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Research papers

Anthropogenic and climatic factors enhancing hypolimnetic anoxia in a temperate mountain lake

Javier Sánchez-España [a,*], M. Pilar Mata [a], Juana Vegas [b], Mario Morellón [c], Juan Antonio Rodríguez [d], Ángel Salazar [a], Iñaki Yusta [e], Aida Chaos [f], Carmen Pérez-Martínez [g], Ana Navas [h]

[a] Department of Geological Resources Research, Spanish Geological Survey (IGME), Calera, 1, 28760 Tres Cantos, Madrid, Spain
[b] Department of Geological Resources Research, Spanish Geological Survey (IGME), Ríos Rosas, 23, 28003 Madrid, Spain
[c] CITIMAC, Facultad de Ciencias, University of Cantabria, Avenida de los Castros s/n, 39005 Santander (Cantabria), Spain
[d] Department of Geoscientific Infrastructure and Services, Spanish Geological Survey (IGME), Calera, 1, 28760 Tres Cantos, Madrid, Spain
[e] Department of Mineralogy and Petrology, University of the Basque Country (UPV/EHU), Apdo 644, Bilbao, Spain
[f] Faculty of Geological Sciences, Universidad Complutense de Madrid, 28040 Madrid, Spain
[g] Department of Ecology, University of Granada, Campus Fuentenueva, 18071 Granada, Spain
[h] Estación Experimental de Aula Dei (EEAD-CSIC), Avda Montañana 1005, 50059 Zaragoza, Spain

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ABSTRACT

Oxygen depletion (temporal or permanent) in freshwater ecosystems is a widespread and globally important environmental problem. However, the factors behind increased hypolimnetic anoxia in lakes and reservoirs are often diverse and may involve processes at different spatial and temporal scales. Here, we evaluate the combined effects of different anthropogenic pressures on the oxygen dynamics and water chemistry of Lake Enol, an emblematic mountain lake in *Picos de Europa* National Park (NW Spain). A multidisciplinary study conducted over a period of four years (2013–2016) indicates that the extent and duration of hypolimnetic anoxia has increased dramatically in recent years. The extent and duration of hypolimnetic anoxia is typical of meso-eutrophic systems, in contrast with the internal productivity of the lake, which remains oligo-mesotrophic and phosphorus-limited. This apparent contradiction is ascribed to the combination of different external pressures in the catchment, which have increased the input of allochthonous organic matter in recent times through enhanced erosion and sediment transport. The most important among these pressures appears to be cattle grazing, which affects not only the import of carbon and nutrients, but also the lake microbiology. The contribution of clear-cutting, runoff channelling, and tourism is comparatively less significant. The cumulative effects of these local human impacts are not only affecting the lake metabolism, but also the import of sulfate, nitrate- and ammonium-nitrogen, and metals (Zn). However, these local factors alone cannot explain entirely the observed oxygen deficit. Climatic factors (e.g., warmer and drier spring and autumn seasons) are also reducing oxygen levels in deep waters through a longer and increasingly steep thermal stratification. Global warming may indirectly increase anoxia in many other mountain lakes in the near future.

1. Introduction

Mountain lakes are greatly vulnerable to climatic changes and anthropogenic pressures (Parker et al., 2008). Their environmental management relies on a delicate equilibrium between resource exploitation and ecosystem preservation (Debarbieux et al., 2014). This turns critical when applied to national parks, which represent a reference in the conservation of natural heritage and biodiversity. A major important threat which environmental managers of aquatic systems must face is that of hypoxia/anoxia, which usually occurs as a natural response to increased organic and/or nutrient loadings (Nürnberg, 2004), but which is also affected by current climatic trends (e.g., Sahoo et al., 2013; North et al., 2014). The study of lakes in these areas is crucial to understand the effects of human impacts, and provides the basis for their conservation and management (Lotter and Psenner, 2004). In lakes affected by combined pressures at different spatial and temporal scales, a precise evaluation of cause/effect relationships can only be achieved through multidisciplinary research including ecology, hydrology, limnology, geochemistry or soil science, among others.

* Corresponding author.
E-mail address: j.sanchez@igme.es (J. Sánchez-España).

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In this study, we evaluate the influence of different environmental pressures on the water quality and oxygen dynamics of a mountain lake situated in a national park. Despite being in a protected natural area, a number of anthropogenic activities in the lake surroundings (including cattle grazing, clear-cutting, alterations of the drainage network and intensive tourism) could be altering the ecological equilibrium of the lake, though the particular effects of these different stressors are presently unknown. We conducted a geochemical, limnological and microbiological study over a period of four years (2013–2016). The objectives of our study were: (i) to obtain an accurate *picture* of the biogeochemical dynamics of the lake (including the extent and duration of anoxia, phytoplankton dynamics and bacterial activity, nutrient and carbon concentration, trophic state) to identify possible indicators of anthropogenic environmental pressures; (ii) to recognize spatial chemical trends in basin soils and lake sediments which may help detect major fluxes of organic matter, nutrients or pollutants (e.g., metals) affecting the lake water quality and/or oxygen dynamics; and (iii) to evaluate the effects of climatic factors on the vertical distribution of dissolved oxygen in the water column by comparing inter-annual variations of lake stratification. These three objectives provide the structural sub-headings used in the Results and Discussion sections.

The results and discussion related to local anthropogenic pressures at the basin scale can be valuable for the future conservation and management of this lake and many others situated in similar mountain areas. The climatic effects on the extent of hypolimnetic anoxia, although based on local inter-annual oscillations of temperature and precipitation, may be useful to enlarge our understanding of the impact of climate change (e.g., global warming) on oxygen dynamics in mountain lakes worldwide.

2. Study site

We centred our study in Lake Enol (1070 m a.s.l.; *Picos de Europa* National Park, NW Spain), which represents a paradigm of environmental protection in Spain. Along with the adjacent Lake Ercina, this lake forms an emblematic mountain landscape known as *Covadonga Lakes* (Fig. 1a) which was the first National Park declared in Spain in 1918. At present, it is the second most visited protected natural area in the nation with 630,000 visitors per year (OAPN, 2015). However, this lake is being subject to different human pressures which threaten its future conservation (Fig. S1 in electronic Supplementary material). The number of tourists increased significantly in the 1980s, and cattle grazing has been also intensified, with the number of cows amplified by a factor of 3 since 2000 (OAPN, *unpubl.*). In addition, shrub clear-cutting is a common practice in the area, and alterations in the drainage network (e.g., construction of roads and hiking trails around the lake) have increased soil erosion rates in recent times (Rodríguez et al., 2016). The lake has a surface area of 133,000 m^2, volume of 10^6 m^3 and maximum depth of 22 m (Fig. 1b), and is regulated by an outlet draining to the East.

3. Materials and methods

3.1. Hydrogeochemical profiling and T/DO recording

Geochemical profiles and sampling of waters and phytoplankton were conducted on a bimonthly basis from spring (May) to fall (November) in 2013, 2014 and 2015, with additional data for November 2016. Physico-chemical parameters (including temperature –T–, specific conductance –SpC–, pH, redox potential –ORP–, dissolved oxygen –DO–, total pressure of dissolved gases –TDG–, photosynthetically active radiation –PAR–, and chlorophyll-a –chl-a–) were taken with multi-parametric datasondes (Hatch®, YSI). ORP values are given as relative to the standard hydrogen electrode (Eh) after temperature correction. PAR was measured with a LI-COR sensor in the wavelength range 400–1100 nm. The chl-a sensor measures relative fluorescence signal and provides an estimate of chl-a concentration. The DO sensor measures oxygen concentration by luminescence (precision = 0.0 1 mg/L O_2).

T and DO data loggers (HOBO, Onset Computer Corporation) were installed at different depths in a mooring line (Fig. 1b). Five thermistors (at depths of 1, 5, 10, 15 and 19 m) and a DO logger (at 19 m) recorded T and DO changes during the studied period at intervals of 1 h. Measuring ranges and resolution were 0–40 °C and 0.1 °C for the thermistors and 0–30 mg/L and 0.02 mg/L for the DO logger.

3.2. Sampling of waters and sediments

Water samples for chemical analyses of major ion and nutrient concentration and for microbiological analyses (phytoplankton, bacteria) were taken from different depths (at 0.5, 5, 10, 15 and 18 m below the lake surface) with a Van Dorn® sampling bottle (KC Denmark). Phytoplankton samples (3 replicates × 125 mL) were immediately fixed in 4% acetic Lugol's solution. Samples for chemical analyses were filtered on site with 0.45 μm nitrocellulose membrane filters (Millipore®), stored in polyethylene bottles (125–250 mL), acidified with HNO_3 (metals), and cool-preserved during transport.

Sediment cores (15–50 cm in length) were taken with a gravity corer (UWITEC) from shallow (4 m depth) to deep (21 m) areas for chemical and microbiological analyses (Fig. 1b). The cores were kept closed under dark conditions at 4 °C. The upper 10 cm of these cores were split lengthwise, sampled at intervals of 1–2 cm and later analysed in the laboratory by the methods described below.

Two sediment traps (HYDRO-BIOS) were installed near the lake centre at depths of 5 and 19 m to collect suspended particulate matter (SPM). These traps were sampled every two months; the collecting bottles (volume = 250 mL) were transported to the laboratory for chemical and mineralogical analyses of SPM and calculation of sedimentation rates.

3.3. Microbiological analyses

Phytoplankton (density, biomass and species composition) were studied at the University of Granada. A volume of 50 mL of each sample were allowed to settle in counting chambers before examination with an inverted microscope. Identification was carried out to species level where possible. A minimum of 400 cells or setting units (colonies, filaments) were counted in order to get reliable abundance estimates. Individual biovolume was calculated according to fitted geometric forms (Hillebrand et al., 1999), taking linear measures of 30 individuals. Biovolume data were converted to biomass (in C) using regressions from Menden-Deuer and Lessard (2000).

The presence of enteric bacteria at different depths (including total coliforms, fecal coliforms and total enterococcus) was analysed in a certified laboratory (Ensayos Microbiológicos, S.L.). Additionally, semiquantitative analyses of bacterial metabolisms (iron-related and sulfate-reducing bacteria –IRB, SRB–, in addition to other bacterial groups including enteric bacteria and *Pseudomonads*) were carried out in waters and sediments by commercial biological activity reaction tests (BART™) from Hach (DBI, 2004). These tests were conducted at room conditions and required between 6 and 10 days for each test.