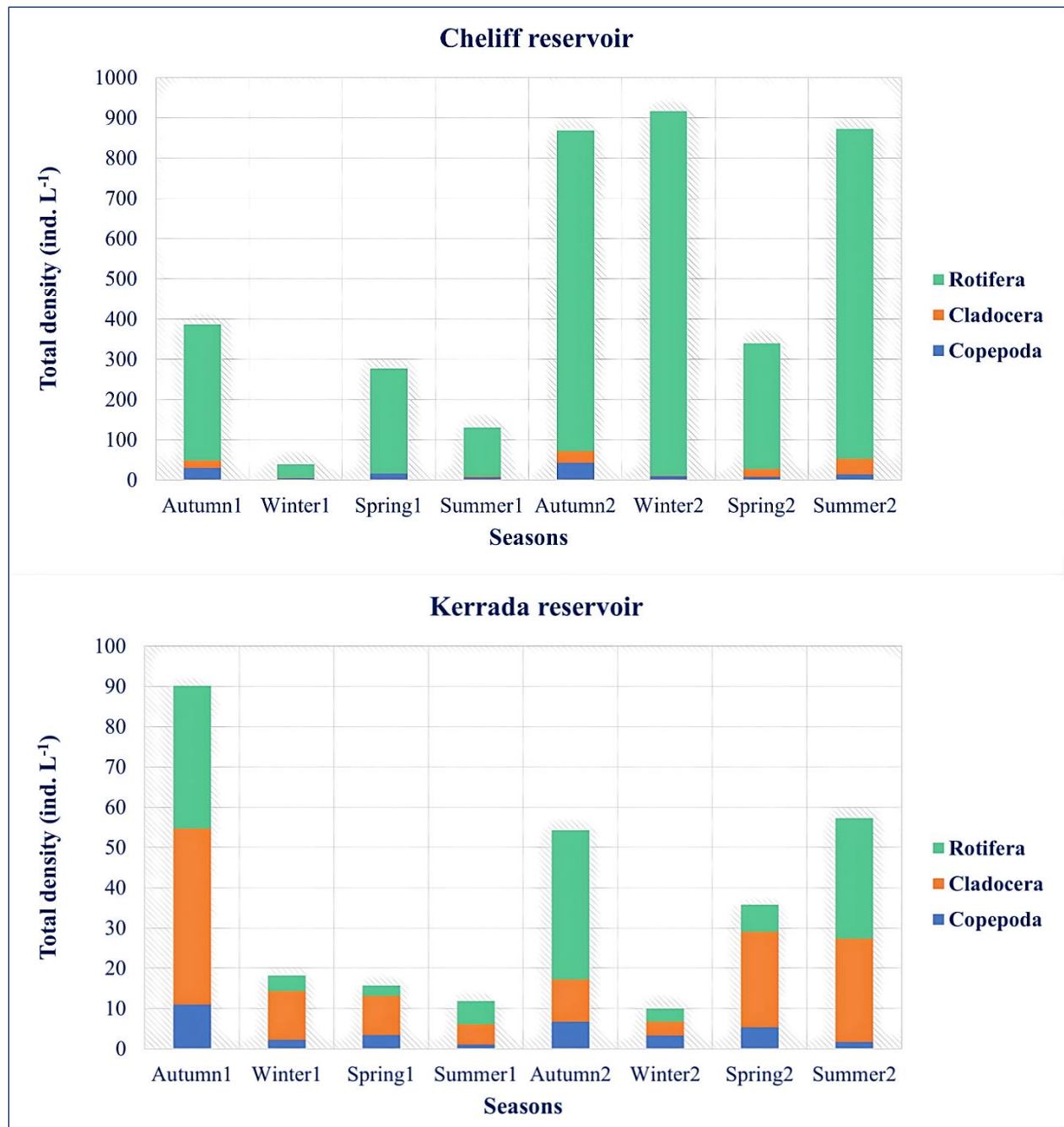
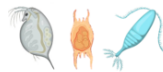
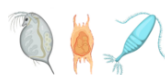


Fig. 52 Seasonal variation of zooplankton density of different groups in Cheliff and Kerrada reservoirs.

V.2.5. Environmental factors driving zooplankton

To explore the relationships between forward-selected environmental variables (blue vectors) and zooplankton species (red vectors), a redundancy analysis (RDA) was performed, including reservoir grouping (Fig. 53). This section was adapted from Baitiche et al. (2025). The first two axes explained 32.19% of the total variance in the dataset. Axis 1 accounted for 26.4% of the variation in zooplankton composition and represented the main environmental gradient separating the two reservoirs. Cheliff reservoir was associated with higher concentrations of PM,



turbidity, OM, NH_4^+ , TDS, NO_2^- , and salinity, whereas Kerrada reservoir was primarily influenced by NO_3^- and depth. Axis 2 explained an additional 5.79% of the variance and was positively correlated with temperature, and negatively correlated with DO (Fig. S12, Fig. S13, Table S6; Supplementary material).

Zooplankton species displayed distinct clustering within the environmental space defined by these axes. In Cheliff reservoir, *A. americanus* (including nauplii and copepodite stages) was strongly associated with NO_2^- , TDS, and salinity. *S. pectinata* and *Keratella cochlearis* f. *tecta* aligned with NH_4^+ , PM, and OM. *B. angularis* and *K. valga* occupied central positions, showing strong correlations with PO_4^{3-} , pH, and turbidity, reflecting their affinity for Cheliff's turbid, organic-rich waters. *B. calyciflorus* was similarly positioned but displayed a moderate association with turbidity, while *P. dolichoptera* was linked to higher temperatures. In Kerrada reservoir, *C. numidicus* (nauplii and juveniles) and *D. brachyurum* were associated with NO_3^- , whereas the rotifer *H. mira* was influenced by elevated temperature and PO_4^{3-} concentrations.

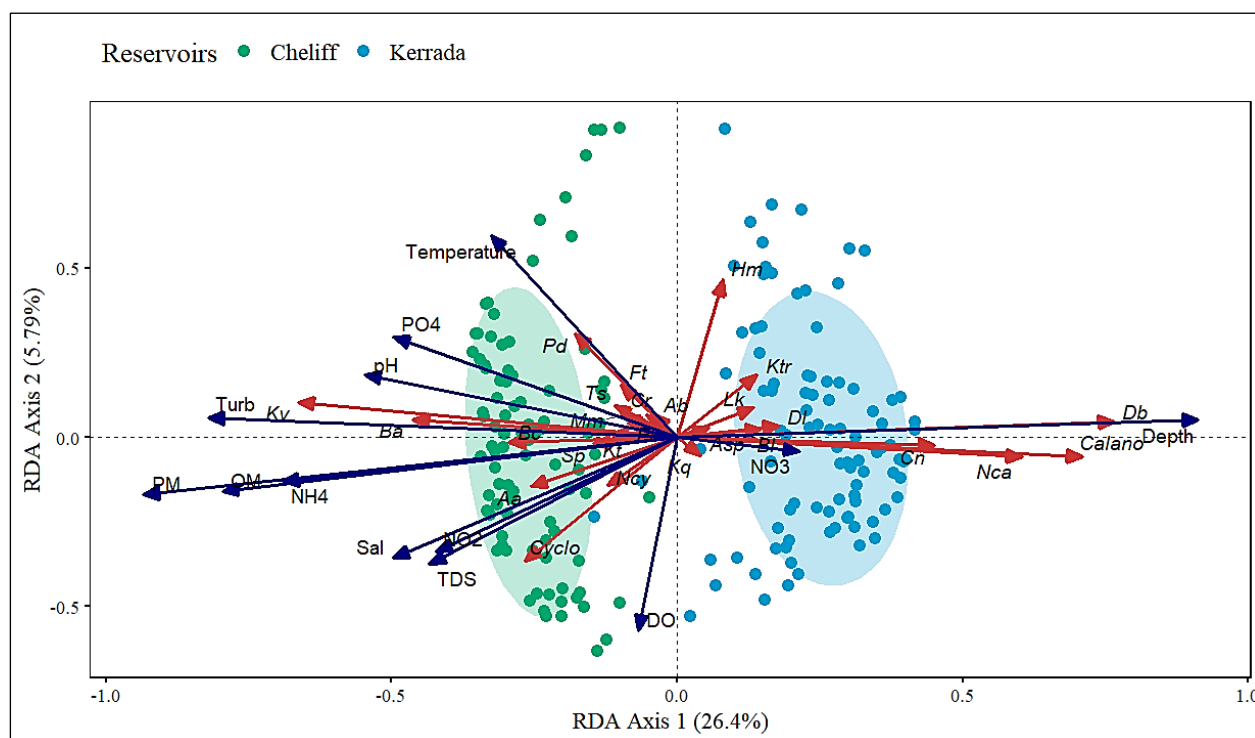
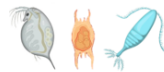


Fig. 53 Redundancy Analyses (RDA) based on the first two axes for dominant zooplankton species (red arrows) and environmental variables (blue arrows) of Cheliff and Kerrada reservoirs. Physicochemical parameters codes as indicated in (Chapter III, Table 3) and zooplankton species abbreviations as indicated in (Supplementary material, Table S5) (Baitiche et al., 2025).



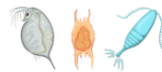
V.2.6. Phytoplankton and zooplankton assemblages

To assess the relative influence of phytoplankton groups and environmental variables on zooplankton composition, a partial redundancy analysis (RDA) was performed with samples grouped by reservoir (Fig. 54). The interpretation presented here focuses specifically on phytoplankton groups and their contribution to structuring zooplankton communities.

The first two RDA axes explained 39.94% of the total variance. Axis 1 accounted for 30.79% and represented the dominant ecological gradient separating Cheliff from Kerrada reservoirs, indicating that the combined effects of phytoplankton assemblages and environmental conditions significantly shape zooplankton community structure. Axis 1 was strongly influenced by PM and chlorophyll-a (each contributing >12%), both of which were positively associated with Cheliff reservoir (Supplementary Material, Fig. S14, Table S7). Among phytoplankton groups, Euglenophyceae contributed the most (>4%), followed by Chlorophyceae (~3%), and both were positively correlated with Cheliff reservoir. However, Axis 2 explained 9.15% of the variance and was positively associated with Dinophyceae and Chrysophyceae (~9%), Desmidiaceae (~5%), and Cyanophyceae (~5%) were positively correlated with Kerrada reservoir. This separation reflects reservoir-specific differences in environmental conditions and phytoplankton composition that ultimately influence zooplankton community patterns.

The two reservoirs supported distinct ecological associated plankton communities. Cheliff reservoir was characterized by higher concentrations of suspended solids, warmer temperatures, and increased nutrient levels. These conditions supported higher abundances of Chlorophyceae, Euglenophyceae, and Bacillariophyceae, and a zooplankton community dominated primarily by rotifers and Cyclopoida copepods, particularly *A. americanus*. In contrast, Kerrada reservoir exhibited greater depth and higher light transparency, conditions that favored cladocerans, especially *D. brachyurum*, as well as calanoid copepods *C. numidicus*. These taxa were associated with phytoplankton groups adapted to clearer, more stable water columns, including Dinophyceae, Chrysophyceae, Cryptophyceae, Desmids, and Cyanophyceae.

Zooplankton taxa displayed distinct clustering along these ecological gradients. *K. valga* and *B. angularis* were strongly associated with Euglenophyceae and the nutrient-rich, shallow waters of Cheliff reservoir. *P. dolichoptera*, *F. terminalis*, and several other rotifers were positively correlated with temperature and Chlorophyceae, indicating an affinity for warmer, productive conditions. In Cheliff, *A. americanus* showed strong positive correlations with NO_2^- , TDS, salinity,



and phytoplankton groups such as diatoms and Euglenophyceae. In Kerrada, cladocerans and *C. numidicus* were closely aligned with depth, transparency, and dissolved oxygen, while NO_3^- exhibited a significant, though shorter, gradient in relation to Cryptophyceae, Chrysophyceae, and depth. A strong correlation was observed between *Keratella tropica* and Chrysophyceae. Overall, the dispersion of zooplankton vectors in the RDA ordination reflects clear differentiation in ecological niches, structured by the interaction between water-column characteristics and phytoplankton community composition.

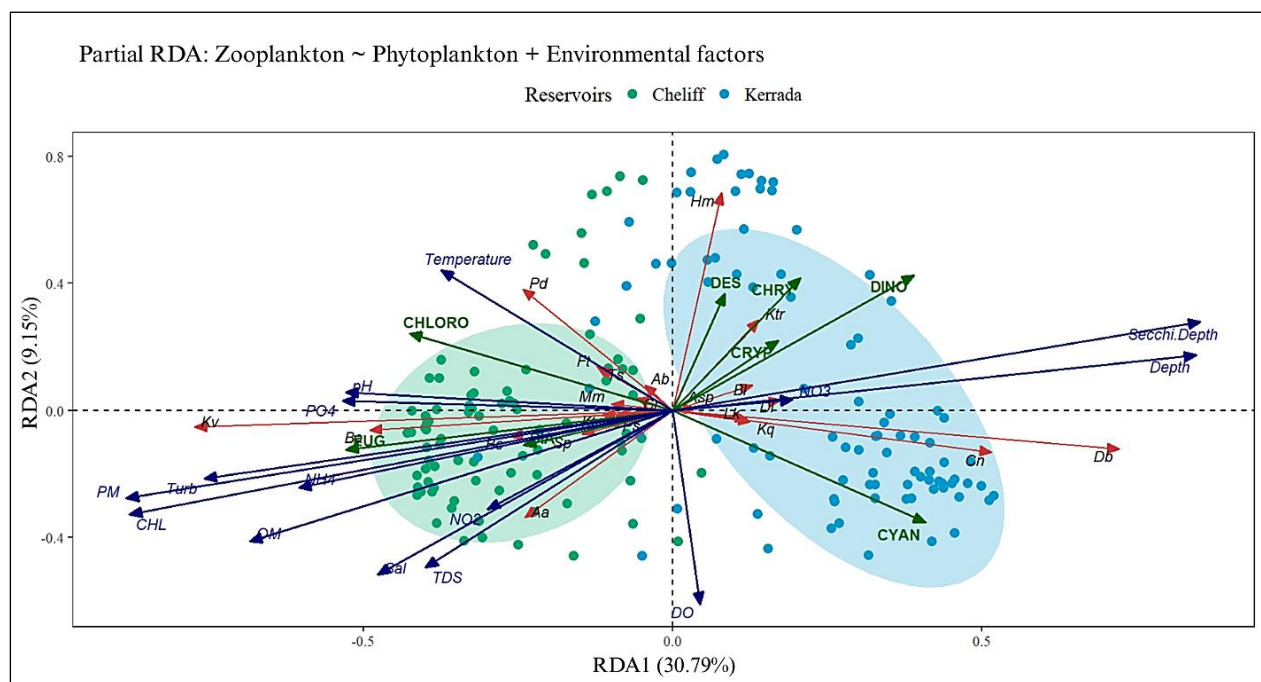
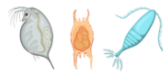


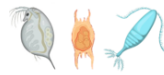
Fig. 54 Redundancy Analyses (RDA) based on the first two axes for phytoplankton groups (green arrows), zooplankton species (red arrows) and environmental variables (blue arrows) of Cheliff and Kerrada reservoirs. Physicochemical parameters codes as indicated in (Chapter III, Table 3) and phytoplankton and zooplankton groups abbreviations as indicated in (Supplementary material, Table S3; Table S5).



V.3. Discussion

Our findings highlight the central role of reservoir morphometry, mixing regimes, and nutrient availability in shaping zooplankton abundance, composition, and seasonal dynamics. Spatial–temporal variability in physicochemical parameters emerged as the principal driver of community structure, consistent with previous studies linking hydrodynamics and nutrient gradients to zooplankton assemblages (Ejsmont-Karabin, 1995; Armengol & Miracle, 1999; Whitman et al., 2004; Haberman & Haldna, 2014; García-Chicote et al., 2019; Liu et al., 2024; Orduna et al., 2024). The contrasting ecological conditions of Cheliff (shallow polymictic diversion reservoir), and Kerrada (deep more stable storage reservoir) further underscore the influence of morphometric and hydrological characteristics on community patterns. As artificial ecosystems, both displayed characteristics typical of reservoirs, including lower zooplankton diversity compared with natural lakes, largely determined by nutrient regimes and habitat structure (Simões et al., 2015). In Cheliff reservoir, higher temperatures were linked to its shallow depth and strong solar heating, unlike the deeper Kerrada reservoir (Magee & Wu, 2017; Boyd, 2020). Seasonal variations followed the patterns of arid and semi-arid climates (Mhamdi et al., 1994). Elevated summer pH likely reflected intense photosynthetic CO₂ uptake (Verspagen et al., 2014), while high spring DO concentrations indicated enhanced primary productivity (Alayat et al., 2013). Conversely, summer oxygen depletion suggests eutrophic conditions, intensified by high temperatures, microbial activity, and organic matter decomposition (Wetzel & Likens, 1991; Jane et al., 2023; Pilla et al., 2023).

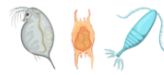
Cheliff reservoir consistently exhibited higher values of salinity, TDS, turbidity, PM, and OM, especially during the dry season, driven by reduced inflows and enhanced evaporation (Dojlido & Best, 1993; Yap, 2013; García-Chicote et al., 2019). High turbidity may also be linked to nutrient enrichment and sediment resuspension by benthic fish (Bidi-Akli et al., 2014), as well as sediment influx from the nearby river during runoff events (Allan & Castillo, 2007). In contrast, Kerrada's greater depth promotes sedimentation and results in lower turbidity and PM. Cheliff reservoir also presented higher concentrations of NO₂⁻, NH₄⁺, and PO₄³⁻ due to agricultural runoff and untreated wastewater inputs (Simões et al., 2015; Guermazi et al., 2023, Pinto et al., 2023). By comparison, Kerrada's elevated NO₃⁻ levels may be explained by higher nitrate retention in its deeper water column.



Zooplankton assemblages in the two reservoirs further reflect these environmental differences. Cheliff reservoir harbored greater species richness, likely driven by its nutrient availability and shallowness, which favor light penetration and productivity (Simões et al., 2015; de Souza Santos et al., 2022; Umi et al., 2024). However, Shannon diversity values (H') were comparable between the two reservoirs. Peaks in diversity during spring and summer suggest that increased productivity enhances food supply for zooplankton (Ochocka & Pasztaleniec, 2016; Sagouis et al., 2017; Pinto et al., 2023). Kerrada reservoir, however, displayed higher species evenness, indicating more balanced communities and greater ecological stability (Magurran, 2013; Roberts, 2019). In contrast, Cheliff reservoir was characterized by lower evenness, reflecting the dominance of opportunistic rotifers such as *S. pectinata*. Year-to-year variability, with higher richness and evenness observed in the second year, suggests climatic influences such as rainfall and nutrient input. Reservoir morphometry and the absence of predation pressure also play a role, with deeper reservoirs supporting more complex habitats and refuges (Hoffmann & Dodson, 2005; Hessen et al., 2006; Dodson et al., 2008; Cabral et al., 2020; de Souza Santos et al., 2022).

Zooplankton densities peaked in Cheliff reservoir during winter and summer 2022, reflecting nutrient enrichment and seasonal productivity typical of Mediterranean ecosystems (Moustaka-Gouni et al., 2014; Orduna et al., 2024). Kerrada reservoir maintained lower and more stable densities, consistent with reduced nutrient availability, a pattern characteristic of oligotrophic reservoirs (García-Chicote et al., 2018; Bonecker et al., 2020; Muñoz-Colmenares et al., 2021). Species distribution was closely tied to environmental gradients. In Cheliff reservoir, rotifers such as *K. valga*, *S. pectinata*, *B. angularis*, *P. dolichoptera*, and *B. calyciflorus* dominated, all linked to eutrophic, turbid, and nutrient-rich conditions (Sládeček, 1983; Picapedra et al., 2019; Liang et al., 2020; Akli-Bidi et al., 2023). Specific patterns supported these interpretations. The sudden proliferation of *S. pectinata* in Winter was associated with nutrient enrichment following mixing and runoff events. The autumn peak of *B. angularis* coincided with low water levels and concentrated organic resources. The vertical distribution of *K. valga* during spring, with mid-depth maxima, reflects feeding strategies and avoidance of surface turbulence, typical of polymictic ecosystems (Dokulil & Teubner, 2000; Umi et al., 2018; Chakrabarti, 2023).

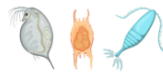
RDA analysis confirmed these associations, with *S. pectinata* peaking in winter under high NH_4^+ and PM, while *P. dolichoptera* was linked to higher temperatures and phosphate availability (Yin et al., 2018; Smaoune et al., 2021; Orduna et al., 2024). In Kerrada reservoir, *H. mira*



dominated under oxygen-rich, low-mineral conditions (Afonina & Tashlykova, 2018), while *K. quadrata* and *P. dolichoptera* highlighted adaptability to varying conditions.

Cladoceran patterns further distinguished the reservoirs. *D. brachyurum*, an indicator of moderate water quality, dominated in Kerrada reservoir (Stamou et al., 2021), while *B. longirostris* and *M. micrura*, typically linked to warm eutrophic ecosystems, were more abundant in Cheliff reservoir (Alonso, 1996; Bouzidi et al., 2010). *D. longispina* in Kerrada reservoir reflected better water quality, whereas *C. reticulata* and *D. magna* occurred exclusively in Cheliff reservoir, thriving in nutrient-rich environments (Almeida et al., 2020; Pinto et al., 2021b). *L. kindtii*, found only in Kerrada reservoir, was associated with its well-oxygenated deep waters (Bowersox et al., 2014). The higher proportion of cladocerans in Kerrada, particularly in spring and winter, suggests lower predation pressure and higher resource availability, while their scarcity in Cheliff reservoir likely reflects elevated salinity (Wagaw et al., 2023). In Cheliff reservoir, Cladoceran density peaks aligned with moderate temperatures, increased phytoplankton availability, and reduced fish predation in late summer and autumn (Bidi-Akli et al., 2014; Sehili et al., 2020; Pinto et al., 2021b; Stamou et al., 2021). The reappearance of *D. magna* further indicates that predation cycles and temperature thresholds regulate large-bodied cladocerans in shallow, nutrient-rich reservoirs (Wagner et al., 2005; Bidi-Akli et al., 2014; Almeida et al., 2020).

Copepods, although less abundant, provided valuable ecological insights. *A. americanus* dominated Cheliff reservoir, strongly correlated with NO_2^- , TDS, and salinity, whereas *C. numidicus* prevailed in Kerrada reservoir, associated with nitrate-rich, oxygenated waters. These results are consistent with their trophic strategies: cyclopoids dominate shallow eutrophic reservoirs (García-Chicote et al., 2019; Montagud et al., 2019), while calanoids thrive in deeper, meso-eutrophic ecosystems (Pinto-Coelho et al., 2005; Parra et al., 2009). The developmental stages of *A. americanus* (nauplii and copepodites) also showed correlations with nutrient-rich conditions, reflecting their omnivorous feeding strategies (Sarma & Nandini, 2019). Meanwhile, *C. numidicus* favored phytoplankton-rich, oxygenated habitats, with reservoir depth providing refuge from predation (Błędzki & Rybak, 2016; Min et al., 2021). The seasonal dynamics of *A. americanus*, autumn population growth, disappearance of adults in winter, and spring nauplii peaks, match reproductive cycles modulated by temperature (Sarma et al., 2019; Alekseev, 2021). *E. serrulatus* appeared intermittently at low densities, likely maintained in littoral refugia (Boxshall & Defaye, 2008).

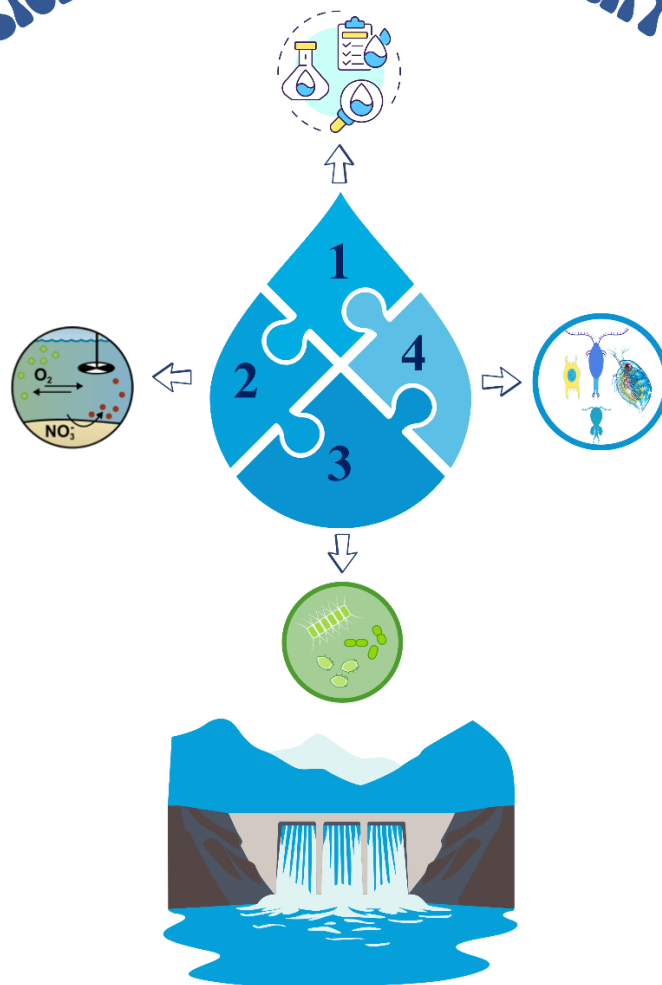


The partial RDA results further clarify how phytoplankton assemblages, in conjunction with environmental gradients, shape zooplankton community structure in the two reservoirs. The dominant axis separating Cheliff from Kerrada reflected higher particulate matter, chlorophyll-a, and the prevalence of Euglenophyceae and Chlorophyceae in Cheliff reservoir, conditions that favored rotifers and cyclopoids such as *A. americanus*. These patterns reflect shallow morphometry, sediment resuspension, and nutrient loading, which accelerate eutrophication (Moss et al., 2011; Simões et al., 2015; García-Chicote et al., 2019). In contrast, Kerrada's clearer, deeper, and more oxygenated waters were associated with Dinophyceae, Chrysophyceae, desmids, and cyanobacteria. These phytoplankton communities supported cladocerans, particularly *D. brachyurum*, and the calanoid *C. numidicus*, taxa known to thrive in meso-eutrophic to oligotrophic ecosystems characterized by higher transparency and reduced disturbance (Pinto-Coelho et al., 2005; Parra et al., 2009; Stamou et al., 2021). The distinct clustering of taxa along these gradients highlights reservoir-specific environmental regimes, where phytoplankton composition, light conditions, and nutrient status jointly determine zooplankton distribution and functional structure (Ejsmont-Karabin, 1995; Armengol & Miracle, 1999; Whitman et al., 2004).

Overall, the study demonstrates the combined effects of reservoir morphometry and water quality in shaping zooplankton biodiversity and dynamics. Key taxonomic groups were strongly associated with nutrient and oxygen gradients, underscoring their value as bioindicators under anthropogenic pressures (Zhang et al., 2019; Umi et al., 2020). Moreover, the interplay between morphometry and trophic status is evident: deeper reservoirs mitigate eutrophication by storing nutrients in deeper layers, while shallow reservoirs like Cheliff reservoir are more prone to nutrient accumulation and eutrophic conditions (Ejsmont-Karabin, 1995; Armengol & Miracle, 1999; Whitman et al., 2004; García-Chicote et al., 2019). These insights provide a basis for further studies evaluating water quality through zooplankton community dynamics.

Chapter VI: Plankton-based assessment of reservoir water quality

INSIGHTS FROM A DROP OF WATER





This chapter evaluates the ecological potential and trophic status of the two study reservoirs through the application of multiple environmental indices and plankton-based metrics. Building upon the physicochemical, phytoplankton, and zooplankton frameworks established in the preceding chapters, this analysis integrates top-down and bottom-up indicators to provide a more holistic and robust assessment of water quality and ecosystem functioning. Specifically, we apply trophic state Carleson indices (TSIs), and both phytoplankton and zooplankton metrics to quantify and compare the ecological status of Cheliff and Kerrada reservoirs. A preliminary overview of trophic state patterns was introduced in previous chapters; here, we expand this assessment through a comprehensive examination of how each index reflects environmental conditions across contrasting reservoir types. Particular attention is given to evaluating the sensitivity, complementarity, and limitations of each index when applied to Mediterranean freshwater ecosystems subject to both anthropogenic pressures and climatic fluctuations. By synthesizing insights from chemical, biological, and functional indicators, this chapter provides the most integrative evaluation of ecosystem functioning in these reservoirs. The findings offer valuable implications for water quality management, ecological monitoring, and the development of robust assessment frameworks for Mediterranean reservoirs.

VI. 1. Material and methods

VI. 1. 1. Sampling procedure

Sampling procedures for environmental factors, phytoplankton, and zooplankton in Cheliff and Kerrada reservoirs were previously described in Chapters III, IV, and V, respectively. In the present section, we focus on water quality assessment using data collected from the four sampling sites at different depth layers in each reservoir. Indices were calculated using average values of physicochemical, phytoplankton, and zooplankton datasets. Results are presented by season, as well as through overall averages across the two-year study period, in order to provide an integrated evaluation of water quality in both reservoirs.

For environmental parameters, total phosphorus (TP) was selected in this chapter instead of orthophosphate (PO_4^{3-}), which was used in the previous analyses. Orthophosphate was previously chosen because it represents the biologically assimilable form directly taken up by phytoplankton and zooplankton; however, total phosphorus is more appropriate for water quality indices, as it reflects the full nutrient pool and is commonly used to assess trophic status and nutrient cycling. Total phosphorus, representing the sum of dissolved and particulate phosphorus



forms in the water column, was quantified according to Rodier (2009) using oxidative digestion (persulfate method) followed by colorimetric determination of orthophosphate with spectrophotometric measurement.

VI. 1. 2. Statistical analyses

VI.1.2.1. Carlson's Trophic State Indices (TSI)

One of the first and most widely applied trophic classification systems is Carlson's Trophic State Index (TSI), which uses Secchi depth (SD), total phosphorus (TP), and chlorophyll-a (CHL) as proxies for phytoplankton biomass (Carlson, 1977) (Table S1, Supplementary material). Although effective, this approach mainly reflects chemical drivers and phytoplankton standing stock.

Formulas of Carlson's indices (1977):

- $TSI(CHL) = 9.81 \times \ln(CHL) + 30.6$;
- $TSI(SD) = 60 - 14.41 \times \ln(SD)$;
- $TSI(TP) = 14.42 \times \ln(TP) + 4.15$.

VI.1.2.2. Phytoplankton-based indices

In this study, we applied Lacustrine Phytoplankton Index (IPLAC) (Feret & Laplace-Tretyure, 2013; Laplace-Tretyure & Feret, 2016), which integrates taxonomic composition (biovolume) and algal biomass (chlorophyll-a). The index was calculated using the PHYTOBS v2.3 tool (Hadoux et al., 2015). Trophic status classes are presented in (Table 4).

Additionally, Phytoplankton Index (IPL) developed by Barbe et al. (2003) was applied. This index incorporates the presence and relative abundance of algal groups:

$$IPL = \sum (Q_i * A_j)$$

Q_i is the coefficient assigned to each algal group, and A_j represents the relative abundance class of each algal group (Table S8 and Table S9, Supplementary material).



Table 4 Trophic classes according to the IPL (Barbe et al., 2003), and IPLAC (Feret & Laplace-Treytore, 2013; Laplace-Treytore & Feret, 2016) indices.

IPL value	< 25	25 < IPL < 40	40 < IPL < 60	60 < IPL < 80	>80
IPLAC value	1 – 0.8	0.6	0.4	0.2	0
Water quality	High	Good	Moderate	Poor	Bad
Trophic class	Oligotrophic	Mesotrophic		Eutrophic	

VI.1.2.3. Zooplankton-based indices

The most commonly applied zooplankton-based indices are those developed by Ejsmont-Karabin (2012). These include: Trophic State Index based on rotifer abundance (TSI_{NROT}), percentage of bacterivorous species (*Anuraeopsis fissa*, *Filinia* spp., *Brachionus angularis*, *Keratella cochlearis*, *Pompholyx sulcata*) in the total numbers of Rotifera (BAC, %) TSI_{BAC} , and percentage of form *tecta* in the population of *Keratella cochlearis* morphotype in the population (TECTA, %) TSI_{Ptecta} .

Additional indices based on microcrustaceans were developed by Ejsmont-Karabin & Karabin (2013), such as the Trophic Index based on crustacean abundance (N, ind. L⁻¹) (TSI_{CR}), and ratio of cyclopoid-to-calanoïd copepods numbers (CY/CA) ($TSI_{CY/CA}$). These indices are widely used for assessing water quality in monomictic, dimictic and polymictic lakes and reservoirs:

- $TSI_{NROT} = 5.38 \times \ln(N_{ROT}) + 19.28$;
- $TSI_{BAC} = 0.23 \times BAC + 44.30$;
- $TSI_{Ptecta} = 0.187 \times TECTA + 50.38$;
- $TSI_{CR} = 25.5 N^{0.142}$;
- $TSI_{CY/CA} = 5.08 \times \ln(CY/CA) + 46.6$.

All trophic state indices were calculated in Excel for both reservoirs.



VI. 2. Results

VI.2.1. Assessment of reservoir trophic Status

VI.2.1.1. Carlson's Trophic State Indices (TSI)

According to the mean values of chlorophyll-a (Chl-a; Cheliff = $47.2 \mu\text{g L}^{-1}$; Kerrada = $7.1 \mu\text{g L}^{-1}$), Secchi depth (SD; Cheliff = 0.4 m; Kerrada = 2.0 m), and total phosphorus (TP; Cheliff = $128.1 \mu\text{g L}^{-1}$; Kerrada = $22.0 \mu\text{g L}^{-1}$), the annual average values of Carlson's Trophic State Index (TSI) were calculated (Table 5).

For Cheliff reservoir, the calculated TSI corresponding to a **eutrophic to hypereutrophic state**. This trophic status reflects very high productivity, linked to elevated nutrient concentrations and dense phytoplankton biomass, which are typical indicators of eutrophication (Carlson, 1977; OECD, 1982) (Table S1, Supplementary material). Such conditions are often associated with oxygen depletion, algal blooms, and reduced water quality (Smith, 2003).

In contrast, Kerrada reservoir exhibited TSI values corresponding to a **mesotrophic state** (Carlson, 1977). This indicates moderate productivity, characterized by average concentrations of nutrients and phytoplankton biomass. Mesotrophic ecosystems are generally more balanced ecologically, but they remain vulnerable to nutrient enrichment, which may lead to progressive eutrophication if external nutrient inputs increase (Carlson, 1977; Wetzel, 2001).

These results are consistent with the physical, chemical, and biological observations at both sites: Cheliff reservoir is subject to poor water quality and high eutrophication risk, while Kerrada reservoir maintains moderate water quality with mesotrophic characteristics.

Table 5 Carlson's Trophic State Index (TSI) values for Cheliff and Kerrada reservoirs during the study period.

Cheliff Reservoir			Kerrada Reservoir		
TSI _{CHL}	TSI _{SD}	TSI _{TP}	TSI _{CHL}	TSI _{SD}	TSI _{TP}
68.4	73.2	74.1	49.8	50	48.7



VI.2.1.2. Phytoplankton-based indices

Phytoplankton composition revealed Cyanophyceae dominance in both reservoirs (Cheliff: 34%; Kerrada: 36%), followed by Chlorophyceae (32% and 34%, respectively). In Cheliff reservoir, Euglenophyceae (19%) and Diatomophyceae (11%) contributed significantly, whereas in Kerrada reservoir other groups represented <10% (Fig. 55a).

The strong prevalence of Cyanophyceae, indicative of eutrophic conditions and poor ecological quality (Dokulil & Teubner, 2000; Reynolds, 2006; Bidi-Akli et al., 2017; Laplace-Tretyure & Feret, 2016) suggests eutrophic to hypereutrophic status in Cheliff reservoir and meso- to eutrophic status in Kerrada reservoir. The considerable proportion of Euglenophyceae in Cheliff reservoir further reflects organic enrichment and oxygen fluctuations, conditions that favor this group (Wołowski, 2011; Kim et al., 2016; Arab et al., 2019). In Kerrada reservoir, occurrence of loricate *Trachelomonas* spp. likely reflects the influence of its deeper, stratified water column and lower organic load, which favor mixotrophic loricate euglenoids adapted to mesotrophic ecosystems (Wołowski & Grabowska, 2007; Coesel & Krienitz, 2008; Wang H. et al., 2024). *Phacus* spp. typically associated with shallow, nutrient-rich, and well-lit environments (Wołowski & Hindák, 2005; Ghosh & Keshri, 2010; Afaf et al., 2022), were absent from Kerrada reservoir but abundant in Cheliff reservoir, consistent with its shallow morphometry and stronger sediment–water interactions. The presence of Chlorophyceae and diatom taxa often linked to balanced and oxygenated environments (Salmaso, 2000; Reynolds, 2006; Padisák et al., 2009; Djabourabi, 2017; Sendra et al., 2019; Benmatouk et al., 2024), highlights pronounced seasonal variability and environmental heterogeneity.

Application of IPL and IPLAC indices confirmed these patterns (Fig. 55b): Cheliff reservoir exhibited high IPL and low IPLAC values, classifying it as eutrophic–hypereutrophic, whereas Kerrada’s lower IPL and higher IPLAC indicated mesotrophy and moderate water quality (Barbe et al., 2003; Feret & Laplace-Tretyure, 2013; Laplace-Tretyure & Feret, 2016).

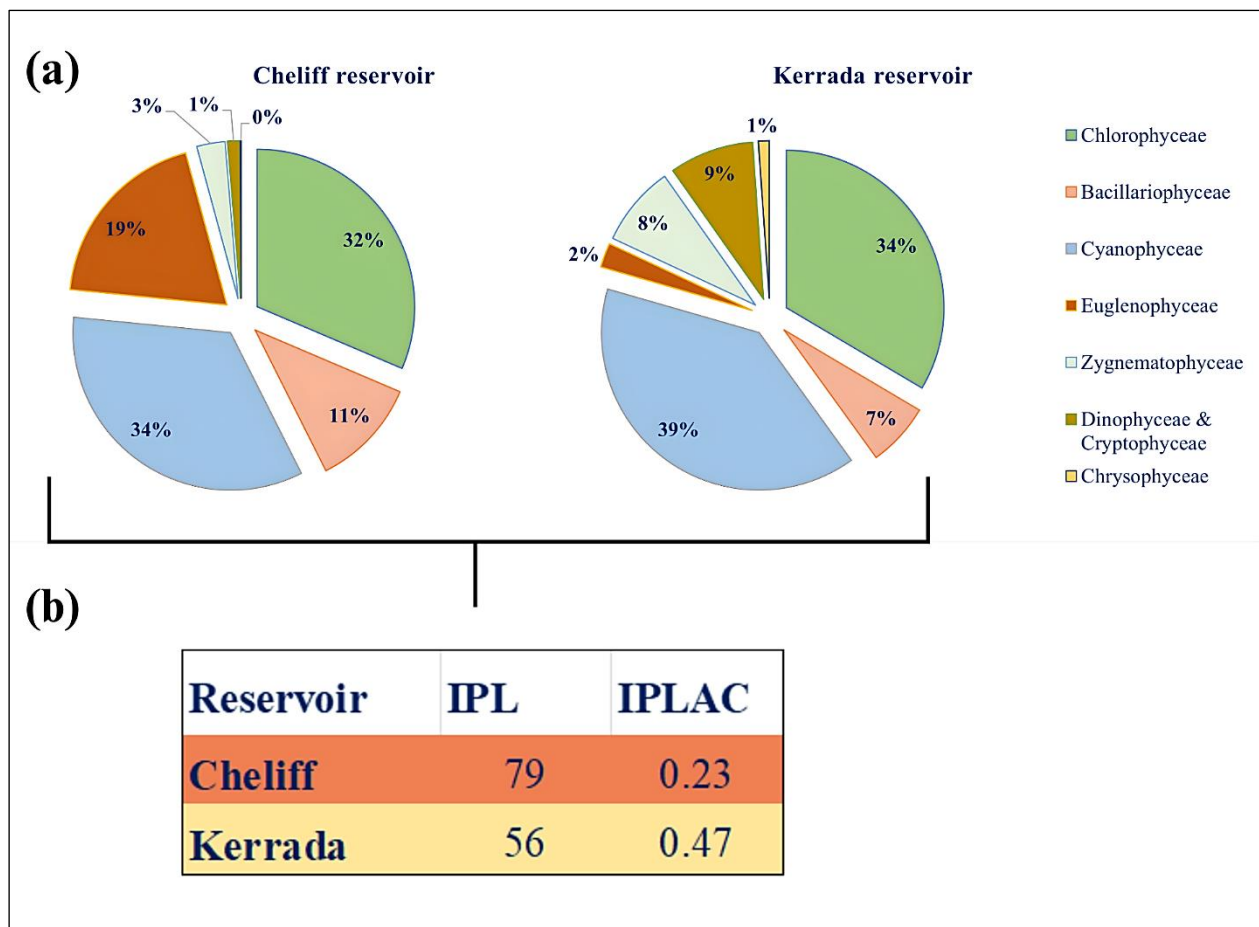


Fig. 55 Trophic state of Cheliff and Kerrada reservoirs based on (a) the relative abundance of algal groups and (b) the IPL and IPLAC indices.

VI.2.1.3. Zooplankton-based indices

Zooplankton community composition revealed clear differences between Cheliff and Kerrada reservoirs, reflecting their trophic states. Cheliff reservoir was strongly dominated by Rotifera (94%), a pattern typical of eutrophic to hypereutrophic ecosystems, as rotifers are opportunistic taxa able to rapidly exploit nutrient-enriched environments (Ejsmont-Karabin, 2012; Umi et al., 2018; García-Chicote et al., 2019; Picapedra et al., 2019; Liang et al., 2020; Muñoz-Colmenares et al., 2021; Akli-Bidi et al., 2023). In contrast, Kerrada reservoir displayed a more balanced structure, with Cladocera (46%) dominating, followed by Rotifera (43%) and Copepoda (11%) (Fig. 56a). The prevalence of cladocerans in Kerrada reservoir is consistent with mesotrophic conditions, as many species prefer deeper, more stable environments (Muñoz-Colmenares et al., 2021; Stamou et al., 2021; Baitiche et al., 2025).

Copepods were less abundant but informative: *A. americanus* dominated Cheliff reservoir, whereas *C. numidicus* was prevalent in Kerrada reservoir. Cyclopoids are generally



associated with shallow eutrophic Mediterranean ecosystems (García-Chicote et al., 2019; Montagud et al., 2019; Muñoz-Colmenares et al., 2021; Pinto et al., 2023; Baitiche et al., 2025), whereas calanoids, primarily filter feeders are linked to clearer, meso-eutrophic environments (Pinto-Coelho et al., 2005; Parra et al., 2009; Akli-Bidi et al., 2023).

These compositional differences are further corroborated by zooplankton-based trophic state indices (Fig. 56b). Cheliff reservoir exhibited higher values of TSI_{Ptecta} and $TSI_{CY/CA}$, together with elevated TSI_{NROT} , all indicative of eutrophic to hypereutrophic conditions (Ejsmont-Karabin, 2012). However, TSI_{BAC} index based on the proportion of bacterivorous rotifers, positioned Cheliff closer to mesotrophy. This reflects the fact that bacterivorous taxa respond primarily to bacterial production rather than phytoplankton biomass, and thus may underestimate eutrophication in shallow polymictic reservoirs where high turbidity or resuspension limits bacterial growth despite high nutrient loads (Ejsmont-Karabin, 2012; Ochocka, 2021). Similarly, TSI_{CR} , based on crustacean abundance, suggested mesotrophy in Cheliff reservoir because crustacean populations are often suppressed in eutrophic, turbid, and fish-dominated shallow ecosystems, where predation pressure and habitat instability reduce their densities independently of nutrient concentrations (Jeppesen et al., 2011; Ejsmont-Karabin & Karabin, 2013; Haberman & Haldna, 2014; Karpowicz et al., 2025).

By contrast, Kerrada reservoir showed TSI_{NBAC} and $TSI_{CY/CA}$ values indicative of mesotrophic conditions. In addition, TSI_{Ptecta} classified the reservoir as eutrophic, reflecting the higher proportion of *Keratella cochlearis* f. *tecta*, a morphotype associated with elevated nutrient levels (Ejsmont-Karabin, 2012). However, low TSI_{NROT} and TSI_{CR} values placed Kerrada in oligotrophic to mesotrophic range. These patterns are consistent with the deeper, stratified nature of the reservoir, where zooplankton densities tend to remain relatively low despite moderate nutrient concentrations, due to habitat stability and seasonal stratification that limit overall zooplankton abundance (Haberman & Haldna, 2014; Karpowicz et al., 2025).

The divergence among individual zooplankton indices therefore does not indicate unreliability but rather highlights the sensitivity of zooplankton communities to ecological drivers beyond nutrient enrichment such as water column depth, predation pressure, thermal stratification, and trophic efficiency. When considered collectively, the microcrustacean and rotifer-based indices sharply differentiate the two reservoirs, confirming eutrophic to hypereutrophic conditions in Cheliff reservoir versus mesotrophic conditions in Kerrada reservoir. This underscores the value



of zooplankton metrics as complementary biological indicators that capture functional dimensions of trophic state not reflected by classical chemical or phytoplankton-based indices.

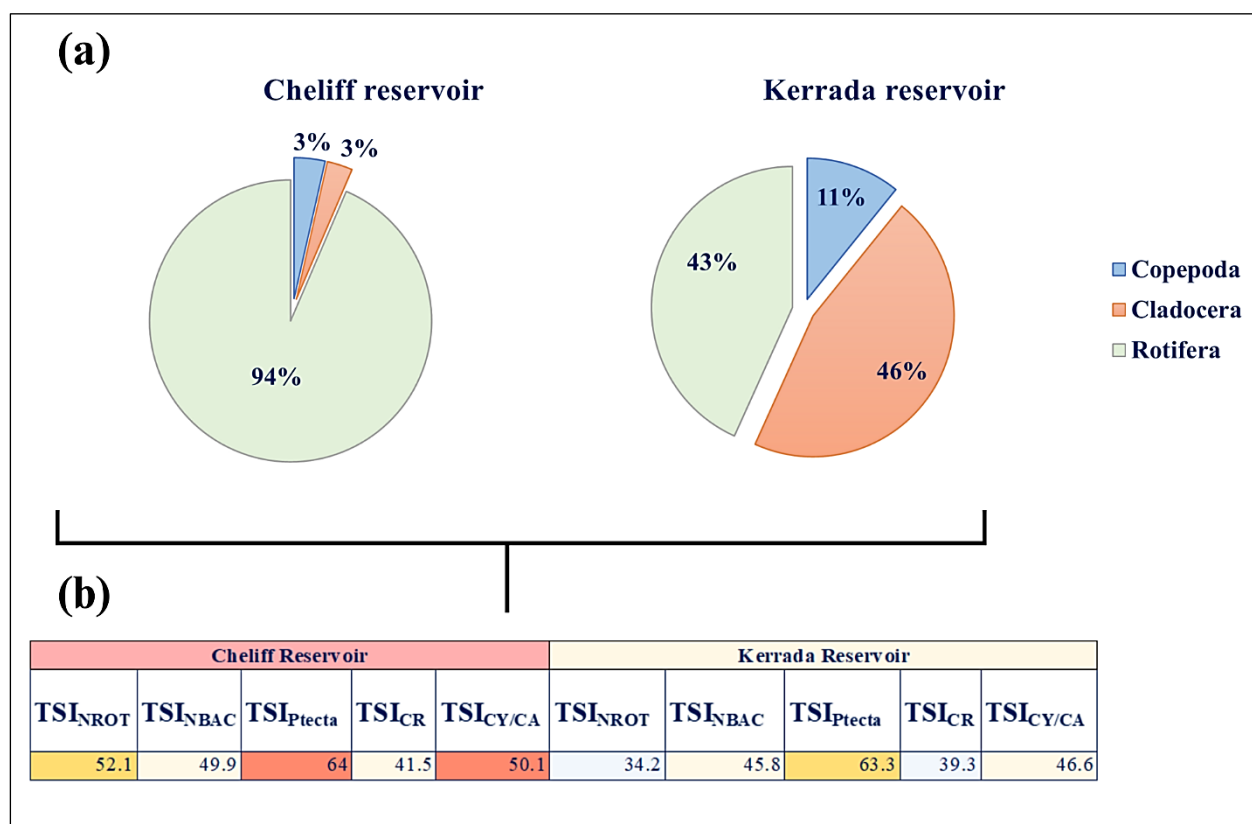


Fig. 56 Trophic state of Cheliff and Kerrada reservoirs based on (a) the relative abundance of zooplankton groups and (b) zooplankton-based trophic state indices.

The projection of Carlson TSI indices and zooplankton-based indices within a single plot provides valuable integrated information (Fig. 57). The figure shows that Carlson TSI values align strongly with nutrient-driven trophic state (eutrophic–hypereutrophic in Cheliff reservoir; mesotrophic in Kerrada reservoir), whereas zooplankton indices capture additional ecological processes such as predation pressure, habitat stability, and community efficiency, resulting in a broader range of trophic classifications. Rather than indicating inconsistency, this divergence highlights the complementary value of zooplankton-based indices for interpreting ecosystem functioning beyond nutrient enrichment alone. The convergence of both methods underscores the reliability of zooplankton as indicators of reservoir water quality.

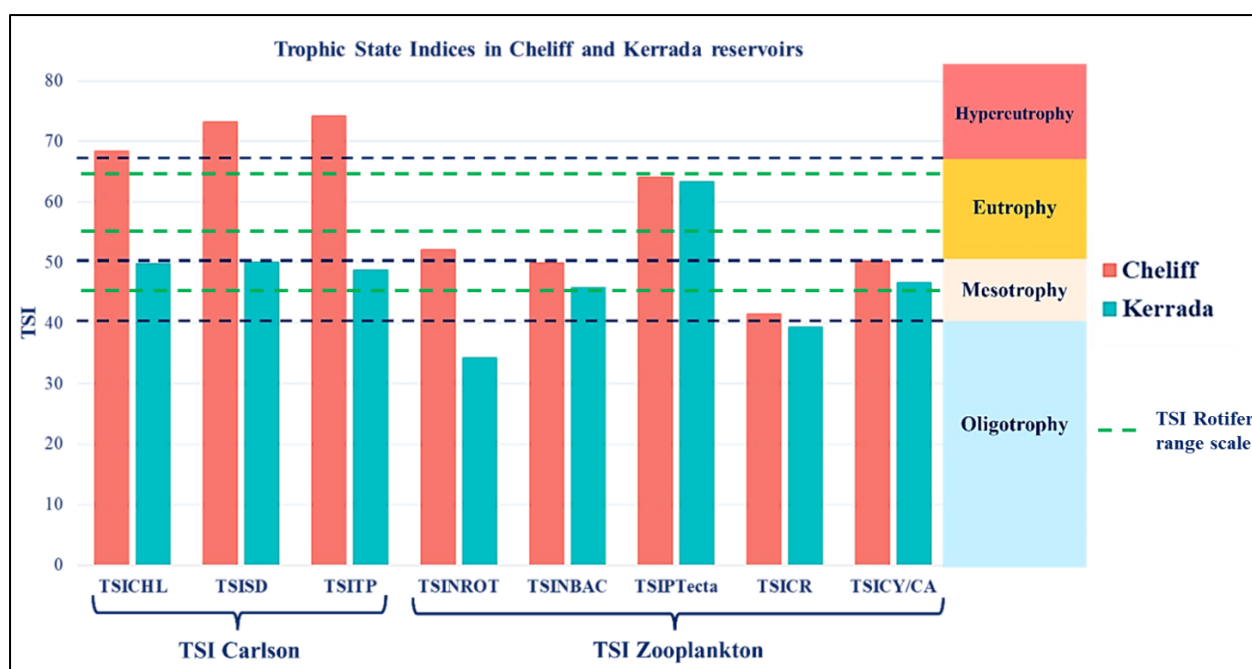


Fig. 57 Cheliff and Kerrada reservoir water quality by Carlson's and zooplankton Trophic State Indices.

VI.2.2. Bloom events and their significance in reservoir eutrophication

The phenomenon of algal blooms and eutrophication was markedly more pronounced in Cheliff reservoir than in Kerrada reservoir, with multiple harmful algal blooms (HABs) recorded throughout the two-year monitoring period. Phytoplankton density data were presented in Chapter IV; here, these bloom events are interpreted in relation to water quality and trophic status. The most intense bloom occurred in Spring 1 and was dominated by *Euglenaria caudata* (5.5×10^6 ind. L^{-1}), followed by *Euglena oblonga* (1.6×10^6 ind. L^{-1}) and *Lepocinclis acus* (1.2×10^6 ind. L^{-1}), indicating severe organic enrichment and elevated nutrient availability typical of highly eutrophic ecosystems. Recurrent blooms of the cyanobacterium *Microcystis aeruginosa* were also observed, particularly during spring, reaching a maximum of 9.1×10^5 ind. L^{-1} (Fig. 58). Its proliferation frequently co-occurred with high abundances of *Pseudanabaena catenata* (4.5×10^5 ind. L^{-1}) and *Pseudanabaena mucicola* ($\sim 2 \times 10^5$ ind. L^{-1}), species commonly associated with cyanobacterial dominance under nutrient-rich, warm, and stable environmental conditions. Additional cyanobacterial blooms included *Aphanocapsa elachista* (3.6×10^5 ind. L^{-1}), *Planktolyngbya limnetica* (2.5×10^5 ind. L^{-1}) in Autumn 2, and *Planktothrix agardhii* (2×10^5 ind. L^{-1}) in Summer 1. Among Chlorophyceae, *Chlorella vulgaris* reached 4.4×10^6 ind. L^{-1} in Winter 1, reflecting its broad ecological tolerance and ability to proliferate under eutrophic conditions even at low temperatures. These recurrent, multispecies blooms collectively demonstrate the advanced eutrophic to hypereutrophic status of the Cheliff reservoir.



In contrast, Kerrada reservoir exhibited substantially lower phytoplankton densities, consistent with mesotrophic to slightly eutrophic conditions and reduced bloom intensity. The highest recorded densities included *Microcystis aeruginosa* (1.5×10^5 ind. L^{-1}) in Winter 2, *Planctonema lauterbaunnii* and *Limnothrix redekei* (both $\sim 10^5$ ind. L^{-1}), and *Parietochloris pseudoalveolaris* ($\sim 10^5$ ind. L^{-1}) in Autumn 1. Minor proliferations of *Aphanocapsa elachista*, *Dolichospermum flos-aquae*, *Pseudanabaena mucicola*, and *Ceratium hirundinella* were also detected, though all remained below 7×10^5 ind. L^{-1} .

The lower intensity and reduced diversity of harmful cyanobacterial blooms in Kerrada reservoir compared to Cheliff reservoir suggest better water quality, lower nutrient loading, and more stable ecological conditions. Overall, the dominance and recurrence of bloom-forming taxa particularly *Microcystis*, *Euglena*, and *Pseudanabaena* in Cheliff reservoir reflect strong eutrophication processes, whereas Kerrada reservoir remains less affected, illustrating the influence of hydrology, trophic status, and anthropogenic pressures on bloom dynamics in the two ecosystems.

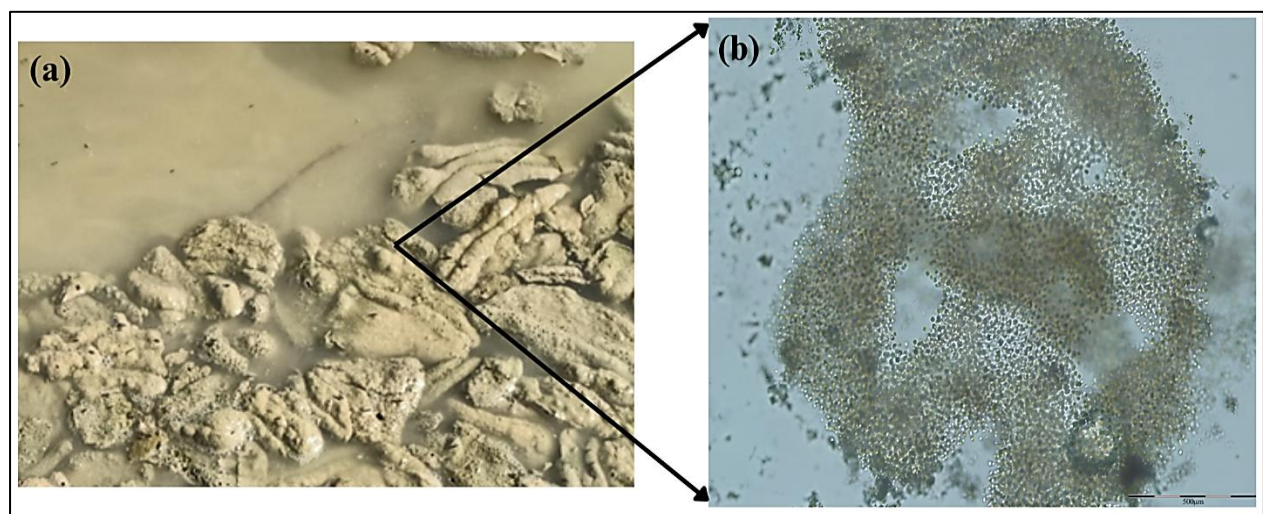


Fig. 58 *Microcystis aeruginosa* bloom: (a) macroscopic view; (b) microscopic view.



VI.3. Discussion

Our findings emphasize the central role of morphometry, hydrodynamics, and nutrient enrichment in shaping the trophic status and plankton community structure of the studied reservoirs. Cheliff reservoir, a shallow diversion ecosystem, and Kerrada reservoir, a deeper storage ecosystem, exhibited contrasting ecological conditions largely determined by nutrient availability, transparency, and depth. These results are consistent with earlier studies showing that hydrodynamics and nutrient gradients strongly influence plankton assemblages and trophic state (Ejsmont-Karabin, 1995; Armengol & Miracle, 1999; Haberman & Haldna, 2014; García-Chicote et al., 2019; Liu et al., 2024; Orduna et al., 2024).

Carlson's Trophic State Index (TSI) clearly differentiated between the two reservoirs. Cheliff reservoir, with high chlorophyll-a, high total phosphorus, and reduced transparency, was classified as eutrophic to hypereutrophic, reflecting intense algal productivity and bloom events typical of shallow, nutrient-enriched systems (Carlson, 1977; OECD, 1982; Smith, 2003). In contrast, Kerrada reservoir exhibited mesotrophic conditions, with moderate nutrient levels and higher water transparency. Similar depth-related contrasts have been reported in Mediterranean reservoirs, where morphometry mitigates eutrophication by promoting nutrient storage in deeper layers (Jeppesen et al., 2005; Søndergaard et al., 2017).

Phytoplankton composition and associated indices provided strong support for these trophic classifications. Cheliff reservoir was dominated by Cyanophyceae and Euglenophyceae, groups strongly associated with eutrophic conditions (Kim et al., 2016; Laplace-Treyture & Feret, 2016; Arab et al., 2019). Recurrent occurrences of *Microcystis aeruginosa*, are well-known eutrophication indicators due to their ability to form harmful blooms and release toxins (Dokulil & Teubner, 2000; Reynolds, 2006; Paerl & Otten, 2013; Wang L. et al., 2024). The presence of nitrogen-fixing taxa such as *Aphanocapsa elachista* suggests nitrogen limitation during stratified periods, favoring diazotrophic cyanobacteria (Paerl, 2017; Bhardwaj et al., 2024). The abundance of Euglenophyceae (e.g., *Phacus* spp.) reflected high organic enrichment and sediment–water interactions typical of shallow polymictic ecosystems (Wołowski, 2011; Afaf et al., 2022). In contrast, Kerrada reservoir supported a more balanced phytoplankton community, with Cyanophyceae, Chlorophyceae, and loricate *Trachelomonas* spp., typical of stratified mesotrophic ecosystems (Coesel & Krienitz, 2008). The contribution of diatoms and green algae further indicated more favorable ecological conditions (Salmaso, 2000; Padisák et al., 2009). Depth-



related contrasts were evident: diatom dominance in Cheliff reservoir was linked to sediment resuspension, whereas pelagic stratification in Kerrada reservoir favored taxa such as *Asterionella formosa* and *Fragilaria capucina*, characteristic of mesotrophic to oligotrophic ecosystems (Morais et al., 2018). Other phytoplankton groups, though less abundant, contributed to ecological functioning. Small colony-forming Chlorophyceae (e.g., *Actinastrum hantzschii*, *Monoraphidium contortum*) indicated eutrophic conditions in Cheliff reservoir (Deyab et al., 2021; Ruggeri et al., 2021). In Kerrada reservoir, cryptophytes such as *Cryptomonas pyrenoidifera* underscored the importance of mixotrophy in nutrient-poor stratified waters (Becker et al., 2010; Wang H. et al., 2024). IPL and IPLAC indices confirmed these patterns, classifying Cheliff as eutrophic–hypereutrophic and Kerrada as mesotrophic (Barbe et al., 2003; Feret & Laplace-Tretyure, 2013; Laplace-Tretyure & Feret, 2016).

The analysis of algal blooms strongly reinforced the trophic distinctions. Cheliff reservoir experienced frequent harmful algal blooms (HABs), including intense euglenoid blooms, and recurrent cyanobacterial blooms. These multispecies blooms reflect severe eutrophication, high nutrient loading, and weak ecological stability (Reynolds, 2006; Nasri et al., 2008; Bidi-Akli et al., 2014; Benayache et al., 2022; Guellati et al., 2024; Paerl et al., 2024). By contrast, Kerrada reservoir exhibited considerably lower bloom intensity and diversity, with maximum densities an order of magnitude lower than those of Cheliff reservoir. Bloom dynamics therefore provided strong ecological confirmation of the trophic state patterns revealed by the indices and plankton communities.

Zooplankton communities also reflected these trophic contrasts. Cheliff reservoir was dominated by rotifers (*Keratella valga*, *Synchaeta pectinata*, *Brachionus angularis*, *Polarthra dolichoptera*, *B. calyciflorus*), taxa strongly associated with eutrophic, turbid conditions (Sládeček, 1983; Picapedra et al., 2019; Liang et al., 2020; Baitiche et al., 2025). Kerrada reservoir, by contrast, supported a more balanced assemblage, with Cladocera and Copepoda co-occurring with rotifers, indicative of mesotrophic environments with more stable conditions and efficient energy transfer (Stamou et al., 2021; Baitiche et al., 2025). Among copepods, the cyclopoid *Acanthocyclops americanus* prevailed in Cheliff reservoir, consistent with shallow eutrophic ecosystems (Montagud et al., 2019; Pinto et al., 2023), whereas the calanoid *C. numidicus* dominated in Kerrada reservoir, a taxon linked to clearer, mesotrophic waters (Pinto-Coelho et al., 2005; Parra et al., 2009). These contrasting distributions highlight the high indicator value of zooplankton in assessing trophic state (Haberman & Haldna, 2014; Montagud et al., 2019; Muñoz-



Colmenares et al., 2021; Ochocka, 2021; Karpowicz et al., 2025). Zooplankton-based trophic state indices, however, displayed some divergence. In Cheliff reservoir, TSI_{Plecta} , $TSI_{CY/CA}$, and TSI_{NROT} confirmed eutrophy–hypereutrophy, while TSI_{BAC} and TSI_{CR} suggested mesotrophy. These lower values do not contradict eutrophication but instead reflect ecological processes specific to shallow polymictic ecosystems where crustaceans are suppressed by predation and habitat instability, and bacterivorous rotifers respond more to bacterial than phytoplankton production (Ejsmont-Karabin, 2012; Ejsmont-Karabin & Karabin, 2013; Ochocka, 2021; Karpowicz et al., 2025).

In Kerrada reservoir, TSI_{Plecta} pointed to eutrophy, while TSI_{NROT} , TSI_{CR} , and TSI_{BAC} indicated oligotrophic to mesotrophic conditions. These patterns are consistent with deeper stratified ecosystems, where zooplankton densities remain relatively low regardless of nutrient levels due to mixing constraints, predation pressure, and reduced habitat complexity (Haberman & Haldna, 2014; Karpowicz et al., 2025). Taken together, these indices do not imply inconsistency but rather illustrate the sensitivity of zooplankton communities to factors beyond nutrients, such as predation, hydrodynamic stability, and trophic efficiency. When integrated, microcrustacean and rotifer indices robustly differentiate the reservoirs and complement chemical and phytoplankton assessments.

Overall, our results demonstrate how morphometry and nutrient regimes jointly determine trophic status and plankton community composition. Cheliff reservoir is currently eutrophic to hypereutrophic, with cyanobacteria and rotifer dominance, reflecting impaired ecological quality. Kerrada reservoir maintains mesotrophic conditions with more balanced plankton assemblages, though it remains vulnerable to nutrient enrichment.

These findings underscore the value of integrating TSI, phytoplankton indices, and zooplankton-based assessments to monitor water quality under anthropogenic pressures. They further confirm that shallow reservoirs are more prone to eutrophication due to nutrient accumulation and limited dilution, while deeper reservoirs provide greater resilience through vertical stratification and sedimentation processes (Ejsmont-Karabin, 1995; Armengol & Miracle, 1999; García-Chicote et al., 2019; Montagud et al., 2019; Muñoz-Colmenares et al., 2021). The combined multi-indicator approach provides a more robust and nuanced assessment than any single metric alone.

Conclusion





Conclusion

This thesis provides a comprehensive and integrated assessment of the ecological functioning and water quality of two contrasting reservoirs in semi-arid northwestern Algeria, Cheliff and Kerrada reservoirs, based on two years of continuous monitoring. By combining physicochemical analyses with phytoplankton and zooplankton community studies, this work advances understanding of how reservoir morphometry, hydrological regimes, and environmental drivers shape plankton dynamics and trophic status in artificial freshwater ecosystems.

The review of existing literature highlighted the ecological importance of reservoirs as complex and dynamic freshwater ecosystems, where phytoplankton and zooplankton play central roles as primary producers and consumers, as well as sensitive bioindicators of water quality. While phytoplankton-based indices are already integrated into monitoring frameworks, zooplankton-based tools remain less standardized. This thesis therefore addressed this gap by integrating both planktonic compartments and adapting plankton-based indices to regional conditions, particularly in Mediterranean and semi-arid reservoirs.

Characterization of the study area revealed that, despite their geographic proximity, Cheliff and Kerrada reservoirs differ markedly in morphometry, hydrological function, exposure to anthropogenic pressures, and ecological settings. Cheliff reservoir, a shallow diversion system, is strongly influenced by agricultural runoff, wastewater inputs, high evaporation rates, and sediment resuspension. In contrast, Kerrada reservoir, a deep storage system, benefits from greater hydrological stability and buffering capacity, although it remains influenced by surrounding land use and by the water quality of Cheliff reservoir, which supplies it. These contextual differences are fundamental to interpreting plankton dynamics and ecological functioning.

Physicochemical analyses revealed pronounced spatial and seasonal variability, strongly influenced by morphometry and watershed characteristics. Cheliff reservoir consistently exhibited higher temperature, turbidity, salinity, and organic matter concentrations, reflecting its shallow depth, strong atmospheric exposure, and intense external nutrient loading. By contrast, Kerrada reservoir displayed greater physicochemical stability due to its depth and seasonal stratification, although it remained sensitive to nitrate enrichment. Seasonal variability was more pronounced in Cheliff reservoir than in Kerrada reservoir, while winter mixing in both reservoirs enhanced dissolved oxygen concentrations. These findings highlight the combined role of climatic forcing, hydrodynamics, and anthropogenic inputs in shaping reservoir water quality.



Phytoplankton community analyses confirmed that reservoir morphometry and seasonal environmental gradients are primary determinants of community structure. Cheliff's eutrophic, polymictic, and turbid conditions favored tolerant taxa such as Chlorophyceae and Euglenophyceae, whereas Kerrada's deeper, seasonally stratified waters promoted dinoflagellates and Cyanophyceae adapted to thermal stability and nutrient partitioning. Seasonal succession patterns were consistent in both reservoirs, with diatom dominance during cooler, well-mixed, oxygen-rich periods and cyanobacterial proliferation under warmer, stratified conditions.

Zooplankton analyses provided complementary insights into ecosystem functioning, trophic structure, and the influence of reservoir typology. Cheliff reservoir supported higher zooplankton densities dominated by rotifers, reflecting frequent mixing, elevated productivity, and environmental stress. In contrast, Kerrada reservoir exhibited lower overall densities and more balanced assemblages, shaped by stratification and reduced disturbance. These findings confirm reservoir typology as a key determinant of zooplankton structure and support the general observation that artificial reservoirs tend to host lower zooplankton diversity than natural lakes, due to altered hydrology and reduced habitat heterogeneity. Zooplankton communities therefore represent valuable indicators of environmental stress and contribute to ecosystem stability through grazing and nutrient recycling.

The application of trophic and plankton-based indices demonstrated that Cheliff reservoir exhibits eutrophic to hypereutrophic conditions, characterized by cyanobacterial dominance and rotifer prevalence, whereas Kerrada reservoir remains predominantly mesotrophic, with more balanced plankton communities. The combined application of Carlson's TSI, phytoplankton indices, and zooplankton metrics provided a coherent ecological assessment, confirming the diagnostic value of plankton communities. The combined use of Carlson's Trophic State Index, phytoplankton indices, and zooplankton metrics provided a coherent and robust framework for assessing ecological status, confirming the diagnostic value of plankton communities.

Overall, this research demonstrates that reservoir morphometry, hydrodynamics, and nutrient regimes are decisive factors shaping plankton communities and trophic status in semi-arid regions. Cheliff and Kerrada reservoirs exemplify the contrasting vulnerabilities of shallow and deep systems to nutrient enrichment and hydrological variability. By integrating physicochemical and biological indicators, this thesis delivers the first comprehensive plankton-based ecological assessment of Algerian reservoirs, addressing a significant gap in regional limnological research.



Beyond its ecological contributions, this work has important socioeconomic implications for water security and sustainable development in northwestern Algeria. By demonstrating how eutrophication processes, plankton dynamics, and reservoir typology directly influence water quality within Cheliff–Kerrada system, the study highlights the strong link between ecosystem degradation and the reliability, safety, and cost of drinking water production for Mostaganem–Arzew–Oran (M.A.O.) corridor. Deterioration of ecological conditions in upstream reservoirs, particularly Cheliff reservoir, can propagate downstream, increasing treatment complexity, operational costs, and public health risks. The integration of plankton-based bioindicators into routine monitoring programs therefore represents a cost-effective and scientifically robust approach to strengthening early warning systems, optimizing reservoir management, and supporting evidence-based policy decisions.

From a management perspective, the findings highlight the need for differentiated strategies tailored to reservoir typology. Shallow ecosystems like Cheliff reservoir require strict control of nutrient and organic matter inputs to mitigate eutrophication and harmful bloom formation, whereas deeper reservoirs such as Kerrada reservoir require sustained monitoring of external nutrient sources to preserve their buffering capacity and ecological stability.

In conclusion, this thesis reinforces the value of plankton as bioindicators of ecological quality, provides an operational framework for applying integrated assessment indices in semi-arid Mediterranean reservoirs, and contributes to a better understanding of reservoir functioning under anthropogenic and climatic pressures. Future research should focus on long-term monitoring, refinement of regional-specific plankton-based indices, and evaluation of climate change impacts on stratification and bloom dynamics, in order to support adaptive water management and biodiversity conservation in North African freshwater ecosystems.



Perspectives





Perspectives

This thesis provides the first integrated limnological assessment of two contrasting Algerian reservoirs by combining physicochemical analyses with phytoplankton and zooplankton community studies and the application of plankton-based indices. Beyond its scientific contributions, the work highlights several avenues for future research and management:

1. Long-term monitoring and climate change impacts

- Extending monitoring efforts over longer time scales is essential to detect interannual variability and to assess the impacts of climate change on hydrological regimes, bloom dynamics, and biodiversity. In particular, future studies should evaluate how projected increases in drought frequency, irregular rainfall, and temperature rise will affect eutrophication risks and plankton succession.

2. Development and adaptation of plankton-based indices

- While plankton indices are applied in Europe, their direct transfer to semi-arid reservoirs may not fully capture regional ecological dynamics. There is a need to adapt and calibrate phytoplankton and zooplankton indices to Mediterranean and North African conditions, incorporating local functional groups, bloom-forming taxa, and morphometric constraints.

3. Integration of functional and molecular approaches

- Moving beyond taxonomic inventories, functional-group analyses and molecular techniques (e.g., DNA metabarcoding) should be developed to provide finer resolution of community structure, particularly for cryptic taxa. Such approaches would improve the detection of early warning signals of eutrophication and biodiversity loss.

4. Linking plankton dynamics to higher trophic levels

- Further research should investigate how variations in phytoplankton and zooplankton structure influence fish populations and reservoir food webs. This would strengthen ecosystem-based management by linking lower trophic dynamics to fisheries productivity and biodiversity conservation.



5. Reservoir management strategies

- The contrasting dynamics observed between shallow and deep reservoirs call for differentiated management strategies. For shallow systems, emphasis should be placed on controlling nutrient inputs and reducing sediment resuspension, while for deeper systems, regulating nitrate inflows and maintaining hydrological stability are priorities. Adaptive management strategies integrating ecological indicators into decision-making are critical to ensuring sustainable use of reservoir resources.

6. Regional and comparative studies

- Expanding limnological research to other Algerian and North African reservoirs will allow comparative analyses of typologies, hydrological variability, and anthropogenic pressures. Building a regional database of plankton and environmental variables would enhance predictive models of reservoir functioning under semi-arid conditions.



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Supplementary material



Supplementary material

Table S1 Carlson's Trophic State Index (TSI) values and lake/reservoir classification (Carlson, 1977).

TSI values	Trophic status	Attributes
< 30	Oligotrophic	Clear water, hypolimnion remains oxygenated year-round.
30 – 40	Oligotrophic	Overall oligotrophic; shallow lakes may become anoxic in summer.
40 – 50	Mesotrophic	Moderately clear water; increasing probability of summer anoxia.
50 – 60	Eutrophic	Lower boundary of classical eutrophy; reduced transparency; fisheries restricted to warm-water species.
60 – 70	Eutrophic	Blue-green algae dominance; likely algal blooms; significant macrophyte issues.
70 – 80	Eutrophic	Severe algal blooms possible throughout summer; often hypereutrophic.
> 80	Hypertrophic	Frequent algal scums; summer fish kills; few macrophytes.

Table S2 Monthly averages of minimum and maximum temperatures and precipitation in the study area (2012–2022).

Months	Jan.	Feb.	Mar.	Apr.	May	Jun.	Jul.	Aug.	Sep.	Oct.	Nov.	Dec.	Total
T _m (°C)	4.4	5.3	7.3	10.3	13.1	17.2	20.4	21.0	15.5	11.3	9.6	6.2	/
T _M (°C)	16.0	16.9	17.8	21.2	25.0	28.1	28.9	29.0	27.2	25.9	19.8	16.2	/
P (mm)	79.47	51.56	57.17	26.76	16.44	3.74	0.46	0.85	17.04	22.62	69.21	54.46	399.81

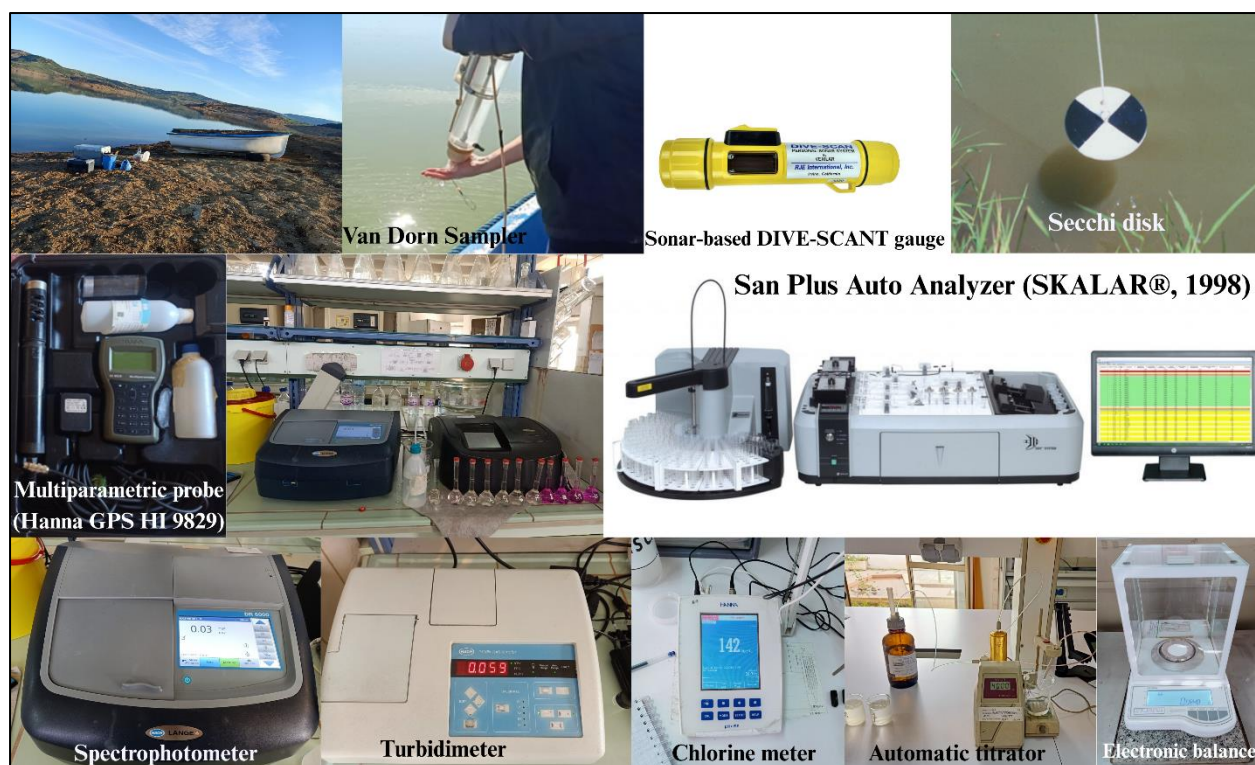


Fig. S1. Materials used for *in situ* water monitoring and *ex situ* analyses at the SEAAL laboratory.

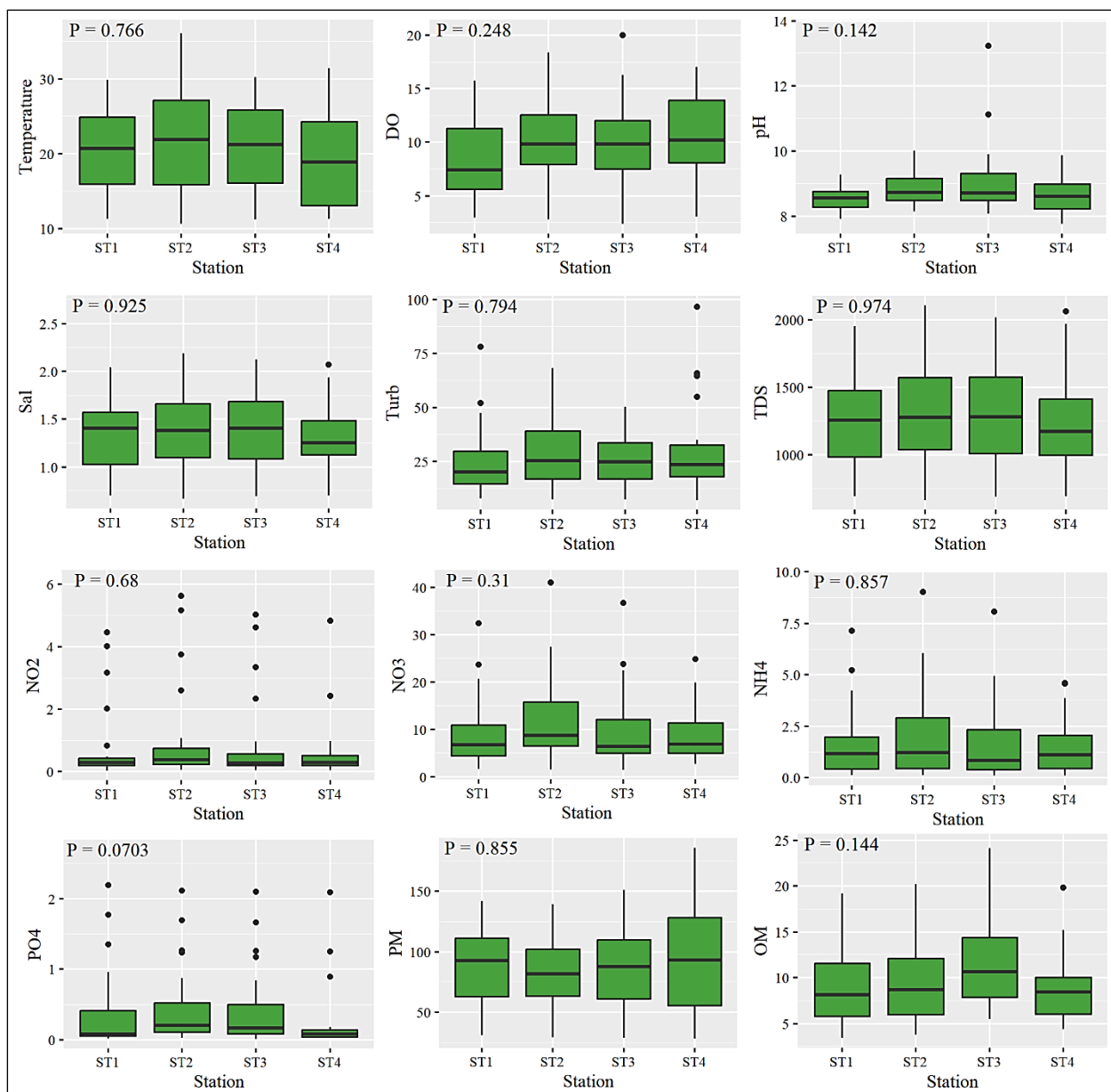


Fig. S2. Comparison of physicochemical parameters in Cheliff reservoir between stations (Kruskal–Wallis test, 95% confidence interval).

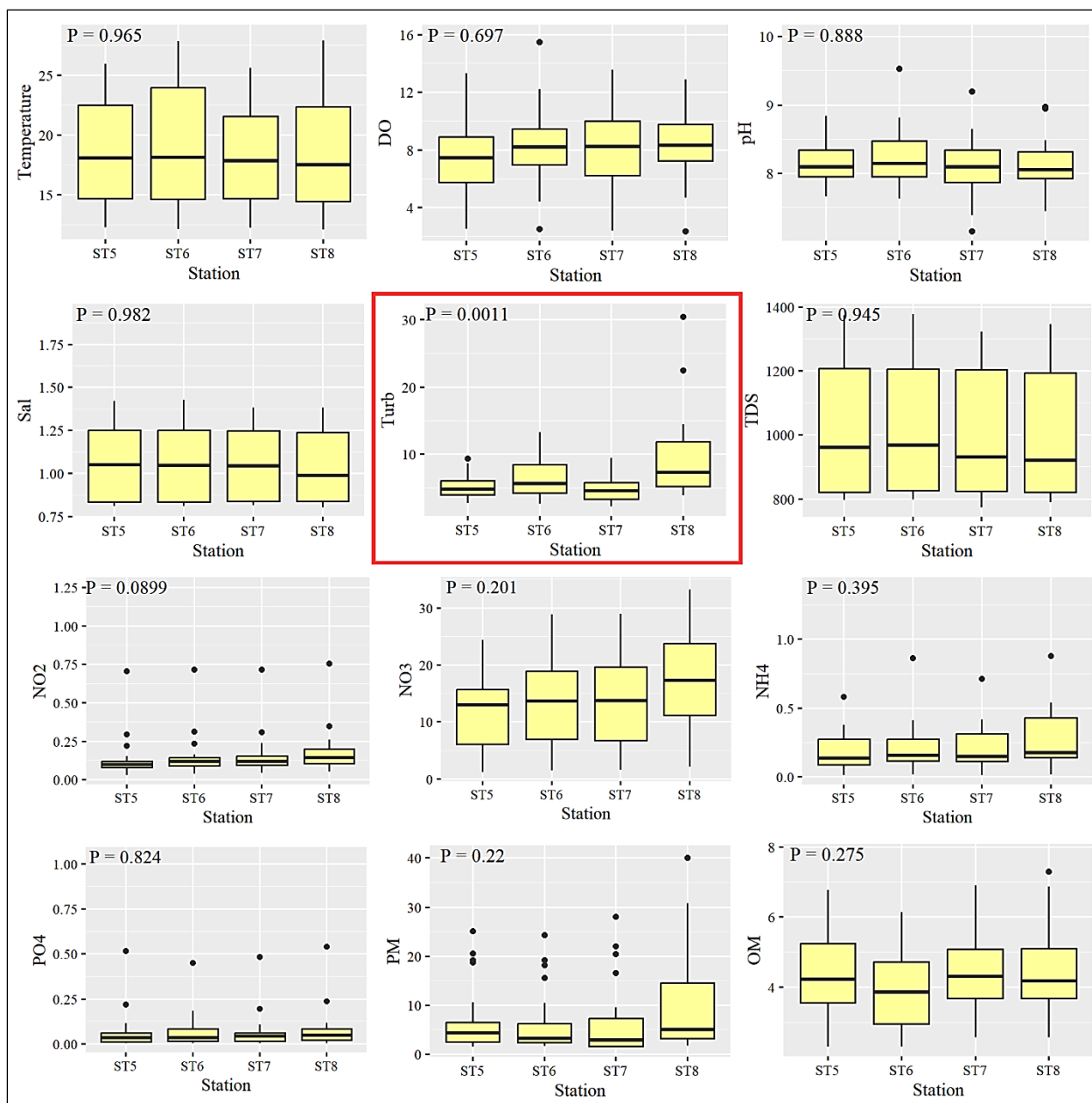


Fig. S3. Comparison of physicochemical parameters in Kerrada reservoir between stations (Kruskal–Wallis test, 95% confidence interval).



Fig. S4. Plankton nets with mesh sizes of 20 μm for phytoplankton and 50 μm for zooplankton.

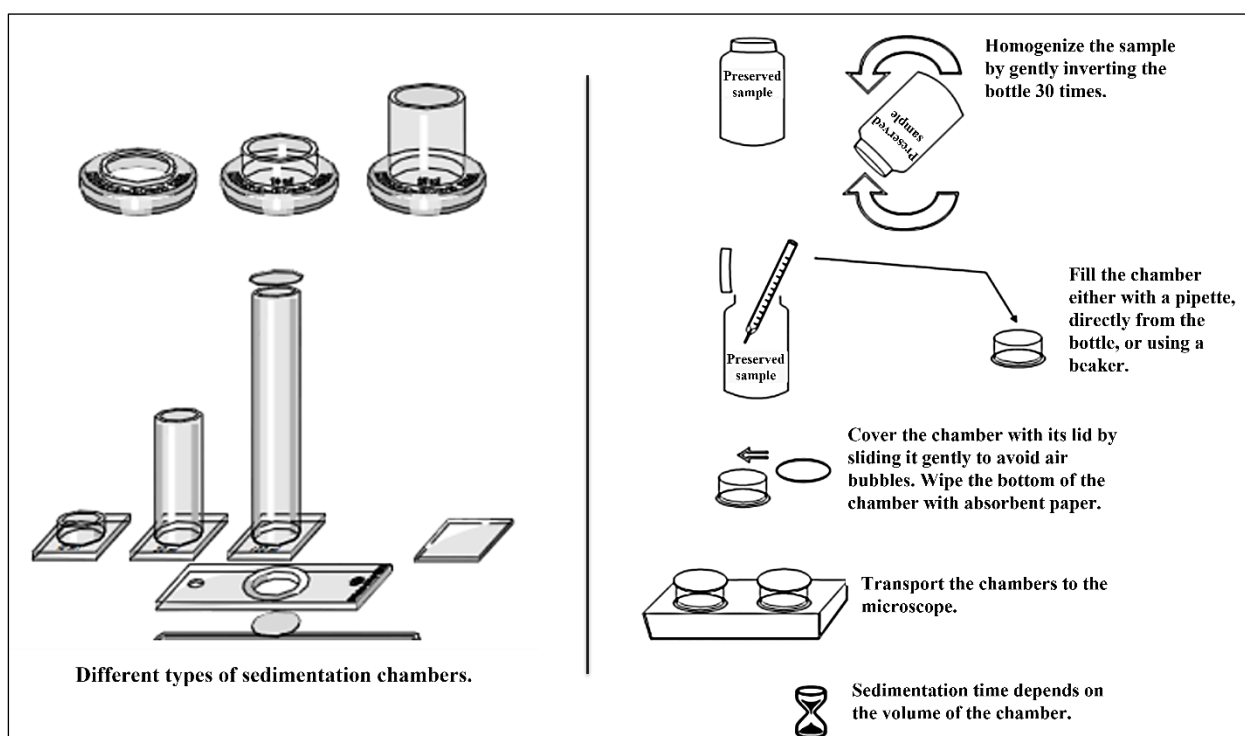


Fig. S5. Utermöhl sedimentation technique (adapted from, Utermöhl, 1958).

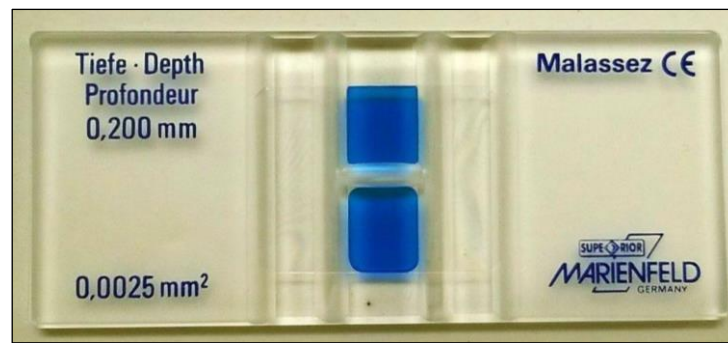


Fig. S6. Malassez chamber.



Fig. S7. Inverted microscope.

Table S3. Species composition and distribution of phytoplankton assemblage in Cheliff and Kerrada reservoirs.

Species name	Code	Dominance (%)	
		Cheliff	Kerrada
Chlorophyceae		31.4%	26.4%
<i>Actinastrum hantzschii</i> (Lagerheim 1882)	CHL1	+	+
<i>Ankistrodesmus fusiformis</i> (Corda 1838)	CHL2	+	/
<i>Botryococcus braunii</i> (Kützing 1849)	CHL3	+	+
<i>Chlamydomonas geitleri</i> (Ettl 1969)	CHL4	+	+
<i>Chlamydomonas</i> sp.	CHL5	+	+
<i>Chlorella vulgaris</i> (Beijerinck 1890)	CHL6	++	++
<i>Coelastrum astroideum</i> (De Notaris 1867)	CHL7	+	/
<i>Coelastrum microporum</i> (Nägeli 1855)	CHL8	+	+
<i>Desmodesmus opoliensis</i> (P.G.Richter) (E.Hegewald 2000)	CHL9	+	+
<i>Desmodesmus armatus</i> var. <i>longispina</i> (Chodat) (E.Hegewald 2000)	CHL10	+	/

<i>Desmodesmus communis</i> (E.Hegewald 2000)	CHL11	+	+
<i>Desmodesmus protuberans</i> (F.E.Fritsch & M.F.Rich) (E.Hegewald 2000)	CHL12	+	/
<i>Elakatothrix genevensis</i> (Reverdin) (Hindák 1962)	CHL13	+	/
<i>Eudorina</i> sp.	CHL14	+	+
<i>Gloeocystis</i> sp.	CHL15	+	+
<i>Golenkinia radiata</i> (Chodat 1894)	CHL16	+	/
<i>Gonium pectorale</i> (O.F.Müller 1773)	CHL17	+	/
<i>Kirchneriella obesa</i> (West & G.S.West 1894)	CHL18	+	+
<i>Lemmermannia tetrapedia</i> (Kirchner) (Lemmermann 1904)	CHL19	+	/
<i>Micractinium pusillum</i> (Fresenius 1858)	CHL20	+	+
<i>Monactinus simplex</i> (Meyen) (Corda 1839)	CHL21	+	/
<i>Monoraphidium contortum</i> (Thuret) (Komárková-Legnerová 1969)	CHL22	+	+
<i>Monoraphidium griffithii</i> (Berkeley) (Komárková-Legnerová 1969)	CHL23	+	+
<i>Mucidosphaerium pulchellum</i> (H.C.Wood) (C.Bock, Proschold & Krienitz 2011)	CHL24	+	+
<i>Neglectella solitaria</i> (Wittrock) (Stenclová & Kaštovský 2017)	CHL25	+	+
<i>Parietochloris pseudoalveolaris</i> (T.R.Deason & H.C.Bold) (Shin Watanabe & G.L.Floyd 1991)	CHL26	+	+
<i>Oedogonium sexangulare</i> (Cleve ex Hirn 1900)	CHL27	+	+
<i>Oocystis borgei</i> (J.W.Snow 1903)	CHL28	+	+
<i>Oocystis lacustris</i> (Chodat 1897)	CHL29	+	+
<i>Oocystis marssonii</i> (Lemmermann 1898)	CHL30	+	+
<i>Pandorina morum</i> (O.F.Müller) (Bory 1826)	CHL31	+	+
<i>Pediastrum duplex</i> (Meyen 1829)	CHL32	+	/
<i>Planctonema lauterbornii</i> (Schmidle 1903)	CHL33	+	+
<i>Planktosphaeria gelatinosa</i> (G.M.Smith 1918)	CHL34	+	+
<i>Pseudopediastrum boryanum</i> (Turpin) (E.Hegewald 2005)	CHL35	+	/
<i>Schizochlamys gelatinosa</i> (A.Braun 1849)	CHL36	+	+
<i>Sphaerocystis schroeteri</i> (Chodat 1897)	CHL37	+	+
<i>Stigeoclonium tenue</i> (C.Agardh) (Kützing 1843)	CHL38	+	+
<i>Tetrademus obliquus</i> (Turpin) (M.J.Wynne 2016)	CHL39	+	/
<i>Tetrademus lagerheimii</i> (M.J.Wynne & Guiry 2016)	CHL40	+	+
<i>Tetraëdron minimum</i> (A.Braun) (Hansgirg 1889)	CHL41	+	+
<i>Tetraspora gelatinosa</i> (Vaucher) (Desvaux 1818)	CHL42	+	+
<i>Tetrastrum heteracanthum</i> (Nordstedt) (Chodat 1895)	CHL43	+	+
<i>Westella botryoides</i> (West) (De Wildeman 1897)	CHL44	+	+
<i>Willea rectangularis</i> (A.Braun) (D.M.John, M.J.Wynne & P.M.Tsarenko 2014)	CHL45	+	+
Bacillariophyceae (Diatomophyceae)		11.3%	7.8%
<i>Achnanthyidium catenatum</i> (Bílý & Marvan) (Lange-Bertalot 1999)	DIA1	+	/
<i>Actinella punctata</i> (F.W.Lewis 1864)	DIA2	+	+
<i>Actinoptychus senarius</i> (Ehrenberg 1843)	DIA3	+	+
<i>Amphora ovalis</i> (Kützing 1844)	DIA4	+	+

<i>Asterionella formosa</i> (Hassall 1850)	DIA5	/	+
<i>Actinoptychus senarius</i> (Ehrenberg 1843)	DIA6	+	+
<i>Cocconeis placentula</i> (Ehrenberg 1838)	DIA7	+	/
<i>Coscinodiscus</i> sp.	DIA8	+	+
<i>Craticula ambigua</i> (Ehrenberg) (D.G.Mann 1990)	DIA9	+	/
<i>Craticula cuspidata</i> (Kützing) (D.G.Mann 1990)	DIA10	+	+
<i>Cyclotella atomus</i> (Hustedt 1937)	DIA11	+	+
<i>Discostella stelligera</i> (Cleve & Grunow) (Houk & Klee 2004)	DIA12	+	+
<i>Cymatopleura elliptica</i> (W.Smith 1851)	DIA13	+	+
<i>Diatoma vulgare</i> (Bory 1824)	DIA14	+	+
<i>Diploneis ovalis</i> (Hilse) (Cleve 1891)	DIA15	+	/
<i>Entomoneis alata</i> (Ehrenberg 1845)	DIA16	+	+
<i>Fragilaria capucina</i> (Desmazières 1830)	DIA17	/	+
<i>Frustulia rhomboides</i> (Ehrenberg) (De Toni 1891)	DIA18	+	+
<i>Gomphonema parvulum</i> (Kützing 1849)	DIA19	+	/
<i>Gyrosigma acuminatum</i> (Kützing) (Rabenhorst 1853)	DIA20	+	+
<i>Hantzschia amphioxys</i> (Ehrenberg) (Grunow 1880)	DIA21	+	+
<i>Hyalodiscus</i> sp.	DIA22	/	+
<i>Navicula gregaria</i> (Donkin 1861)	DIA23	+	+
<i>Eolimna minima</i> (Grunow) (Lange-Bertalot, nom. illeg. 1998)	DIA24	+	+
<i>Nitzschia acicularis</i> (Kützing) (W.Smith 1853)	DIA25	+	+
<i>Nitzschia dissipata</i> (Kützing) (Rabenhorst 1860)	DIA26	+	/
<i>Nitzschia draveillensis</i> (Coste & Ricard 1980)	DIA27	+	+
<i>Nitzschia palea</i> (Kützing) (W.Smith 1856)	DIA28	+	+
<i>Nitzschia sigmaidea</i> (Nitzsch) (W.Smith 1853)	DIA29	+	+
<i>Nitzschia vermicularis</i> (Kützing) (Hantzsch 1860)	DIA30	+	+
<i>Pantocsekiella ocellata</i> (Pantocsek) (K.T.Kiss & Ács 2016)	DIA31	+	+
<i>Stephanocyclus meneghinianus</i> (Kützing) (Kulikovskiy, Genkal & Kociolek 2022)	DIA32	++	+
<i>Stephanodiscus neoastraea</i> (Håkansson & Hickel 1986)	DIA33	+	+
<i>Surirella ovalis</i> (Brébisson 1838)	DIA34	+	/
<i>Surirella librile</i> (Ehrenberg 1845)	DIA35	+	/
<i>Thalassiosira</i> sp.	DIA36	+	+
<i>Ulnaria ulna</i> (Nitzsch) (Compère 2001)	DIA37	+	+
Cyanophyceae		33.4%	46.6%
<i>Aphanizomenon flos-aquae</i> (Ralfs ex Bornet & Flahault 1886)	CYAN1	+	+
<i>Aphanocapsa elachista</i> (West & G.S.West 1894)	CYAN2	+	+
<i>Aphanothece variabilis</i> (J.Schiller) (Komárek 1995)	CYAN3	+	+
<i>Chlorogloea microcystoides</i> (Geitler 1925)	CYAN4	+	+
<i>Chroococcus minutus</i> (Kützing) (Nägeli 1849)	CYAN5	+	+
<i>Coelomorion pusillum</i> (Van Goor) (Komárek 1988)	CYAN6	+	+

<i>Coelosphaerium</i> sp.	CYAN7	/	+
<i>Raphidiopsis raciborskii</i> (Wołoszyńska) (Aguilera & al. 2018)	CYAN8	+	+
<i>Dolichospermum flos-aquae</i> (Bornet & Flahault) (P.Wacklin, L.Hoffmann & Komárek 2009)	CYAN9	+	+
<i>Entophysalis arboriformis</i> (Kaštovský, Fucíková, Hauer & Bohunická 2011)	CYAN10	/	+
<i>Gloeocapsa rupestris</i> (Kützing 1847)	CYAN11	+	+
<i>Gomphosphaeria aponina</i> (Kützing 1836)	CYAN12	+	+
<i>Leptolyngbya granulifera</i> (J.J.Copeland) (Anagnostidis 1936)	CYAN13	+	+
<i>Limnospira platensis</i> (Gomont) (K.R.S.Santos & Hentschke 2023)	CYAN14	+	+
<i>Limnothrix redekei</i> (Goor) (Meffert 1987)	CYAN15	+	+
<i>Merismopedia tenuissima</i> (Lemmermann 1898)	CYAN16	+	+
<i>Microcoleus fonticola</i> (Kirchner) (Strunecky, Komárek & J.R.Johansen 2013)	CYAN17	+	+
<i>Microcystis aeruginosa</i> (Kützing 1846)	CYAN18	+	+
<i>Microcystis botrys</i> (Teiling 1942)	CYAN19	+	+
<i>Microcystis flos-aquae</i> (Wittrock) (Kirchner 1898)	CYAN20	+	+
<i>Oscillatoria limosa</i> (C.Agardh ex Gomont 1892)	CYAN21	/	+
<i>Oscillatoria tenuis</i> (C.Agardh ex Gomont 1892)	CYAN22	+	+
<i>Planktolingbya limnetica</i> (Lemmermann) (Komárková-Legnerová & Cronberg 1992)	CYAN23	+	+
<i>Planktothrix agardhii</i> (Gomont) (Anagnostidis & Komárek 1988)	CYAN24	+	+
<i>Pseudanabaena catenata</i> (Lauterborn 1915)	CYAN25	+	+
<i>Pseudanabaena mucicola</i> (Naumann & Huber-Pestalozzi) (Schwabe 1964)	CYAN26	+	+
<i>Raphidiopsis curvata</i> (F.E.Fritsch & M.F.Rich 1930)	CYAN27	+	+
<i>Synechococcus nidulans</i> (Pringsheim) (Komárek 1970)	CYAN28	+	+
<i>Woronichinia naegeliana</i> (Unger) (Elenkin 1933)	CYAN29	+	+
Euglenophyceae (Euglenoids)		19.8%	3.1%
<i>Euglena oblonga</i> (F.Schmitz 1884)	EUG1	+	+
<i>Euglenaria caudata</i> (E.F.W.Hübner) (Karnkowska-Ishikawa & E.W.Linton 2010)	EUG2	+++	+
<i>Lepocinclis acus</i> (O.F.Müller) (B.Marin & Melkonian 2003)	EUG3	+	+
<i>Lepocinclis oxyuris</i> (Schmarda) (B.Marin & Melkonian 2003)	EUG4	+	+
<i>Phacus acuminatus</i> (A.Stokes 1885)	EUG5	+	+
<i>Phacus curvicauda</i> (Svirénko 1915)	EUG6	+	/
<i>Strombomonas fluviatilis</i> (Lemmermann) (Deflandre 1930)	EUG7	+	+
<i>Trachelomonas armata</i> (Ehrenberg) (F.Stein 1878)	EUG8	+	/
<i>Trachelomonas caudata</i> (Ehrenberg) (F.Stein 1878)	EUG9	+	+
<i>Trachelomonas hispida</i> (Perty) (F.Stein 1878)	EUG10	+	+
<i>Trachelomonas volvocina</i> (Ehrenberg 1834)	EUG11	+	+++
Zygnematophyceae		2.9%	6.9%
<i>Closterium lanceolatum</i> (Kützing ex Ralfs 1848)	DES1	+	+
<i>Closterium aciculare</i> (T.West 1860)	DES2	++	++
<i>Closterium acutum</i> (Brébisson 1848)	DES3	++	+

<i>Closterium acutum</i> var. <i>variabile</i> (Lemmermann) (Willi Krieger 1935)	DES4	+	+
<i>Cosmarium depressum</i> (Kützing 1849)	DES5	+	+
<i>Cosmarium granatum</i> (Brébisson ex Ralfs 1848)	DES6	+	/
<i>Cosmarium humile</i> (Nordstedt ex De Toni 1889)	DES7	+	+
<i>Cosmarium punctulatum</i> (Brébisson 1856)	DES8	+	+
<i>Spirogyra crassa</i> (Kützing) (Kützing 1843)	DES9	/	+
<i>Staurostrum gracile</i> (Ralfs ex Ralfs 1848)	DES10	+	+
<i>Staurostrum tetracerum</i> (Ralfs ex Ralfs 1848)	DES11	/	+
Dinophyceae (Pyrrophyceae)		1.1%	7.8%
<i>Ceratium hirundinella</i> (O.F.Müller) (Dujardin 1841)	DINO1	+	+++
<i>Gymnodinium</i> sp.	DINO2	/	+
<i>Peridinium bipes</i> (F.Stein 1883)	DINO3	++	+
<i>Peridinium cinctum</i> (O.F.Müller) (Ehrenberg 1832)	DINO4	+++	++
Chrysophyceae (Golden algae)		0.1%	0.9%
<i>Dinobryon pediforme</i> (Lemmermann) (Steinecke 1916)	CHRY1	++	+++
<i>Mallomonas caudata</i> (Iwanoff [Ivanov] 1899)	CHRY2	+++	++
Cryptophyceae (Cryptomonads)		0.04%	0.5%
<i>Cryptomonas pyrenoidifera</i> (Geitler 1922)	CRYP1	++++	++++

“/” absence in the corresponding reservoir. (+ < 25%, 25% < ++ < 50%, 50% < +++ < 75%, ++++ ≥ 75%).

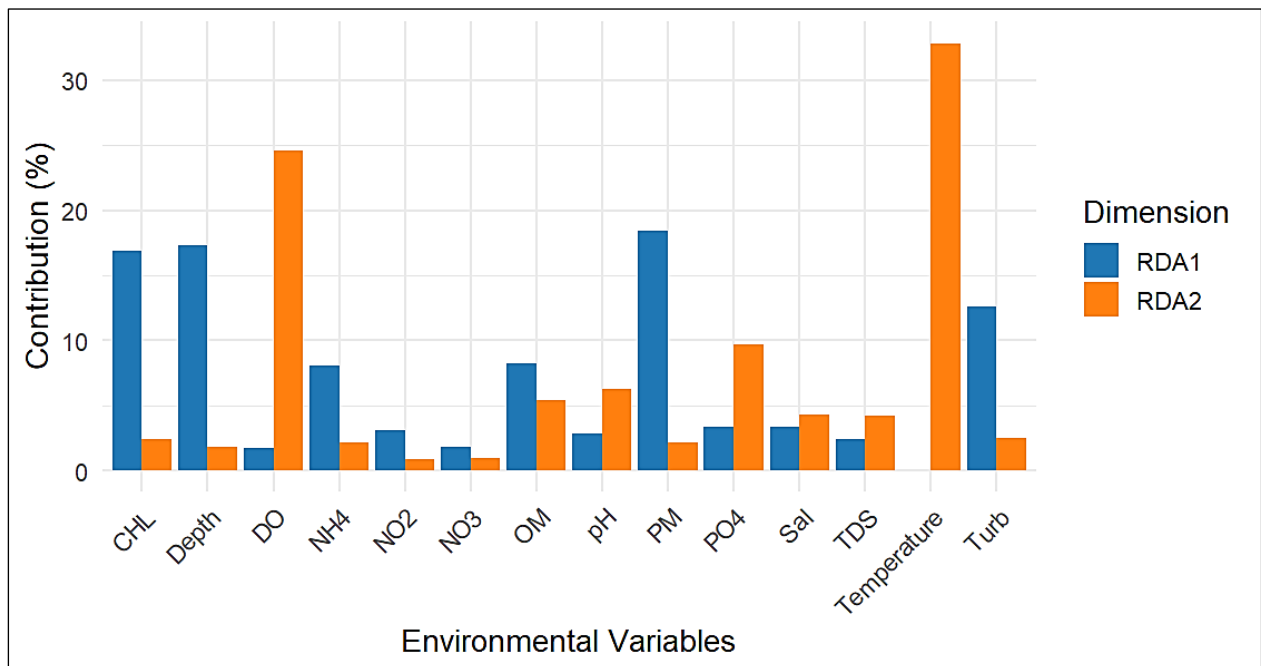


Fig. S8. Contribution of environmental factors to Axes 1 and 2 in the phytoplankton–environment RDA.

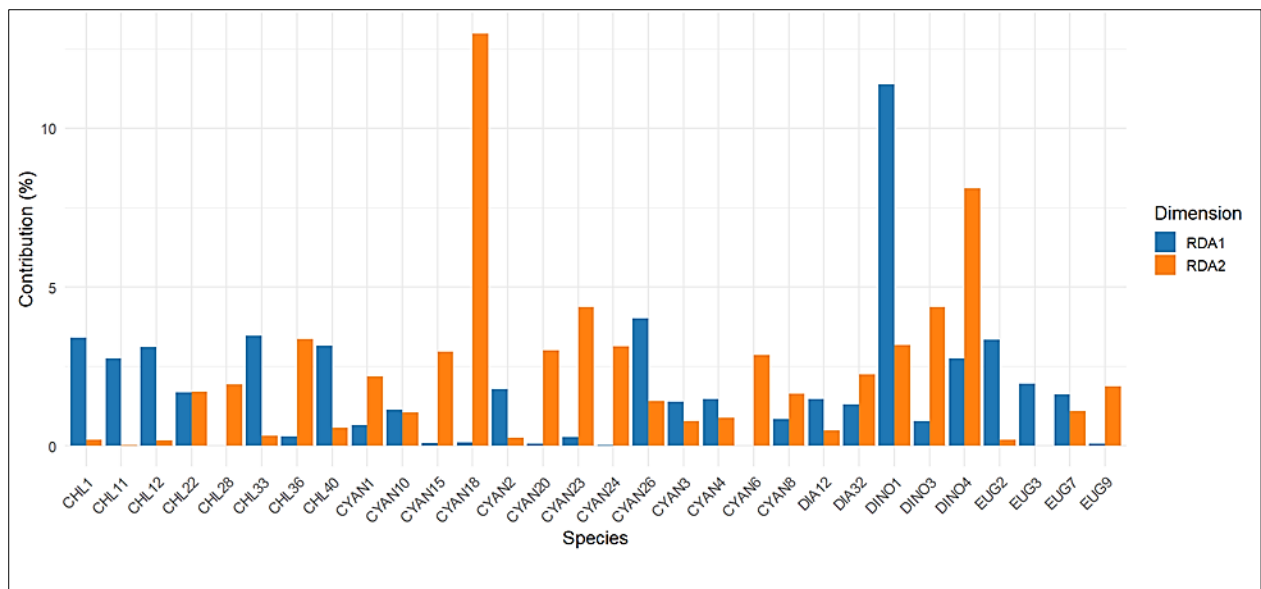


Fig. S9. Contribution of top 30 phytoplankton species to Axes 1 and 2 in the phytoplankton–environment RDA.

Table S4. Redundancy Analysis (RDA) axis scores of phytoplankton species related to axis 1 and axis 2.

	RDA1	RDA2
Phytoplankton species		
DINO1	0.379826	-0.1302
CYAN18	0.035585	0.263206
DINO4	0.186538	-0.20797
CYAN26	-0.2257	-0.08672
DINO3	0.099203	-0.15288
CYAN23	-0.05964	-0.15285
CHL33	0.209553	-0.04177
CHL40	-0.20009	-0.05471
CHL36	-0.06246	0.13396
CHL1	-0.2078	0.032283
DIA32	-0.1283	0.109691
EUG2	-0.20556	0.031397
CHL22	-0.14573	-0.09528
CHL12	-0.19863	-0.02966
CYAN24	-0.0165	-0.12948
CYAN20	0.029078	0.126814
CYAN15	0.033099	-0.12592
CYAN6	-0.00233	0.123781
CYAN1	-0.09125	-0.10827
CHL11	-0.18661	-0.01293
EUG7	-0.14322	-0.0764
CYAN8	-0.10378	-0.09369
CYAN4	0.136257	0.069032
CYAN10	0.120072	0.075033
CYAN3	0.132345	0.064772
CYAN2	0.150444	0.037088

EUG3	-0.15773	0.006577
DIA12	-0.13618	0.050865
EUG9	-0.03042	-0.09979
CHL28	-0.00846	-0.10151



Fig. S10. Stereoscope (binocular microscope).



Fig. S11. Sedgewick–Rafter chamber.

Table S5. Species composition and distribution of zooplankton assemblage in Cheliff and Kerrada reservoirs.

Species name	Code	Dominance (%)	
		Cheliff	Kerrada
Rotifera		94%	42%
<i>Adineta</i> sp	Asp	+	+
<i>Asplanchna brightwellii</i> (Gosse, 1850)	Ab	+	+
<i>Brachionus angularis</i> (Gosse, 1851)	Ba	+	+
<i>Brachionus calyciflorus</i> (Pallas, 1766)	Bc	+	+
<i>Filinia terminalis</i> (Plate, 1886)	Ft	+	+
<i>Hexarthra mira</i> (Hudson, 1871)	Hm	+	++
<i>Keratella cochlearis</i> (Gosse, 1851)	Kc	+	+
<i>Keratella quadrata</i> (Müller, 1786)	Kq	+	+
<i>Keratella cochlearis</i> f. <i>tecta</i> (Gosse, 1851)	Kt	+	+
<i>Keratella tropica</i> (Apstein, 1907)	Ktr	+	+
<i>Keratella valga</i> (Ehrenberg, 1834)	Kv	++	+
<i>Polyarthra dolichoptera</i> (Idelson, 1925)	Pd	+	+
<i>Trichocerca similis</i> (Wierzejski, 1893)	Ts	+	+
<i>Keratella hiemalis</i> (Carlin, 1943)	Kh	+	/
<i>Pompholyx sulcata</i> (Hudson, 1885)	Ps	+	/
<i>Synchaeta pectinata</i> (Ehrenberg, 1832)	Sp	++	/
<i>Brachionus durgae</i> (Dhanapathi, 1974)	Bd	+	/
<i>Brachionus plicatilis</i> (Müller, 1786)	Bp	+	/
<i>Colurella colurus</i> (Ehrenberg, 1830)	Cc	+	/
Cladocera		3%	46%
<i>Bosmina longirostris</i> (O.F. Müller, 1776)	Bl	++	++
<i>Daphnia longispina</i> (O.F. Müller, 1776)	DI	+	+
<i>Diaphanosoma brachyurum</i> (Liévin, 1848)	Db	+	+++
<i>Leptodora kindtii</i> (Focke, 1844)	Lk	/	+
<i>Moina micrura</i> (Kurz, 1875)	Mm	++	+
<i>Ceriodaphnia reticulata</i> (Jurine, 1820)	Cr	+	/
<i>Daphnia magna</i> (Straus, 1820)	Dm	+	/
Copepoda		19.8%	3.1%
<i>Acanthocyclops americanus</i> (Marsh, 1893)	Aa	++++	+
<i>Eucyclops serrulatus</i> (Fischer, 1851)	Es	+	/
<i>Copidodiaptomus numidicus</i> (Gurney, 1909)	Cn	+	++++

“/” absence in the corresponding reservoir. (+ < 25%, 25% < ++ < 50%, 50% < +++ < 75%, ++++ ≥ 75%).

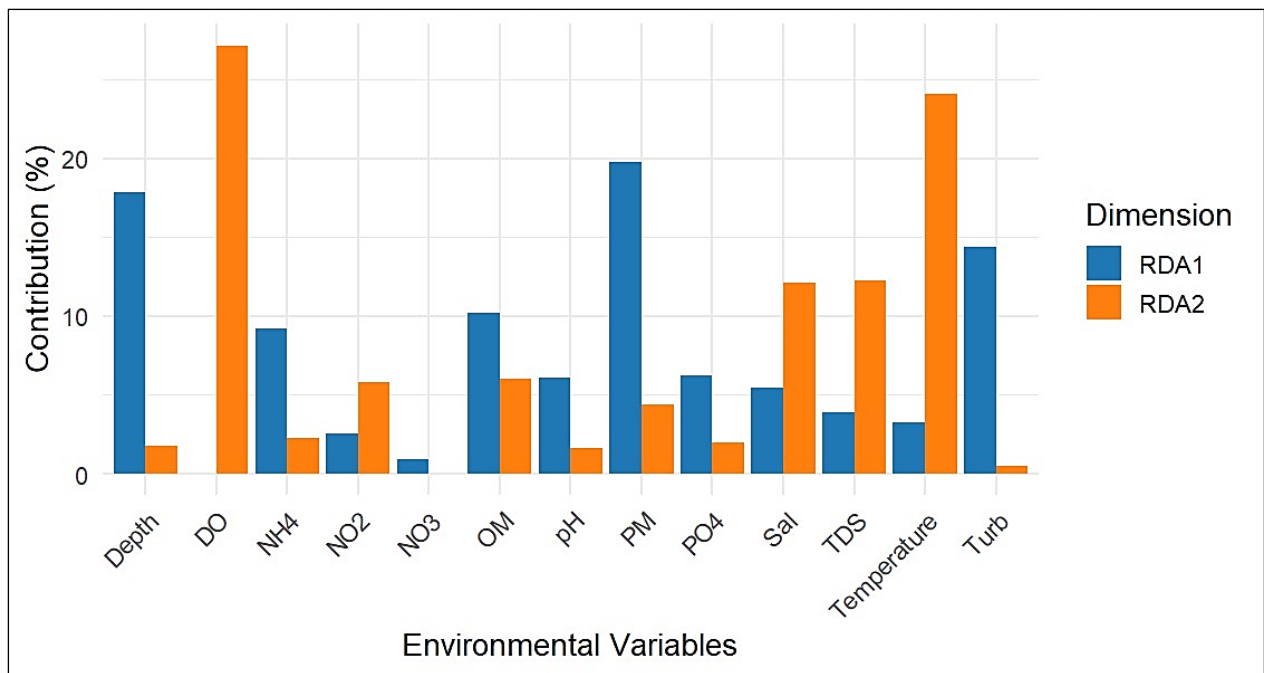


Fig. S12. Contribution of environmental factors to Axes 1 and 2 in the zooplankton–environment RDA.

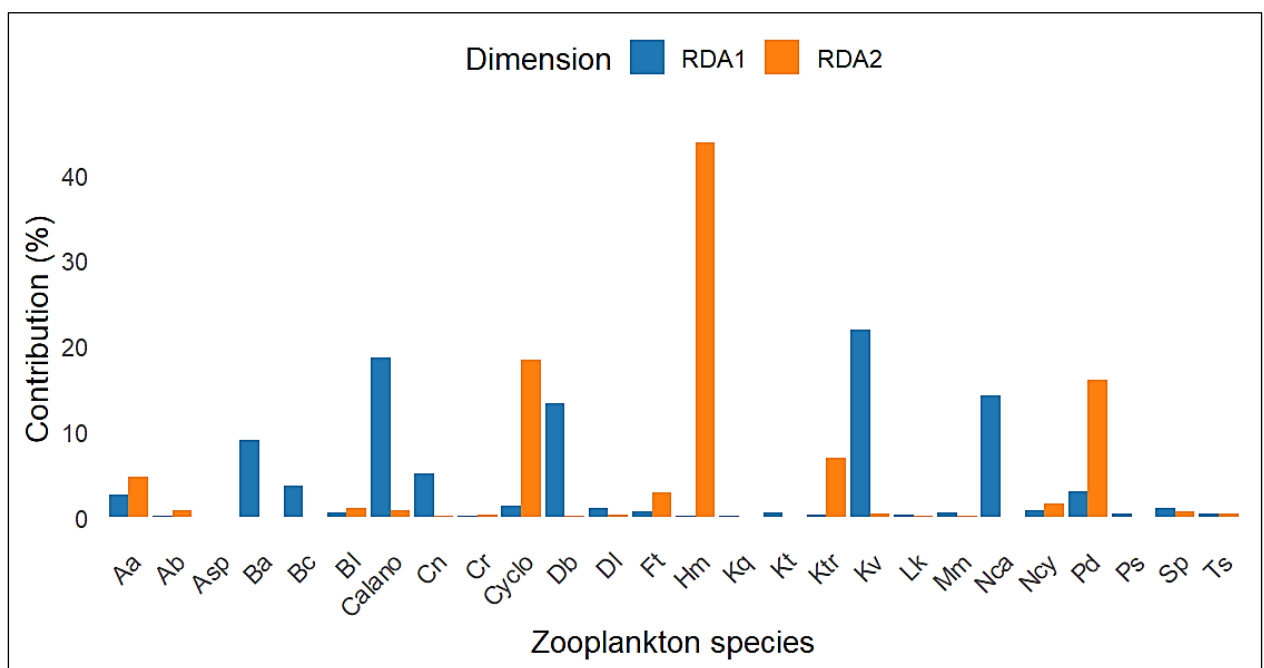


Fig. S13. Contribution of zooplankton species to Axes 1 and 2 in the zooplankton–environment RDA.

Table S6. Redundancy Analysis (RDA) axis scores of zooplankton species related to axis 1 and axis 2.

	RDA1	RDA2
Zooplankton species		
Db	0.76502	0.04551
Calano	0.70889	-0.05856
Nca	0.60339	-0.05871
Cn	0.45113	-0.02361
DI	0.17789	0.03324
BI	0.14725	0.02542
Ktr	0.13970	0.18449
Lk	0.13487	0.08768
Hm	0.08102	0.46578
Asp	0.05579	0.02767
Kq	0.04036	-0.05319
Ab	-0.03919	0.06768
Cr	-0.05335	0.06815
Ps	-0.08517	0.04134
Mm	-0.09271	0.06545
Ft	-0.09848	0.15460
Ts	-0.10941	0.09280
Ncy	-0.12189	-0.14340
Kt	-0.11496	0.00971
Sp	-0.14631	-0.01781
Pd	-0.17823	0.30646
Aa	-0.25596	-0.14467
Cyclo	-0.26695	-0.36588
Bc	-0.29479	-0.01420
Ba	-0.46296	0.05306
Kv	-0.66379	0.10400

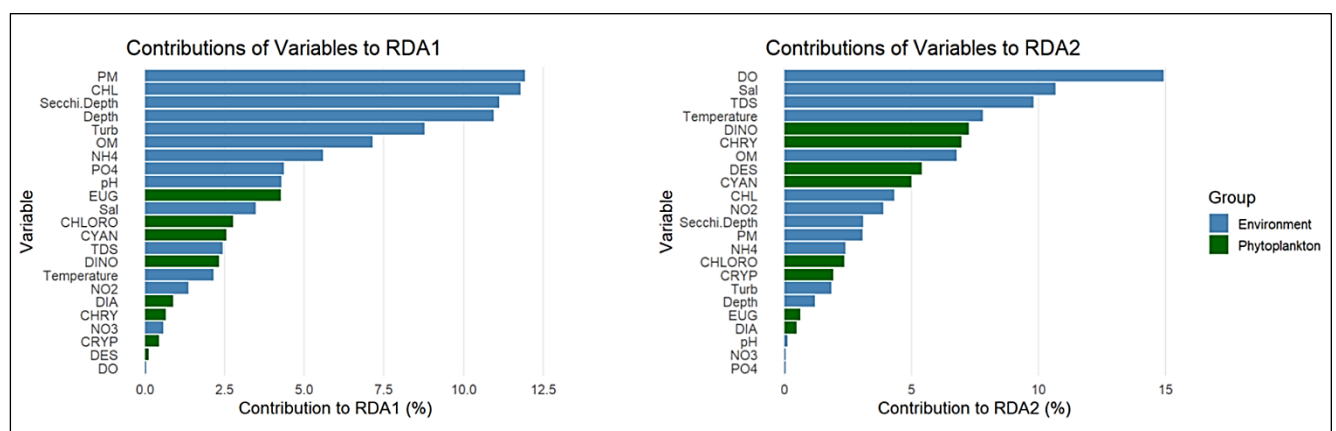
**Fig. S14.** Contribution of environmental factors and phytoplankton groups to Axes 1 and 2 in the zooplankton ~ Phytoplankton + environment RDA.

Table S7. Redundancy Analysis (RDA) axis scores of environmental factors and phytoplankton groups related to axis 1 and axis 2.

	RDA1	RDA2
Aa	-0.2397	-0.3379
Cn	0.51563	-0.1323
Bl	0.12964	0.08019
Dl	0.17463	0.0312
Db	0.7226	-0.1221
Lk	0.12048	-0.0264
Mm	-0.0991	0.02025
Asp	0.03715	0.00797
Ab	-0.0461	0.07972
Ba	-0.4902	-0.0622
Bc	-0.2631	-0.0827
Ft	-0.1239	0.13878
Hm	0.07863	0.68583
Kq	0.12569	-0.0355
Kt	-0.1144	-0.0163
Ktr	0.13881	0.28434
Kv	-0.7733	-0.0529
Pd	-0.2433	0.382
Ts	-0.1175	0.13376
Ps	-0.0922	-0.0148
Sp	-0.1472	-0.0743
Cr	-0.0588	0.03899

Table S8. Algal Groups and their coefficients (Qi) for the IPL index (Barbe et al., 2003).

Algal Groups	Qi
Desmids	1
Diatoms	3
Chrysophyceae	5
Dinophyceae and Cryptophyceae	9
Chlorophyceae (except Desmids)	12
Cyanophyceae	16
Euglenoids	20

Table S9. Relative abundance scale for the IPL index (Barbe et al., 2003).

%	0 – < 10	10 – < 30	30 – < 50	50 – < 70	70 – < 90	90 – < 100
Aj	0	1	2	3	4	5