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RESEARCH ARTICLE

Trophic status and sedimentation rate as effective indicators of condition in alpine lake restoration

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Abstract

Introduction: Oligotrophic alpine lakes are highly vulnerable to eutrophication and shoreline erosion under intense recreational and livestock pressure. Lake Peñalara serves as a case study to evaluate ecotourism impacts on mountain lakes and their recovery following the implementation of management and restoration measures.
Objectives: This study aims to evaluate long-term changes in eutrophication and soil erosion in response to targeted management actions, offering guidance for similarly impacted high mountain lakes.
Methods: We applied change-point analysis to identify distinct recovery phases in trophic indicators (total phosphorus [TP], chlorophyll *a* [Chl], and Secchi transparency) and sedimentation rate (as a proxy for catchment erosion); and trend analysis to detect longer-term tendencies.
Results: Two distinct recovery phases were identified for both eutrophication and shoreline erosion. Eutrophication recovery began with a 5-year period marked by a sharp decline in TP and Chl, along with increased water transparency. This was followed by a 20-year phase with moderate-to-low nutrient levels; although a slight long-term increasing trend was observed. Shoreline erosion recovery also occurred in two phases: an initial 8-year period of decline, followed by a sustained phase of low erosion rates.
Conclusions: Recovery was achieved by reducing external nutrient inputs and trampling through visitor management (bans on swimming and camping, park rangers, limits on prolonged stays, and defined trails), by revegetating eroded areas, and by restricting livestock access with perimeter fencing. These results underscore the effectiveness of targeted activity management over merely reducing visitor numbers and highlight the need for decades-long monitoring to capture ecosystem recovery dynamics.
Implications for Practice: Trophic status and sedimentation rate are effective indicators of recreational impact and restoration effectiveness. In small alpine lakes, eutrophication can recover relatively quickly following management interventions, whereas shoreline erosion typically requires active restoration measures, such as revegetation, to achieve long-term stability. High visitor numbers can be compatible with the conservation of aquatic ecosystems if nutrient inputs and trampling are effectively managed. Key measures include: visitor regulation, park rangers, clear signage, brochures, marked trails, alternative routes, and physical barriers like fencing or electric shepherds. Long-term monitoring is essential to detect ecosystem trends and recovery trajectories.

Key words: erosion, eutrophication, livestock, long-term, management, sedimentation rate, trampling, visitor

Introduction

Between 64% and 71% of global wetlands have disappeared since 1900 AD (Davidson 2014), and most of the remaining ones are degraded to varying degrees (Zedler & Kercher 2005; Dudgeon et al. 2006). In this context of wetland loss and degradation, there is increasing social pressure to reverse this trend through natural recovery or active restoration actions (Geist & Hawkins 2016; Penca & Tănăsescu 2024), with the aim of halting biodiversity loss (Zedler & Kercher 2005) and restoring ecological functions and ecosystem services (Geist & Hawkins 2016; Grizzetti et al. 2016). However, there is a deficit of long-term studies documenting ecosystem recovery and the time required to reach a recovered status (Jones et al. 2018; Abell et al. 2022). Long-term monitoring of restoration outcomes is highly valuable (Sayer et al. 2010; Hughes et al. 2017), but it often ends prematurely (Nilsson et al. 2016).

In Spanish high mountain lakes, the main local anthropogenic impacts are recreational pressure, damming, and hydropower exploitation (Fernández Sañudo et al. 2000), as well as livestock presence (Sánchez-España et al. 2017) and introduced fish (Miro & Ventura 2020). A paradigmatic case is Lake Peñalara (Sierra de Guadarrama, Spanish Central System), which was heavily impacted by unsustainable recreational use during the

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1970s and 1980s, leading to lake eutrophication, shoreline erosion, and fish introduction (Toro & Granados 2002; Toro et al. 2006). Since 1990, a series of management and restoration actions have been implemented to mitigate eutrophication and control shoreline erosion and eradicate the introduced fish, accompanied by a monthly limnological monitoring program that has provided valuable long-term data on the lake's recovery.

The increase in trophic status in mountain lakes due to recreational pressure is relatively well documented (Bigler et al. 2007; Ravnikar et al. 2016; Senetra et al. 2020), often exacerbated by livestock presence (Sánchez-España et al. 2019) or fish introductions (Brancelj et al. 2000). However, there is limited information on the long-term recovery of alpine lakes from eutrophication following restoration actions, especially for lakes of the Mediterranean region.

Trampling by both hikers (Martin & Butler 2017) and cattle (Hiltbrunner et al. 2012) is one of the main drivers of soil erosion and vegetation loss in mountain environments. Nevertheless, both the implementation of management strategies (mitigation, rehabilitation, and restoration) and advances in understanding the relationship between trampling, erosion, and sedimentation remain scarce (Salesa & Cerdà 2020).

Therefore, in this study, we present a rare case of long-term (decadal scale) recovery of a Mediterranean high mountain lake, with the following objectives: (1) to describe the response of eutrophication- and soil erosion-related variables to the management and restoration measures implemented to mitigate nutrient inputs and trampling associated with unsustainable recreational and livestock use, and (2) to highlight key lessons for visitor management in other high mountain lakes facing high tourist pressure.

Methods

Study Area

Lake Peñalara. Lake Peñalara is a Mediterranean high mountain lake located at 2018 m a.s.l. in the Peñalara Massif (40.83985°N, 3.957267°W; Sierra de Guadarrama, Spanish Central Range). It has a surface area of 6770 m^2 and a maximum depth of approximately 5 m. The lake's catchment area covers 0.476 km^2 and consists of a glacial cirque with steep rocky areas and scree (63.2%), mountain scrub (29.5%), psicroxerophilous grasslands (6.3%), and wet grasslands or small peatland areas (1%). The soils are classified as ranker type (Lithic Cryorthents).

The mean annual precipitation at Puerto de Navacerrada (1984 m a.s.l.), the closest official meteorological station located 6.9 km from the lake, is 1223 mm, with an average of 111 rainy days and 71 snowy days per year. The mean maximum temperature is 22.4°C in July, while the mean minimum temperature is −3.2°C in January (AEMET 2021). Temperature trends in the Sierra de Guadarrama show a progressive warming since the mid-twentieth century, which has intensified during the 2000–2018 period (Vegas-Cañas et al. 2020). No significant trend in annual precipitation has been detected.

According to Lewis Jr (1983), it can be classified as a cold discontinuous polymictic lake. The lake remains ice-covered for an average of 4 months/year; although since 1990, the number of ice-covered days has decreased at a rate of 1 day/year, and years with intermittent ice cover have become more frequent. The mean water column temperature increased by 0.6°C per decade between 2005 and 2022. The mean annual water residence time is 12.2 days, extending to 118 during the summer season. The highest water renewal rate occurs during snowmelt and autumn (Granados 2021).

Unsustainable Recreational Use and Restoration. The Peñalara Massif is a well-known natural area located approximately 50 km from Madrid, with good accessibility by car and train. Today, it is part of the Sierra de Guadarrama National Park. According to a paleolimnological study (Toro & Granados 2002), between circa 1920 and 1970, the lake was oligotrophic and exhibited a low sedimentation rate (SR), comparable to other high mountain lakes. In the early 1970s, a growing influx of visitors led to the massification of the area and unsustainable recreational use. During this period, the non-native Brook trout (*Salvelinus fontinalis*) was introduced into Lake Peñalara for recreational fishing (Toro & Granados 2002; Toro et al. 2006), while visitors frequently swam in the lake and camped in the surrounding area. By the late 1980s, the lake showed clear signs of eutrophication, such as summer phytoplankton blooms and reduced water transparency. Additionally, trampling by visitors and the regular presence of livestock triggered severe erosion along a large section of the lake's shoreline (Fig. 1A), where the soil became compacted and devoid of vegetation (Fig. 1B). There have never been any known point sources of pollution in the lake.

Following its designation as a protected area in 1990, a restoration plan was implemented with several management measures. To reduce eutrophication, swimming in the lake and camping in the massif were banned, and a ranger service was created to enforce regulations. Large and prolonged sedentary stays within the lake catchment were discouraged. Access was limited to a maximum of 50 people per day in organized groups (e.g. school groups) and no more than 250 people simultaneously within the catchment. Visitor movement was restricted to well-marked trails, with wooden walkways installed in the most sensitive areas, such as wet meadows and peat bogs.

In 1995, a single-cable perimeter fence was installed around the lake to prevent visitors from accessing the shoreline, especially in areas with incipient to severe erosion (Fig. 1A). Inside this perimeter, a solar-powered electric fence was set up to keep livestock away from the eroded areas. Small information signs were placed along the outer fence to explain the purpose of the management measure.

Shortly afterwards, a project to re-vegetate the shoreline with Matgrass (*Nardus stricta*) began. Seeds were collected from the surrounding area and grown in a nursery. Several 2-year-old plants were transplanted following a patchy arrangement, resembling small islands in the eroded zone, from which the remaining bare soil was gradually colonized through vegetative growth. Between 1999 and 2002, the non-native Brook trout was eradicated from the lake using gillnet fishing (Toro et al. 2020). The restoration plan was supported by a limnological characterization (Toro & Montes 1993) and a monthly

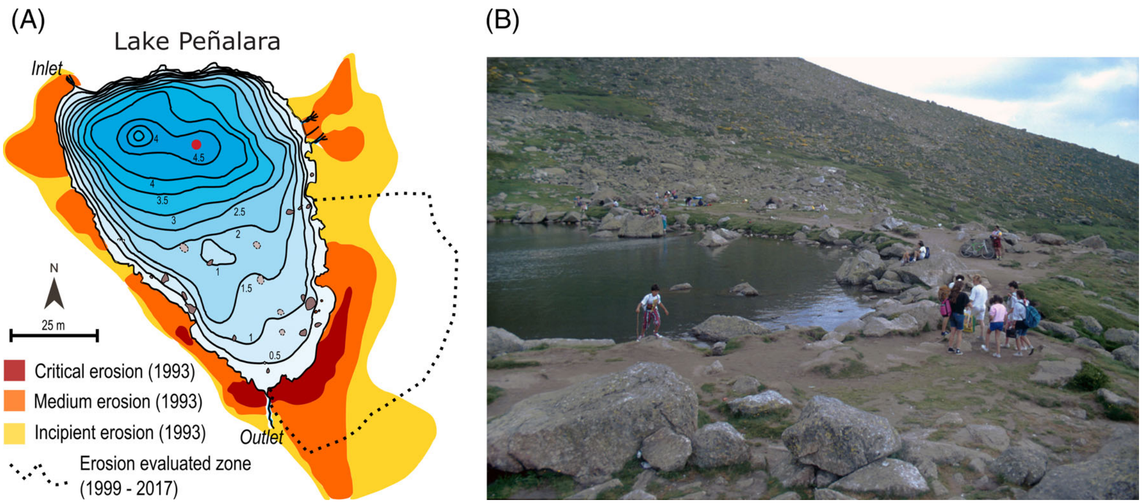

Figure 1. (A) Bathymetry, eroded areas in 1993 and area examined to assess recovery between 1999 and 2017 (modified from Toro & Granados 2002; and Granados et al. 2014). (B) Lake shore erosion and compaction in 1990.

limnological monitoring program that began in 1995. The management measures were generally well accepted by visitors, anglers, and livestock owners, with little opposition due to the area's protected status and availability of alternatives.

This article includes data up to 2017, as a snow avalanche in 2018 altered the lake's morphometry, leading to significant changes in the zooplankton community and SRs (Granados et al. 2025).

Assessment of the Recreational Impact

Visitor data for the Peñalara Massif were obtained from the public use monitoring program of Sierra de Guadarrama National Park. The percentage of revegetation on the eroded shore was determined using the erosion map from 1993 (Toro & Granados 2002) and through the analysis of aerial orthophotos and field surveys conducted between 1999 and 2017 (Granados et al. 2014) over an evaluated area of 3110 m^2 (Fig. 1A).

Lake Parameters

Between 1990 and 1994, an average of five samplings per year was conducted. From July 1995 onwards, monthly sampling was performed, except during extreme weather conditions. Water samples were collected at the deepest point of the lake, either from a boat during summer or using an ice auger in winter. Total phosphorus (TP) was analyzed using the persulfate digestion method (APHA-AWWA-WPCF 1992). Chlorophyll *a* (Chl) concentration was determined spectrophotometrically on an acetone extract following Picazo et al. (2013). Secchi depth was measured only during the ice-free season.

Sediment traps were deployed in October 1996, with the first retrieval in 1997, at the maximum depth zone, positioned 1.3 and 2.6 m above the sediment. However, in this study, only the data from the deepest trap were used. Each trap consisted of three replicates composed of transparent methacrylate tubes enclosed in opaque polyvinyl chloride (PVC) tubes measuring 58 cm in length and 55 mm in internal diameter (23.75 cm^2 collecting area). The sedimented material was recovered monthly during the ice-free season. However, adverse weather conditions occasionally prevented sampling, resulting in a longer exposure period for the trap. During winter, the trap remained under the ice cover (circa 4 months/year) until it was retrieved after thawing.

Dry weight (DW) and loss on ignition (LOI) were determined by standard sequential weighing techniques (Heiri et al. 2001) after drying (105°C, 24 hours) and ignition (550°C, 2 hours) of the settled material in the sediment traps. SR in $g\ m^{-2}\ day^{-1}$ was calculated by dividing DW by the collecting surface area and the number of days the trap was deployed. LOI, a widely used proxy for organic matter content (Heiri et al. 2001), was expressed as a percentage of DW (percentage of organic matter [%OM]).

Data Analysis

Multiple time series related to eutrophication and sedimentation were analyzed to detect changes in trend or seasonality using a Bayesian approach (Bayesian Estimator of Abrupt change, Seasonal change, and Trend [BEAST]) with the *Rbeast* R package (Zhao et al. 2019). For each time series, BEAST estimates the probability of having a given number of change points, the probability of each detected change point, and the corresponding credible intervals. This allows for the identification of tipping points across multiple series. The time series were normalized prior to the analysis. The models were fitted to annual data (eutrophication series) without seasonality or to monthly data (sedimentation series) with a harmonic seasonal structure assuming an annual cycle. Change-point detection was limited to a maximum of two in both trend and seasonality, with a minimum separation of 4 years and located at least 4 years from the beginning and end of the time series. The model was estimated using Markov Chain Monte Carlo (MCMC) with 8000 samples retained after 2000 burn-in iterations and a 95% credible interval level.

In the case where a tipping point was detected with some degree of uncertainty, its presence was further evaluated by testing for changes in the mean and variance of individual time series using the *changepoint* R package (Killick & Eckley 2014). For single time series, change-point detection was performed under the assumption of an exponential distribution, identified as the best-fitting model using the *fitdistrplus* R package (Delignette-Muller & Dutang 2015). A minimum segment length of 4 years was enforced, and the Hannan–Quinn criterion was used as the penalty function when allowing for multiple change points. However, since only one change point was detected, an asymptotic penalty was subsequently applied to control the type I error rate, ensuring a 0.05 significance level for rejecting the null hypothesis of no change.

Trend analyses of TP, Chl, SR, and %OM were calculated as Theil-Sen slope with the *OpenAir* R package (Carslaw & Ropkins 2012), that takes account of auto-correlated data using block bootstrap simulations. The periodicity of SR and %OM was analyzed by calculating a periodogram using the *lomb* R package (Ruf 1999) for non-uniformly spaced time series.

Results

Figure 2 shows the eutrophication and erosion variables from 1990 to 2017, along with visitor data for the Peñalara Massif, the percentage of vegetation cover on the eroded shore, and some relevant dates of management and restoration actions. The mean annual number of visitors (Fig. 2A) was 106,400 visitors/year between 1997 and 1999 and rose to 148,100 visitors/year from 2004 to 2017, with values ranging between 131,800 and 163,900 visitors/year.

Taking as a reference a portion of the shore in 1993 (40.4% bare soil), the erosion process intensified until 1999 (67.4% bare soil), despite the implementation of access limitation measures (shore fence, 1995) and revegetation efforts (1997). Between 1999 and 2001, there was hardly any improvement in the extent of bare soil (Δ 1.1% recovery/year), but between 2001 and 2004, the recovery process began to accelerate (Δ 4.5% recovery/year), followed by an average of Δ 7.0% recovery/year between 2004 and 2011, when bare soil had decreased to just 5%. Full recovery (Matgrass, *Nardus stricta*, with no bare soil) was achieved by 2017.

Long-Term Dynamics of the Trophic Status in Lake Peñalara

The summer maximum of TP and Chl, along with the Secchi disk depth minimum, was used to identify a potential tipping point based on these three time series related to the trophic status (Fig. 2C–E). Table 1 shows, for each time series, the probability of having a given number of change points, the probability of each detected change point, and the corresponding credible intervals at the 95% confidence level. The BEAST algorithm detected with complete certainty a change point in 1995 for both maximum TP and Chl, although Secchi depth showed only a slight indication that a change point might exist in that year.

Consequently, two distinct phases were identified: a high trophic state phase up to 1994 and a medium-to-low trophic state phase thereafter. In 1990, the mean annual TP concentration was 222 μg P/L (with a summer maximum of 443 μg P/L), the mean annual Chl concentration was 28.2 μg/L (with a summer maximum of 68.1 μg/L), and the minimum Secchi disk depth was 1.7 m. During this first phase of high trophic state (1990–1994), the mean TP concentration was 110 μg P/L and the mean Chl concentration was 12.7 μg/L. A rapid decline in TP and Chl concentrations, along with a moderate increase in Secchi disk transparency, was observed during this phase (1990–1994), associated with management actions such as limiting the number of simultaneous visitors, prohibiting swimming and camping, and implementing a ranger service.

Between 1995 and 2017, the mean TP concentration was 18.2 μg P/L and the mean Chl concentration was 1.73 μg/L, with a mean minimum Secchi disk depth of 2.8 m. From 1995 onwards, conditions became much more stable compared to the previous phase (1990–1994), with substantially lower values for these two variables. However, despite this apparent stabilization, significant long-term trends indicative of a gradual increase in trophic status were observed. These included significant increases in the annual mean TP concentration ($p < 0.05$, Δ $+0.89$ μg P L^{-1} yr^{-1}) and the annual mean Chl concentration ($p < 0.05$, Δ $+0.04$ μg L^{-1} yr^{-1}).

Long-Term Dynamics of Sedimentation in Lake Peñalara

Apart from eutrophication, soil erosion abatement was one of the main objectives of the management and restoration measures. To assess it, we analyzed the daily SR (Fig. 2F) and the %OM in the settled material (Fig. 2G) collected in the sediment trap located at the bottom of the lake. The mean coefficient of variation between the three replicates of each sediment trap was 11.7% for DW and 6.2% for LOI.

The SR exhibited a clear pattern, with maximum sedimentation peaks during the summer months and minimum values during the ice-covered season. This periodicity was highly significant ($p < 0.0001$), with the maximum peak in the periodogram at 359.1 days. Similarly, the %OM followed an annual pattern, with lower values in summer. This periodicity was also highly significant ($p < 0.0001$), with the maximum peak in the periodogram at 357.7 days, both fitting with a yearly pattern.

Based on the monthly SR and %OM time series, the BEAST algorithm identified a 96.5% probability of a change point in the seasonality of SR in 2003, that coincides also with a probable change in SR trend and %OM seasonality; although in the later cases it is most probable that there are no credible change points (Table 1). Therefore, a second, non-Bayesian change point analysis was performed to confirm this 2003 tipping point, focusing on the annual maximum sediment flux. This analysis divided the series into two phases: 1997–2003 and 2004–2017. Consequently, the period 2004–2017 was considered a single phase of low sedimentation, with SRs approximately one order of magnitude lower than in the previous phase (1997–2003).

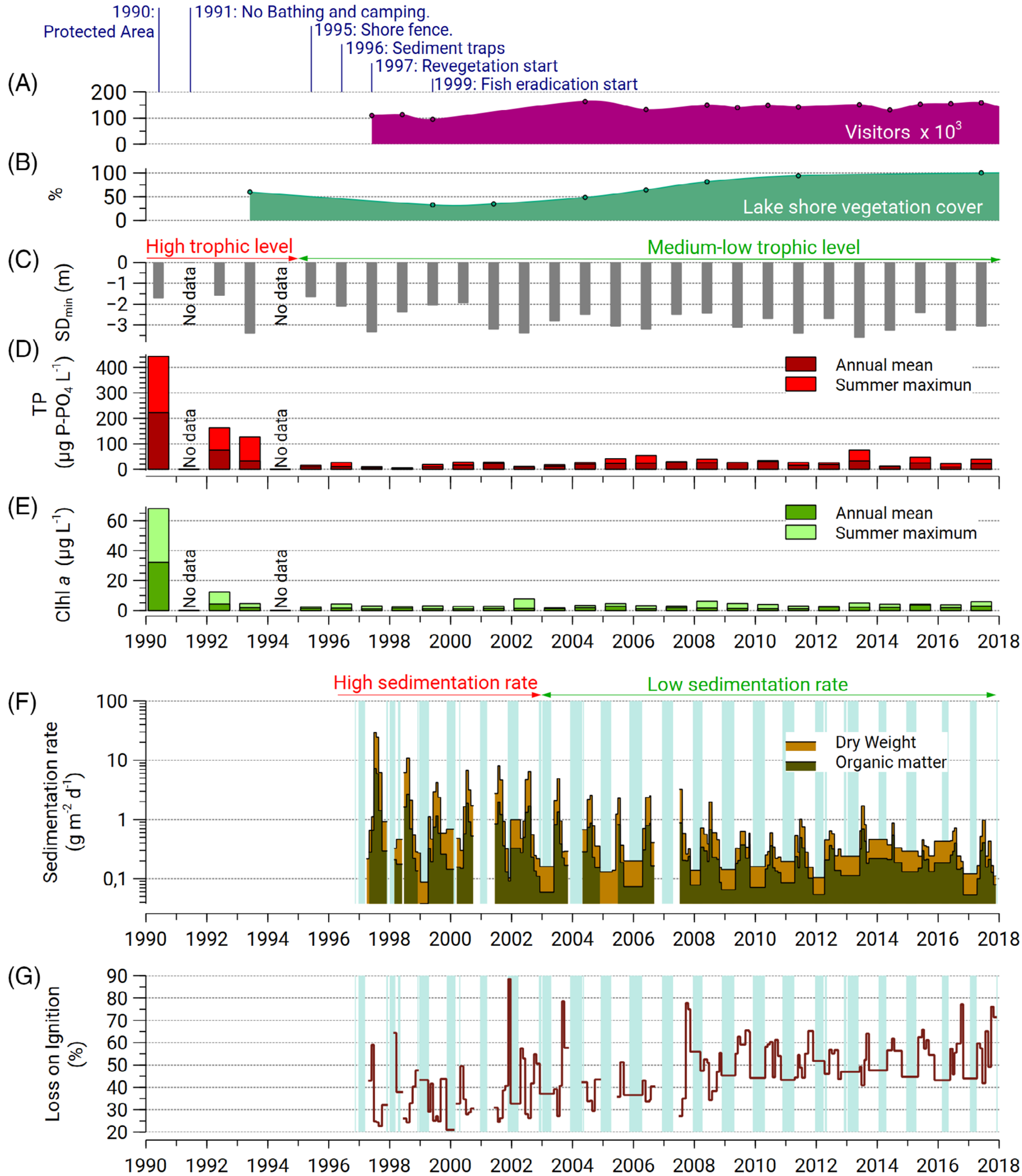

Figure 2. Eutrophication and erosion variables between 1990 and 2017. (A) Thousands of visitors to the Peñalara massif. (B) Lake shore vegetation cover percentage measured in a 3110 m^2 plot. (C) Minimum annual Secchi disk depth. (D) Annual mean and maximum total phosphorus concentration in summer. (E) Annual mean and maximum summer chlorophyll a concentration. (F) Daily sedimentation rate (dry weight and organic matter) in the sediment trap at the bottom of the lake. (G) Loss on ignition of the deposited material in the sediment trap. Blue bands indicate periods with ice cover in (F) and (G).

During the high sedimentation phase (1997–2003), the mean annual maximum SR was 10.06 g m^{-2} day^{-1}, and %OM had a weighted mean of 39.7% (range: 22.7–86.7%). However, a sharp decline in SR was observed over this phase, starting in the summer of 1997 with a maximum SR of 29.28 g m^{-2} day^{-1} and dropping to 4.90 g m^{-2} day^{-1} by 2003. During the low sedimentation phase (2004–2017), the mean annual maximum SR was 1.44 g m^{-2} day^{-1}, and %OM had a weighted mean of 50.9% (range: 27.2–76.5%); this is much lower sedimentation with a relatively higher organic matter content. It should be noted that, in absolute terms, the input of organic matter to the sediment was higher during the high sedimentation phase (1997–2003) than during the low sedimentation phase (mean annual maximum OM: 2.56 vs. 0.57 g m^{-2} day^{-1}; Fig. 2F), even

Table 1. Change point analysis results with BEAST algorithm. For each time series: probability of having a given number of change points, the probability of each detected change point, and the corresponding credible intervals at the 95% confidence level.

	Time series	*Change point type*	*Number of change points probability*	*Year change point probability*	*Credible interval (95%)*
Eutrophication	Summer TP max.	Trend	1: 100%	1995: 100%	1994.05–1995.99
	Summer Chl max.	Trend	1: 100%	1995: 100%	1994.05–1995.95
	Summer Secchi min.	Trend	0: 65.3%	2008: 14.0%	2000.66–2012.41
			1: 29.8%	1995: 13.0%	1994.05–2001.42
			2: 4.9%		
Sedimentation	SR	Seasonality	0: 3.6%	2003.54: 96.2%	2003.30–2007.26
			1: 95.7%		
			2: 0.7%		
		Trend	0: 62.6%	2003.54: 30.4%	2003.36–2009.85
			1: 37.1%		
			2: 0.3%		
	%OM	Seasonality	0: 57.5%	2003.88: 39.7%	2003.48–2008.67
			1: 41.2%		
			2: 1.3%		
		Trend	0: 72.6%	2007.79: 26.9%	2006.41–2008.76
			1: 27.3%		
			2: 0.2%		

though its percentage relative to total sedimentation was lower in the former, due to the overall much greater sediment input.

During the low sedimentation phase (2004–2017), a significant long-term trend was detected for SR ($p < 0.01$, Δ $-2.167\mathrm{e}{-09}$ g m^{-2} day^{-1}) and %OM ($p < 0.001$, Δ $+3.582\mathrm{e}{-08}$% day^{-1}); although no significant trends were observed when analyzing the periods 2004–2008 and 2009–2017 separately.

Diagram of Recovery Phases

A conceptual diagram has been developed to synthesize the primary outcomes of the limnological monitoring in relation to catchment activities and the potential ecological processes that may have been at play. Figure 3A shows the initial and final states of the two main phases identified by the change-point analysis regarding eutrophication reversal. The first element (left) represents the initial conditions at the start of the restoration project (1990), while the second element (center) reflects the conditions at the end of the phase characterized by high nutrient levels (1995). Listed below the arrow connecting these two elements are the main events and outcomes that occurred throughout this phase (1990–1994). Similarly, the second element (center) marks the beginning of the moderate-to-low nutrient phase, and the third element (right) shows its final state in 2017. Again, the arrow linking these two stages includes the key events and outcomes that took place during this second phase (1995–2017).

Figure 3B focuses on the reversal of shoreline erosion in Lake Peñalara. The first element (left) represents the typical winter conditions observed throughout the entire study period, regardless of the different phases identified by the change-point analysis. The second (center) and third (right) elements illustrate the conditions observed during the high-erosion phase (1997–2003) and the low erosion phase (2004–2017), respectively.

Discussion

Eutrophication Reversal

The restrictions on recreational use came into effect in 1991, so the phosphorus and chlorophyll concentrations from 1990, along with the Secchi disk depth, can be considered as the baseline levels when the lake was affected by swimming, camping, and livestock presence. By 1992, phosphorus concentration had decreased by 63% and Chl by 82%. The change point analysis indicates that relatively stable conditions were reached within 5 years after the management and restoration actions were set, decreasing to just around 5% of the initial phosphorus and Chl concentrations. Summer water transparency (Secchi disk) also improved from 1995 onwards. These results are consistent with those observed in other lakes subjected to reduced nutrient inputs (Jeppesen et al. 2005; Tiberti et al. 2019a), showing even greater and faster recovery. This high resilience (quick recovery) is largely driven by the lake's high renewal rate, which allows an annual "resetting" of the water body's hydrochemistry.

During the years when the trophic status dropped substantially (1990–1995), the number of visitors did not change appreciably, although permitted activities were restricted. Livestock numbers around the lake also remained stable, although their access was partially limited through active management by park rangers. Extended camping and using the lake for swimming or washing cooking pots was prohibited. In addition, control measures for daily visitors, such as banning swimming, reducing sedentary stays in the catchment, and limiting organized groups, effectively reduced the external nutrient load reaching the lake,

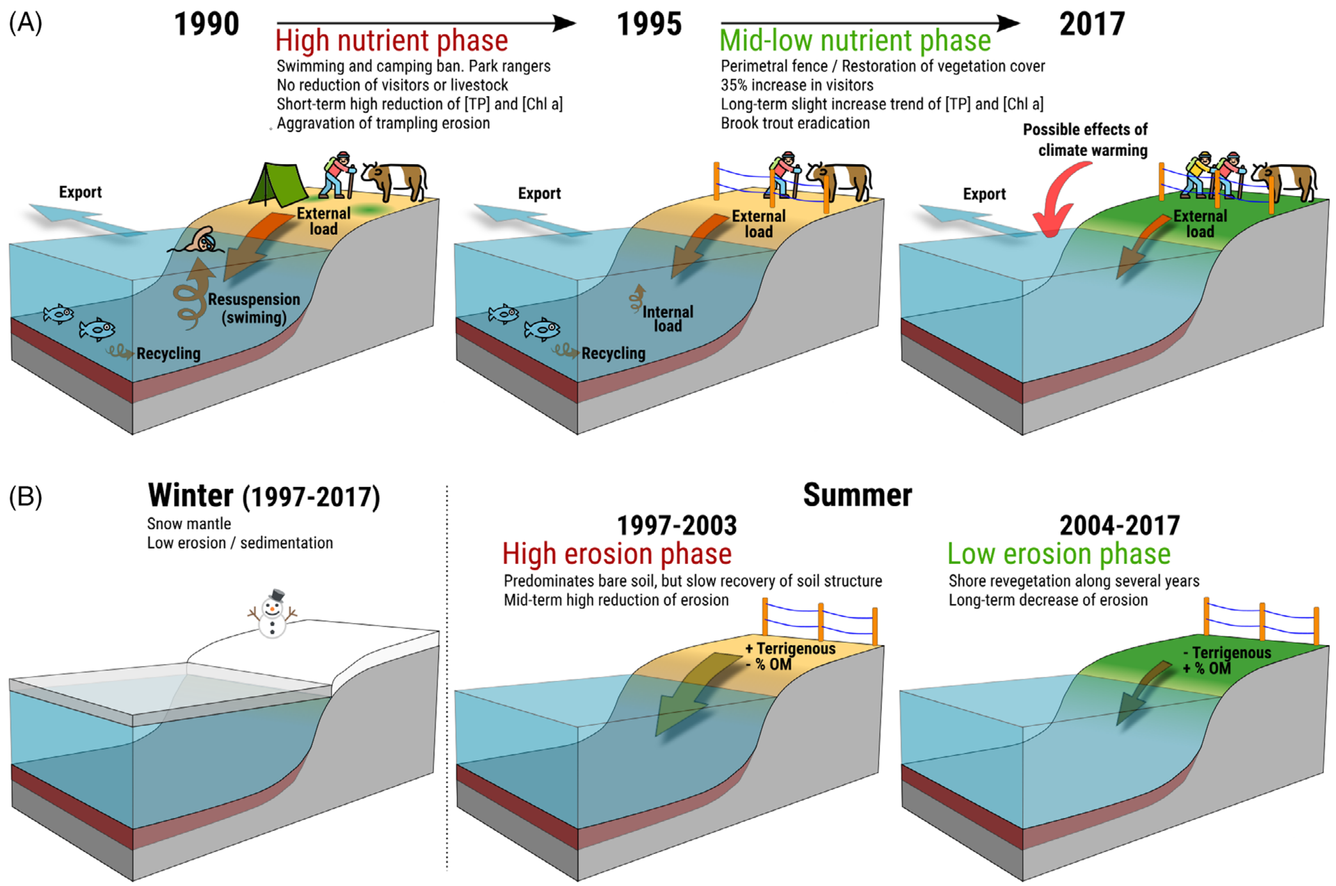

Figure 3. Conceptual diagram summarizing the main phases, key management actions, and ecological responses observed during the limnological monitoring of Lake Peñalara. (A) Phases of eutrophication reversal based on change-point analysis, showing initial and final conditions and major events within each phase. (B) Phases of shoreline erosion and recovery, illustrating seasonal conditions and long-term changes associated with management and restoration efforts.

which is essential to reverse eutrophication (Abell et al. 2022). However, shoreline degradation due to trampling persisted, as the marked trail ended at the lakeshore. The 4-year time lag to reach a stable state in a lake with a high renewal rate suggests that shoreline erosion acted as an external nutrient source at least until revegetation work began in 1997. Cattle contribute to nutrient input through fecal deposits, an effect intensified in eroded catchments due to reduced vegetation retention (Sánchez-España et al. 2017). Further, the sediment may have been acting as an internal nutrient load after a long period of eutrophication (Søndergaard et al. 2013). In fact, sediment resuspension caused by swimmers and livestock in the littoral zone likely played a major role in the summer phytoplankton blooms between 1980 and 1990 (Toro & Granados 2002).

Between 1995 and 2017, the number of visitors to the lake increased by 35%; but access to the shoreline was restricted by a perimeter fence, and swimming and camping were prohibited. During this phase, a moderate increasing trend in TP and Chl was observed. The medium-to-long-term slight increase in trophic status occurred despite the eradication of Brook trout in 2002 (Toro et al. 2020), which should have contributed to the oligotrophication process (Abell et al. 2022). In contrast, climate change is known to stimulate eutrophication through air and water warming, which increases nutrient uptake and primary production (Oleksy et al. 2020). It can also act indirectly by enhancing nutrient release from catchment soils (Kopáček et al. 2024; Scholz & Brahney 2022) and increasing atmospheric dust deposition (Tsugeki et al. 2012). In the Sierra de Guadarrama region, the mean summer air temperature increased by 1.42°C between the 1990s and the 2010s, with an accelerated warming trend during summer and autumn in 2000–2020 compared to previous decades (Vegas-Cañas et al. 2020). In the Iberian Peninsula, the frequency and intensity of African dust outbreaks are also increasing (Salvador et al. 2022; Jimenez et al. 2018). Although no significant trend in precipitation has been observed at the nearby weather station, some studies suggest an increase in dry periods since the 1990s (Ruiz-Labourdette et al. 2014), which may also lead to major changes in lake primary production (Catalan et al. 2024). The fact that the upward trend in trophic status became evident only after reducing local anthropogenic pressure suggests that, in this case, the combined effect of climate change and atmospheric deposition could play a dominant role after restoration measures. This contrasts with other mountain lakes where tourism impact can mask the effects of climate change (Frossard et al. 2023). Our results suggest that managing visitor activities is far more important than simply reducing the number of visitors when it comes to limiting nutrient inputs.

No hydrochemical data are available prior to 1990, but the paleolimnological study conducted as part of the restoration project (Toro et al. 2006) revealed an increase in organic matter sedimentation from the early 1980s to 1990, concurrent with a sharp rise in visitor numbers and the proliferation of diatoms indicative of a higher trophic state (Toro & Granados 2002). Therefore, the reversal of eutrophication in Lake Peñalara suggests a recovery toward its original condition. However, with the available data, it is not possible to determine whether the predisturbance state was fully restored or whether internal nutrient loading and climate change have prevented complete recovery.

Soil Erosion and Sedimentation Response to Restoration

Soil is vulnerable to erosion where it is exposed to water or wind flow and where topographic conditions (steep slopes), soil conditions (soil cohesion) or anthropogenic uses (compaction, mud formation, and vegetation loss) multiply the effect of the erosive agent or decrease soil cohesion (Harden 2001). Terrigenous input from the catchment is more likely to occur when soil is more vulnerable to erosion. In months when the catchment is partially or completely covered by snow, the soil is sheltered from the erosive force of water or wind (Maruffi et al. 2022), effectively preventing erosion. In contrast, in months when the soil is exposed to rain and wind, there is a complex combination of factors that determines erosion intensity (Holz et al. 2015): previous soil moisture, porosity, soil roughness, texture, soil aggregation, and vegetation cover. Though no data are available for all these variables in Lake Peñalara catchment, the presence of bare, compacted, and partly decapitated soils (i.e. soils missing the first mineral soil layer and seed bank) necessarily implies a greater input of terrigenous materials carried by wind, snowmelt, heavy rains, or summer storms. In the summer season, when residence time increases, sedimentation of these transported materials into the lake is enhanced (Gardner 1980).

The change-point analysis discriminated a high sedimentation phase (1997–2003) and a low sedimentation phase (2004–2017) on an interannual scale. In 1993, only 59.6% of the area was covered by vegetation (Matgrass, *Nardus stricta*), decreasing to 32.6% by 1999. After the implementation of management and restoration measures, vegetation cover in the eroded area recovered to nearly 50% by 2004 and reached 95% by 2011. Thus, the high sedimentation phase coincided with the period in which erosion was still advancing or relatively stabilized, while the low sedimentation phase coincided with a period of accelerated vegetation recovery. Therefore, these phases of high and low sedimentation correspond to periods of high and low erosion, respectively.

The presence of bare soil is a critical factor for higher erodibility (Gyssels et al. 2005), whereas compacted soils facilitate water erosion (Bodoque et al. 2017; Martin & Butler 2017). During the first phase after access limitations (1997–2003) the bare soil coverage did not change substantially, although sedimentation showed a clear downward trend. Exclusion of visitor trampling increases saturation capacity, permeability, total porosity, and organic matter, while bulk density decreases (Sutherland et al. 2001; Özcan et al. 2013; Ramos-Scharrón et al. 2014). In this sense, visitor and livestock presence is equally important. By limiting access to the eroded areas (interrupted since 1996), compaction and mudding were reduced, allowing the soil to begin an incipient recovery of its structure even though it was not yet colonized by vegetation. In addition, the surrounding vegetation in areas where it had not previously disappeared was also improved by the absence of trampling and grazing pressure from livestock. Regarding the low erosion phase (2004–2017), the long-term trend toward lower sedimentation corresponds to the progressive improvement of the (still) bare soil surface and the improvement of erodibility conditions following the cessation of trampling along the lake shore.

The %OM confirms the above interpretation of sedimentation dynamics: on average it was significantly lower in the high sedimentation/erosion phase than in the low sedimentation/erosion phase. The organic matter content in bare or decapitated soils is usually much lower (Davidson et al. 2001), and consequently materials from these areas must necessarily have a lower %OM. On the other hand, after corrective management, the %OM showed a significant increase following the expansion of vegetation cover and the improvement of soil condition (Peng et al. 2021). However, it should be noted that the total input of organic matter (not as a percentage of the sedimented material) is higher under high-erosion conditions than under low erosion conditions. Although no data are available for total input prior to 1996, it is reasonable to assume that it was relatively high when the soil was bare, and therefore likely contributed to slowing the recovery of the trophic state.

Erosion control occurred in a context of a 39.2% increase in the average number of visitors when the periods 1997–1999 and 2004–2017 are compared. Therefore, appropriate management of recreational use can make compatible the visit to the protected area with the conservation of the lake. Anyway, the lake's response to erosion control was slower than that to eutrophication control; it was not until more than a decade later (2008) that sedimentation remained below 2 g m^{-2} day^{-1} and higher than 30% organic matter content. This difference reflects the difficulty of vegetation cover recovery in Mediterranean mountain areas, where growth is slow and limited by both the snow cover period and the summer drought. Our study demonstrated that sediment traps are an excellent method for assessing anthropogenic erosion in high mountain lakes subjected to intense public use, especially under conditions of trampling and degradation of vegetation cover.

Extension to Other Mountain Lakes

This case study in a small high mountain lake provides valuable insights that could be applied to other small to medium-sized high mountain lakes with (1) relatively short water retention times and (2) high visitor pressure and/or excessive livestock presence, as often occurs in protected areas (Pastorino et al. 2024). While most studies in aquatic ecosystems span only a few months or years, long-term monitoring over multiple decades has revealed trends and patterns that remain undetectable in short-term studies (Dodds et al. 2012; Rogora et al. 2016; Steingruber et al. 2021). Some of these studies have focused on high mountain lakes, although they typically address long-term changes in water chemistry in relation to climate change and atmospheric pollution rather than local impacts or restoration actions (Tiberti et al. 2019b; Rogora et al. 2020; Schreder et al. 2023).

Our findings are consistent with previous studies that emphasize the need to reduce external nutrient inputs to reverse eutrophication (Jeppesen et al. 2005; Abell et al. 2022). In our case study, we observed that reducing external nutrient-loading sources, such as recreation and livestock, and halting internal nutrient recycling due to fish presence was sufficient to achieve eutrophication reversal in the short to medium term, especially due to the high water renewal rate of the lake (Sochuliaková et al. 2018). However, the longer a lake remains eutrophicated, the more the internal nutrient load can delay recovery after external sources are removed, as the sediment may act as a nutrient reservoir that can be released into the water column due to bottom oxygen depletion during ice-cover periods or sediment resuspension during the open-water season. Therefore, expectations for recovery in other lakes should be adjusted according to their water renewal rates, with faster improvements expected in lakes with higher turnover.

While in Lake Peñalara, reducing allochthonous nutrient inputs was enough to restore the trophic status; reversing the erosion process required active vegetation restoration. This highlights the importance of protecting the soil before erosion occurs, as addressing this issue is much more complex and requires a longer recovery time.

It should be emphasized that the reduction of recreational impacts can be achieved through proper management of visitor activities rather than simply reducing the total number of visitors. While limiting visitor numbers may be desirable in some cases, managing visitor behavior is essential in all situations. Effective measures implemented in Lake Peñalara include: (1) regulations defining permitted and prohibited activities; (2) the presence of park rangers to enforce these regulations; (3) clear communication of rules and protection measures through signs and brochures; (4) a network of marked trails to minimize soil erosion; (5) offering alternative routes crossing less vulnerable areas; (6) physical barriers, such as fencing or electric shepherds, to protect the most sensitive areas from both visitors and livestock, respectively; and (7) continuous limnological monitoring to assess the lake's conservation status.

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