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Are protected areas effective in preserving Alpine stream morphology and biodiversity? A field study in the oldest Italian National Park

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Abstract

Global changes and local pressures related to the exploitation of water resources are significantly reducing streams' biodiversity and threatening their ecological balance. This trend concerns both the lowland rivers flowing in densely populated areas and the mountain headwaters, where the effects of global change are dramatically evident and often accompanied by alterations in river hydro-morphology. From mountains to lowlands, regulation and morphological alterations such as bank reinforcement, water abstractions, dams, and weirs are among the most significant threats for river ecosystems. Protected areas and especially large national parks constitute an effective strategy to face the loss of biodiversity, but little is known about their effectiveness regarding lotic environments. To examine the recent trend in aquatic communities in Alpine protected areas, we carried out biological sampling of benthic communities and hydro-morphological status assessment in 12 high-altitude streams within the oldest Italian National Park, that is, the Gran Paradiso National Park, located in the heart of the Western Alps, and we compared results with a previous survey performed in 2005, by keeping the same experimental design. Our results detected minimal changes in the hydro-morphology of the studied watercourses. Bio-monitoring indices associated with benthic communities likewise did not provide evidence of significant differences. Conversely, diatom communities were more uniform in 2020 compared to 2005, and a slight species turnover occurred over the 15-year interval. Despite this, our findings underline the effectiveness of protected areas for the conservation of running water environments because they limit hydro-morphological alterations, thus increasing the resilience of aquatic communities to climate change.

KEYWORDS

diatoms, hydro-morphology, macroinvertebrates, mountain rivers, RHS, β -diversity

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1 | INTRODUCTION

Habitat loss and fragmentation, over-exploitation of natural resources, climate change, and the introduction of invasive species are listed among the main causes of the current environmental crisis, characterized by a notable extinction rate (Dirzo et al., 2022). Increasing and extending protected areas is one of the most useful and immediate strategies implemented for contrasting this negative trend (Acreman et al., 2020; Lovejoy, 2006). Unfortunately, however, while creating natural parks can be effective in protecting and conserving terrestrial ecosystems and species, the difficulties are often much greater regarding lotic environments and organisms living within them. In fact, because of their longitudinal and dendritic shape, streams are subject to stressors that affect their structure and function over large spatial scales. Usually, protected areas cover only a part of the catchment through which a river flows, so the pressures exerted in the downstream and especially in the upstream sections, can have dramatic repercussions on the integrity of the protected area. For these reasons, despite being among the most threatened environments (Harrison et al., 2018), stream ecosystems are, paradoxically, among the most difficult to protect, unless the protected area includes the headwaters of the drainage area.

Mountain low-order lotic systems are unique hotspots of biodiversity, rich in highly sensitive and stenothermic taxa that are strongly adapted to harsh conditions such as low temperatures, seasonal high flows, and physical instability of channels (Petts et al., 2000; Smith et al., 2001). Unfortunately, these environments are increasingly threatened by factors acting on a global scale, such as temperature increases and changes in precipitation amounts, patterns, and seasonality, and at a local scale, in particular human-derived alterations of the hydro-morphological characteristics (Doretto et al., 2021; Piano et al., 2020). Even in relatively pristine areas, the hydro-morphology of Alpine rivers is often altered by human actions, such as bank reinforcement, water abstractions, and the presence of dams and weirs. These actions are becoming more and more frequent in recent decades due to the growing need for water, and modifications to the natural hydrological cycle induced by climate change, representing a major threat to Alpine lotic ecosystems and related aquatic communities (Carolli et al., 2020).

About 90% of all Alpine watercourses are used and capitalized on for producing electricity through hydropower (CIPRA, 2014), also in protected areas. Regardless of the plant size and typology, water storage and abstraction imply morphological alterations of river beds and banks, which can threaten aquatic biodiversity. In recent years, the estimation of local and species contributions to β -diversity has received increasing scientific interest (Bo et al., 2020; Vilmi et al., 2017). Indeed, β -diversity, which corresponds to species turnover among sites in a given region, provides information about the degree of differentiation among biotic communities, playing a key role in conservation and bioassessment viewpoints. Indeed, it allows the identification of spatial trends and sites that most contribute to regional diversity, and may address specific restoration programs. This can be achieved by considering species and local contribution to

β -diversity in addition to more traditional metrics widely used in environmental bioassessment, such as biotic or diversity indices.

In 2005, we carried out the first evaluation of the morphological alterations of 12 high-altitude rivers within the Gran Paradiso National Park, the oldest national park in Italy, and their effect on the benthic (macroinvertebrates and diatoms) communities included in the EU Water Directive Framework 2000/60 (Bona et al., 2008).

In the present study, we sought to assess if (i) the presence of the protected area prevented significant local alteration of the stream habitat; (ii) the aquatic biodiversity and the ecological quality of the sampled stretches were comparable after 15 years or, conversely, if significant changes occurred, by analyzing taxonomical community composition, biotic indices, and β -diversity-related metrics. With this aim, we repeated the same samplings, in the same sites and with the same team and methods.

2 | MATERIALS AND METHODS

2.1 | Study area and data collection

The study was conducted in the Gran Paradiso National Park (hereinafter GPNP), which is the oldest protected area in Italy, created in 1856 as a royal hunting reserve for Alpine ibex (*Capra ibex* L. 1758) and then established as Natural Park in 1922. GPNP is located in north-western Italy, in the Graian Alps region between Piedmont and the Aosta Valley, encompassing an area of 703 km² and ranging from 800 to 4061 m a.s.l. The landscape is typically alpine, with a small portion (approximately 16%) used for agriculture and pasture and the rest occupied by alpine prairies, woods, rocks, and glaciers.

At 15 years after the sampling campaign of 2005, we sampled the same 12 stream reaches (Figure 1). The two sampling campaigns are faultlessly comparable because we sampled the same stream reaches in the same month (July), with the same sampling team, and by applying the same methods.

Here, we describe the methods in brief, while more detailed information is reported by Bona et al. (2008). All sampled reaches were characterized by a prevailing siliceous substrate, an altitude ranging from 1200 to 1900 m a.s.l., and similar human activities nearby (hiking, summer tourism, and pasture) but different levels of morphological alterations. To quantify them, we applied the South European River Habitat Survey (hereafter RHS) and related indices (Buffagni, 2004; Buffagni & Kemp, 2002; Raven et al., 1998): the Habitat Quality Assessment (HQA), the Habitat Modification Score (HMS), and the Lentic-Lotic River Descriptor (LRD). We also examined the available climatic data prior to the two sampling years (data available for the meteorological station inside the GPNP, located in Ceresole, and available from 1999; www.arpa.piemonte.it). We considered data from 1999 to 2005 (before the summer 2005 sampling campaign) and from 2014 to 2020 (before the summer 2020 sampling campaign), focusing on temperature (average, minimum, and maximum monthly values) and precipitation (monthly averages of precipitation and number of rainy days).

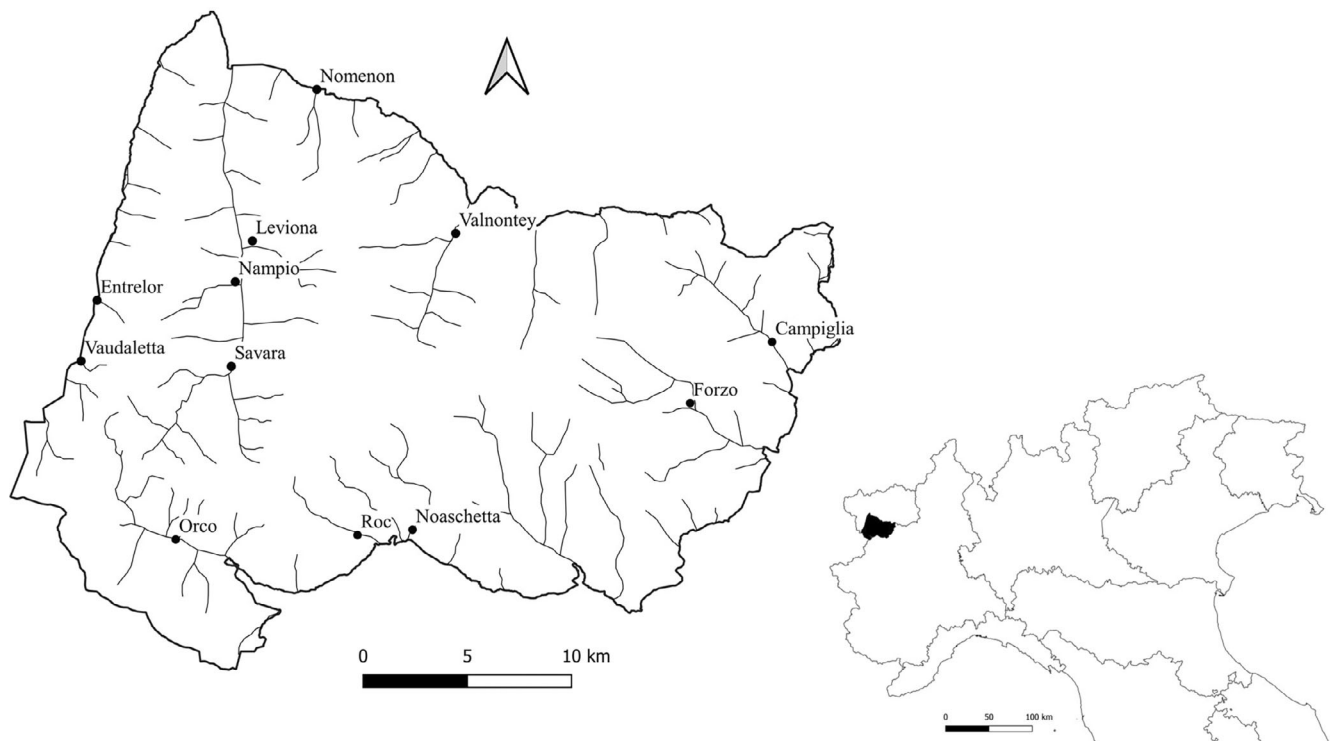


FIGURE 1 Study area with sampling sites.

All aquatic and benthic samples were taken in spot-check 2 of the RHS procedure. Physical and chemical parameters of water were measured following the methods reported in Bona et al. (2008). Benthic diatoms were collected by scraping five cobbles for each sampling station and analyzed following the same procedure as Bona et al. (2008). To assess changes in diatom communities, we computed the following indices: Indice de Polluosensibilité Spécifique (IPS) (Coste, 1982), Trophic Index (TI) (Rott et al., 1999), and Intercalibration Common Metric (ICMi) (Mancini & Sollazzo, 2009). This latter index has been adopted in Italy for the river ecological classification as the normative index (Environment Ministry Decree 260/2010, 2010). In addition, we also compared the 2 years in terms of richness (number of species) and diversity (Shannon Index \log_2 and evenness). Furthermore, we analyzed diatom communities in terms of ecological preferences (Carayon et al., 2019), guilds (Passy, 2007; Rimet & Bouchez, 2012), and by classifying each taxon according to the most recent diatom Red List (Hofmann et al., 2018). For macroinvertebrates, we followed the same procedure proposed by the 'Indice Biotico Esteso' (IBE, Ghetti, 1997) as in 2005. IBE is a non-quantitative method to assess the river biological quality based on presence/absence data. Benthic macroinvertebrates were sampled using a 250- μm -mesh kick net on diverse habitats and substrate grain sizes within each sampling station. The taxonomy level went from species to family (for some Trichoptera, Coleoptera, and Diptera). The following indices were calculated: BMWP (Biological Monitoring Working Party), ASPT (Average Score Per Taxon), IBE, N families, N taxa, N EPT (Ephemeroptera Plecoptera Trichoptera) families, and % stenoecious taxa (see Bona et al., 2008, for more details).

For both communities, we also calculated the local and the species contribution to β -diversity, as explained in the following section.

2.2 | Statistical analysis

To detect significant differences in single metrics between the two sampling campaigns, we applied the Wilcoxon signed test for paired samples. To highlight possible correlations between biological metrics (such as taxa richness, biodiversity, etc.) and indices derived from the RHS (i.e., HQA, HMS, and LRD), we applied the Spearman test. For community analyses, we ran a non-metric multidimensional scaling (NMDS) on taxa composition and performed the permutational multivariate analysis of variance (PERMANOVA) test to detect possible significant differences between years. All tests were performed on the basis of relative abundance for diatoms and on presence/absence data for macroinvertebrates. To test biotic homogenization in both diatom and macroinvertebrate communities, we performed the test of homogeneity for multivariate dispersion (Anderson, 2006). Homogenization generally leads to communities with a low number of species and is characterized by generalist taxa. In particular, we wanted to detect if 2020 assemblages were a subset of 2005 ones. These analyses were based on the Bray–Curtis distance on diatom abundance data and on the Jaccard metric for macroinvertebrates.

We followed the approach of Legendre and De Cáceres (2013) to estimate the proportion by which each sampling site and each taxon contributed to the overall area diversity. This approach calculates: (1) the local contribution to β -diversity (LCBD). LCBD is a measure of

the ecological uniqueness of the single sampling site: the higher uniqueness of the site, the higher its contribution to the total richness; (2) the species contribution to β -diversity (SCBD), which is a score calculated for each species (diatoms) or taxon (macroinvertebrates). The higher the SCBD for a single species, the higher its contribution to the variance among sites. We calculated LCBD and SCBD on Hellinger-transformed abundance data with the 'beta.div' function of the values using adespatial R package (Legendre & De Cáceres, 2013). All statistical analyses were performed in R (R Core Team, 2020).

3 | RESULTS

3.1 | RHS and water quality

To detect possible changes in the habitat quality of the Gran Paradiso rivers from 2005 to 2020, we applied the RHS method to evaluate their hydro-morphological status, and we measured the main chemical water parameters. In Figure 2, we report the boxplots of the three indices derived from the RHS application to the 12 river stretches.

For HQA, HMS, and LRD, there was no significant difference between the 2 years (Wilcoxon signed test $p = 0.56$, $p = 0.90$, and $p = 0.24$, respectively). The median value of HQA was slightly lower in 2020, whereas HMS remained almost the same, and the LRD highlighted an even more lotic character in 2020. Therefore, from 2005 to 2020, the hydro-morphology of the examined stretches did not highlight any significant differences. Analyzing the individual sites, Leviona suffered some deterioration in the HQA, while Valnontey was

the site that has improved the most. In terms of HMS, Forzo, Roc, and Nampio indicated the most marked increases in morphological alterations, while Entrelor, Vaudaletta, and Noaschetta were the most improved.

Regarding the physical and chemical water parameters, Table 1 reports the summary for the 2 years.

The only significant differences involved nitrates, which increased in the 2020 survey, and pH, which showed a decrease (Wilcoxon signed test $p < 0.05$ for both parameters). A slight yet not significant increase was observed in 2020 SRP and conductivity data.

3.2 | Climatic data

The most notable difference between the two periods considered was in the maximum temperature values in the summer and winter months (Figure 3); in both seasons, temperatures reached higher values in 2020 than in 2005. Furthermore, the annual trend of the average monthly rainfall shows clear differences, with higher values characterizing the years preceding the 2005 samplings.

3.3 | Benthic communities: Comparison among ecological indices

The comparison of diatom data between the two sampling campaigns is reported in Figure 4. From 2005 to 2020, we observed a slight increase in species richness and diversity, despite not being statistically relevant (Figure 4a-c). Diatom indices also depicted a

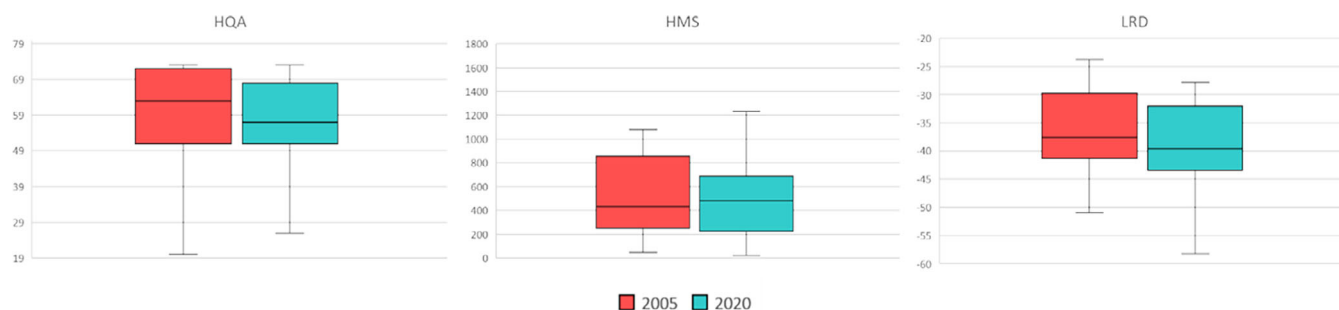


FIGURE 2 Boxplots of the indices derived from the River Habitat Survey application in 2005 and 2020. Box = 25th–75th percentiles; line = median; whiskers = $1.5 \times$ IQR (interquartile range). HMS, Habitat Modification Score; HQA, Habitat Quality Assessment; LRD, Lentic-Lotic River Descriptor. [Color figure can be viewed at [wileyonlinelibrary.com](https://onlinelibrary.wiley.com)]

TABLE 1 Summary data for water chemistry and temperature.

| Year | Water temperature (°C) | Conductivity ($\mu\text{S cm}^{-1}$) | pH | DO (%) | Soluble Reactive Phosphorus SRP (mg L^{-1}) | Nitrate (mg L^{-1}) |
|----------------|------------------------|--|--------------|-------------|--|--------------------------------|
| 2005 | 10.4 (2.9) | 93.6 (97.0) | 7.91 (0.418) | 104 (4.37) | 0.013 (0.011) | 0.250 (0.103) |
| 2020 | 10.5 (2.8) | 116 (87.7) | 7.51 (0.403) | 96.1 (10.1) | 0.038 (0.088) | 0.760 (0.400) |
| <i>p</i> value | 0.359 | 0.610 | 0.011 | 0.058 | 0.455 | 0.011 |

Note: Mean and standard deviations are reported. *p* values refer to the Wilcoxon test applied to 2005 and 2020 data.

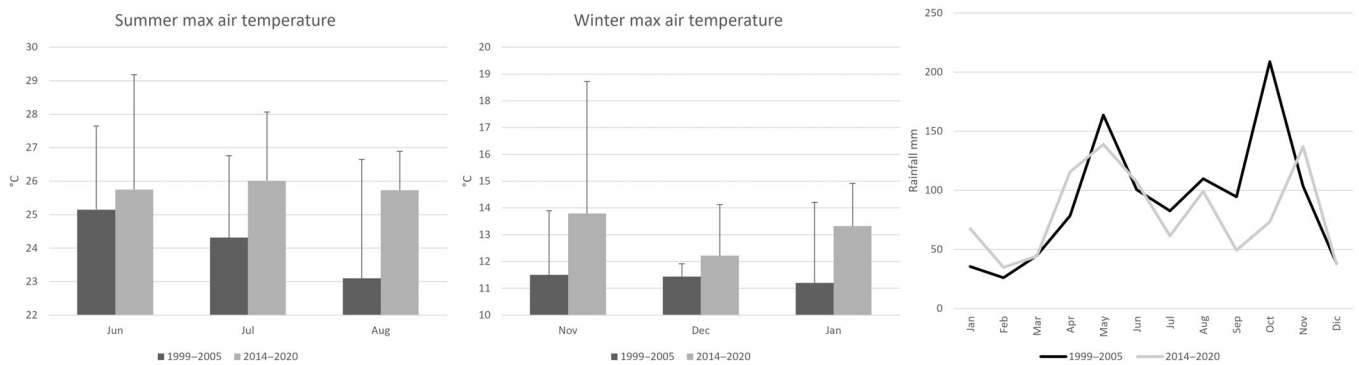


FIGURE 3 Comparison between climatic data in the years 1999–2005 and 2014–2020. Maximum air temperature: (a) summer monthly data and (b) winter monthly data; (c) average monthly rainfall (Source: Arpa Piemonte, www.arpa.piemonte.it). Bars represent standard deviations.

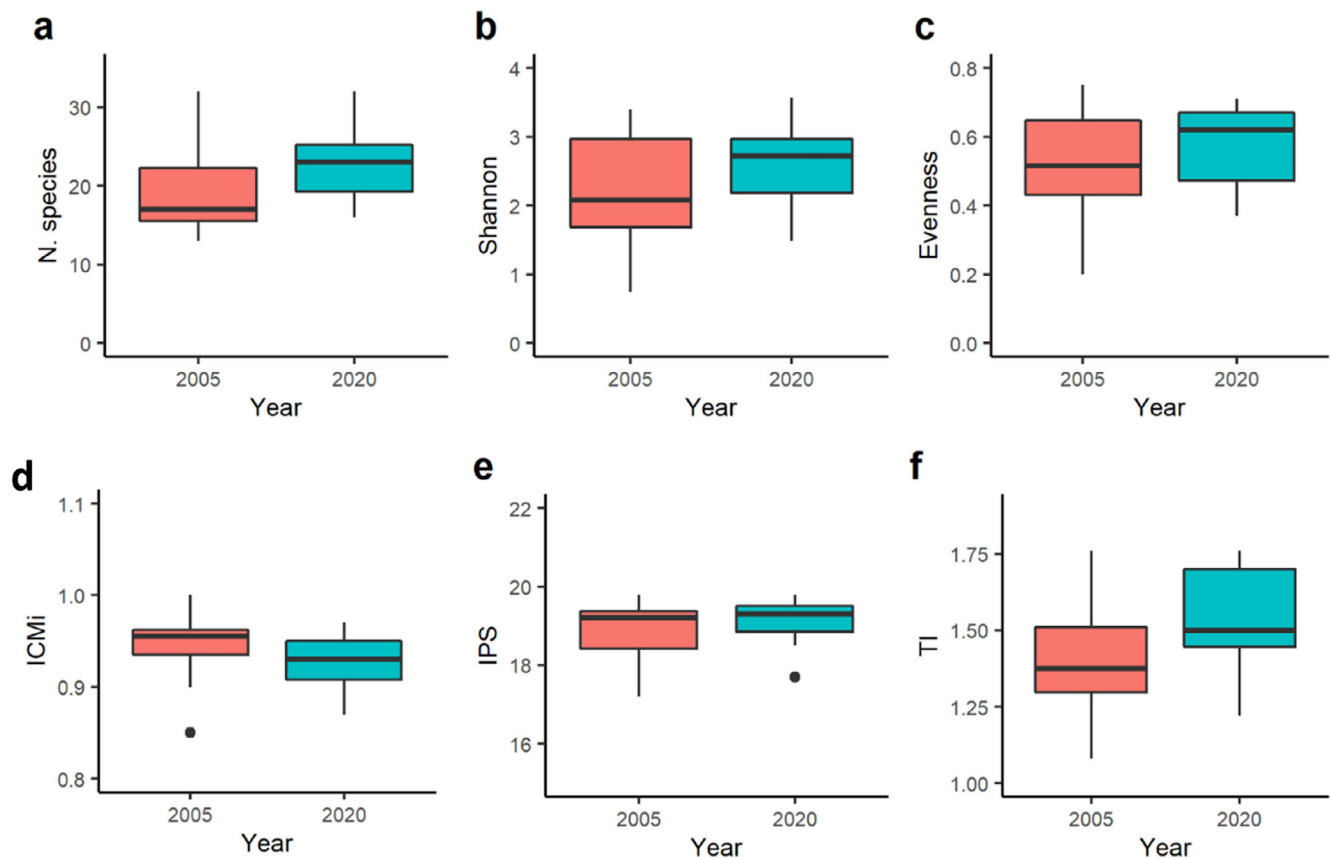


FIGURE 4 Comparison between 2005 and 2020 diatom community indices. Boxplots from (a) to (c) show richness and diversity indices. Boxplots from (d) to (f) show the most common European diatom indices: IPS, Indice de Polluosensibilité Spécifique (Coste, 1982); ICMi, Intercalibration Common Metric Index (Mancini & Sollazzo, 2009); TI, Trophic Index (Rott et al., 1999). Boxplot statistical limits are as in Figure 2. [Color figure can be viewed at wileyonlinelibrary.com]

substantially unchanging situation (Figure 4d–f), except for the TI, which was significantly higher in 2020 (Wilcoxon paired test $V = 11.5$, $p < 0.05$). An increase in TI corresponds to a decrease in environmental quality as this index increases along a gradient of human impact. The worsening in TI particularly affected Valnontey, Orco, Noaschetta, and Nampio streams.

In both years under investigation, diatom communities were mainly composed of alkalobiontic and alkalophilous taxa, adapted to

mean conductivity and very high oxygen saturation levels (99.5%). Diatom communities were dominated by oligosaprobous taxa with a small proportion of mesosaprobous ones, which slightly increased in 2020, and by oligotrophic (more than 90%) and oligo-mesotrophic taxa. In terms of ecological guilds, diatom assemblages of 2005 were, on average, dominated by low-profile species (76.5%), that is, species that are favored in oligotrophic waters and stretches characterized by frequent physical disturbance. Around 13.4% of the taxa belonged to

the high-profile guild, which usually develops in nutrient-rich sites, characterized by lentic flows and low disturbance conditions. Only 1.9% of the species were motile ones, which have the physical capability of selecting the most suitable habitat thanks to rapid movements. These are mainly epipelagic taxa that are also able to colonize finer substrates such as sand and silt. In comparison, samples from 2020 were characterized by a higher proportion of low-profile taxa (81.1%), which corresponded to a decrease in high-profile diatoms (10.3%). Motile species also increased, reaching 3%, and the remaining 5.6% was composed of planktic (free-floating in the water column) or unclassified taxa.

In both campaigns, *Achnanthyidium* was by far the most common genus in all sites, with the three dominant species, *Achnanthyidium lineare*, *Achnanthyidium pyrenaicum*, and *Achnanthyidium minutissimum*, being very common in Alpine rivers. *Achnanthyidium* species are low-profile and resistant to current velocity thanks to physiological adaptation mechanisms that allow them to tightly adhere to the substrate. Some other interesting species reached consistent relative abundances, at least in some sites. This was the case of *Fragilaria arcus*, a rheophilic species, very sensitive to pollution and typical of waters characterized by low conductivity ($<120 \mu\text{S cm}^{-1}$), found in high percentages in Forzo and Orco in 2005, whereas in 2020 this taxon was widely distributed but with low relative abundance everywhere. *Achnanthyidium thienemanni* was found in Campiglia in 2020 with a 10.7% relative abundance. *Epithemia goeppertiana*, a large motile species, usually epiphytic, was exclusively found in Nampio in 2020. *Cymbella excisiformis*, widely spread in headwater, was the third species besides *A. pyrenaicum* and *A. lineare* in Leviona in 2020, while *Encyonema silesiacum*, a high-profile species typical of siliceous rocks, was abundant in Roc.

It is worth noting that many of the species sampled in the GPNP are included in the Red List of diatom species (Hofmann et al., 2018) as endangered at different levels. The sites sheltering the highest number of endangered species were Noaschetta in 2005 and Leviona and Vaudaletta in 2020. On average, samples from 2005 hosted a lower abundance (39.5%) of individuals belonging to endangered species than those from 2020 (50.2%).

For macroinvertebrates, the list of taxa is reported in Table S1 (Supplementary Material). From this list, we computed seven metrics, none of which showed a significant change between the two sampling years (Wilcoxon test $p > 0.05$; boxplots in Figure 5). In general, macroinvertebrate community composition is characterized in both sampling campaigns by a relatively small number of well-adapted orophilic and rheophilic taxa. Considering together the two campaigns, the mean number of taxa equals 16, with a minimum value of 9 in Savara (2005) and a maximum of 23 in Campiglia (2020). In all the sampling sites and for both years, stenocious taxa accounted for at least 50% of the total taxa (average 64%, minimum 47% in Campiglia 2020, and maximum 87.5% in Gran Nomenon 2005). Overall, the macroinvertebrate communities of the GPNP are mainly composed of orophilic and stenobiont organisms strictly adapted to cold and harsh environments, such as highly sensitive Diptera (namely *Haplotrix lugubris*), Ephemeroptera Heptageniidae (in particular, *Epeorus alpicola*), and Plecoptera

Systemlognatha, such as *Dictyogenus alpinus*, *Perlodes intricatus*, and *Siphonoperla montana*.

No noticeable differences in taxonomic or functional compositions were recorded when comparing the two campaigns. For example, the occurrence of stenocious Plecoptera remained almost constant, as was the case for Perlidae (7 out of 12 sites in 2005 and 2020), Chloroperlidae (5 sites), and Perlodidae (8 sites in 2005, 7 in 2020). A similar pattern of occurrence was reported for Blephariceridae, found in 9 sites in 2005 and 10 in 2020.

We applied the Spearman correlation test to verify the association between HQA and biotic indices. HQA resulted as being significantly and positively correlated with the Shannon diversity index of diatoms ($p < 0.05$) and with two macroinvertebrate indices: EPT and IBE ($p < 0.05$).

3.4 | Benthic communities: Multivariate analysis

For diatoms, we applied the NMDS to community species abundances. Results are shown in Figure 6a, where the 2020 samples are nested in the 2005 data. The PERMANOVA test confirms a significant difference between the 2005 and the 2020 assemblages ($F_{1,22} = 2.692$, $p < 0.05$). Moreover, the analysis of multivariate homogeneity of group dispersions highlighted significant differences between the two sampling years ($F_{1,22} = 8.953$; $p < 0.01$). The 2005 communities showed a higher heterogeneity within sites in the taxonomic composition than the 2020 communities, as shown by a higher (and statistically significant) distance from the centroid in 2005, while the 2020 sites appeared more homogenous and therefore graphically closer to each other.

The same analysis applied to macroinvertebrate communities (in this case, on presence-absence data) did not detect a significant difference between the two sampling years, with only a slightly higher distance from the centroid in the 2005 assemblages (Figure 6b).

Figures 7 and 8 report the results concerning LCBD and SCBD applied to diatom and macroinvertebrate data, respectively. The 2 years did not show a significant difference in terms of LCBD (Figures 7a and 8a). For this reason, the following comments focus on the most recent campaign.

A. minutissimum and *A. lineare* had the highest SCBD score in 2020 (0.08 for both species). The other species that contributed most to β -diversity, in decreasing order, were *C. excisiformis*, *A. thienemanni*, *E. silesiacum*, and *Fragilaria pectinalis*.

The analysis of macroinvertebrates showed that *S. montana* had the highest SCBD score, followed by several other taxa attaining similar scores (e.g., Blephariceridae, Elmidae, *Nemoura* sp., Limoniidae). Chironomidae had by far the lowest score (0.002).

Regarding the contribution of sites to local diversity, Leviona, Roc, and Gran Nomenon were the sites displaying the highest LCBD in 2020 based on diatom community composition; and Roc Entrelor, Savara, and Campiglia based on macroinvertebrate data. Averaging the LCBD values obtained in the two communities, Roc, Entrelor, and Leviona were the sites that contributed most to local diversity.

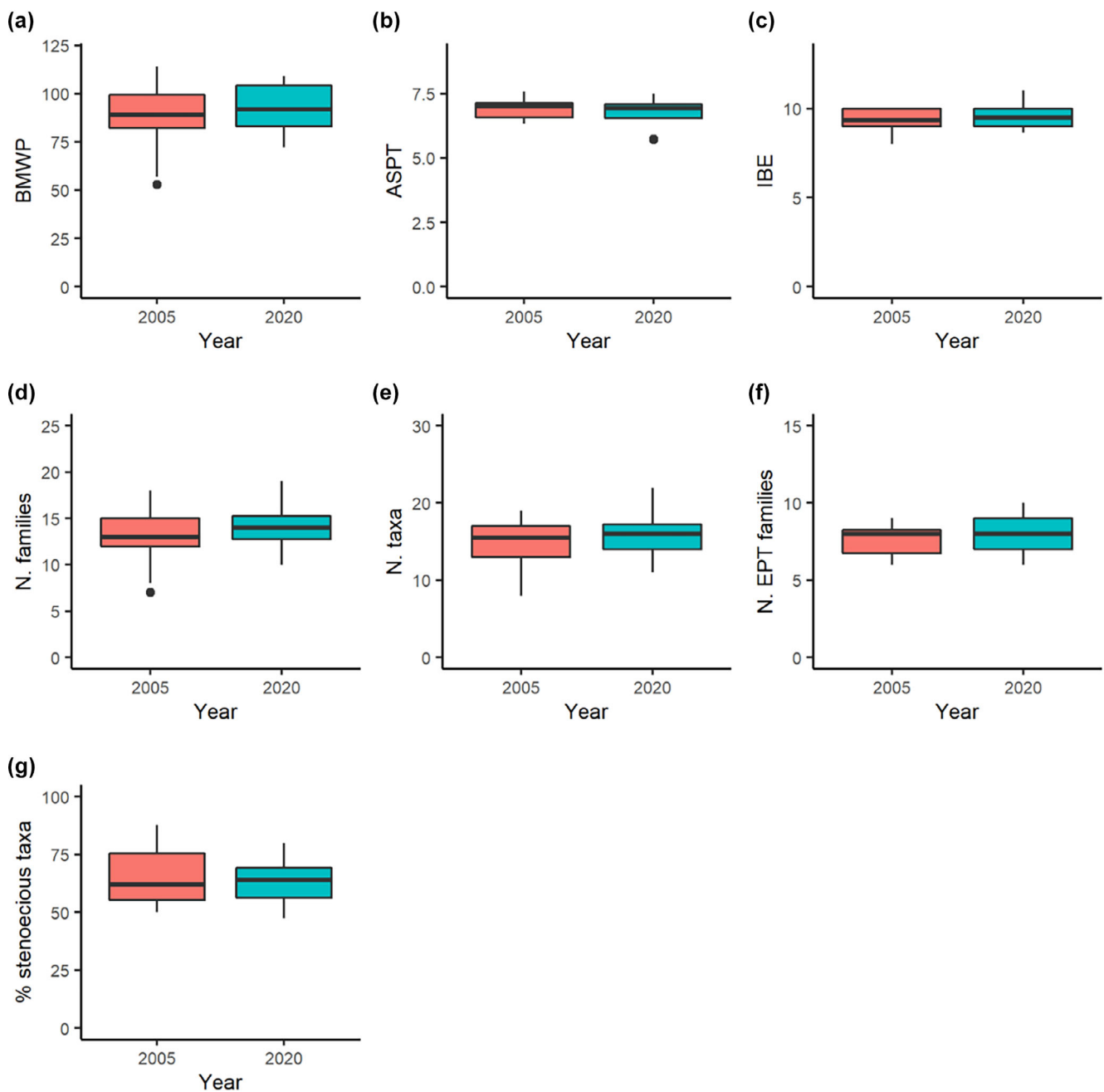


FIGURE 5 Comparison between 2005 and 2020 macroinvertebrate community indices. IBE, Italian Biotic Index (Ghetti, 1997). Boxplot statistical limits are as in Figure 2 BMWP, ASPT, IBE, N families, N taxa, N EPT families, % stenocious taxa. [Color figure can be viewed at [wileyonlinelibrary.com](https://onlinelibrary.wiley.com/doi/10.1002/rna.4124)]

4 | DISCUSSION

In freshwater environments, biodiversity loss is dangerously threatened by several anthropogenic pressures acting on streams and river ecosystems both at local and global scales, often acting together. Indeed, the impact of climate change in mountain areas is undoubtedly overwhelming and also has direct and indirect effects at a local scale by increasing the human water demand and related actions. This study is the first to examine possible temporal changes in hydro-morphology and aquatic biodiversity in an Alpine protected area. Our

study demonstrated that, overall, the hydro-morphology of the examined stretches within the Gran Paradiso area did not significantly vary over the last 15 years, as demonstrated by the indices derived from the RHS. HQA and HMS values indicated the presence of some modifications in the bank structure and the re-sectioning of the river course, as well as the strong lotic character, which in some cases reduces habitat diversification. The slight decrease in HQA or increase in HMS recorded in a few sites was mainly due to a decrease in the river flow and channel heterogeneity, as in the case of Leviona, or to bank reinforcements, as in Forzo, Roc, and Nampio. In some cases,

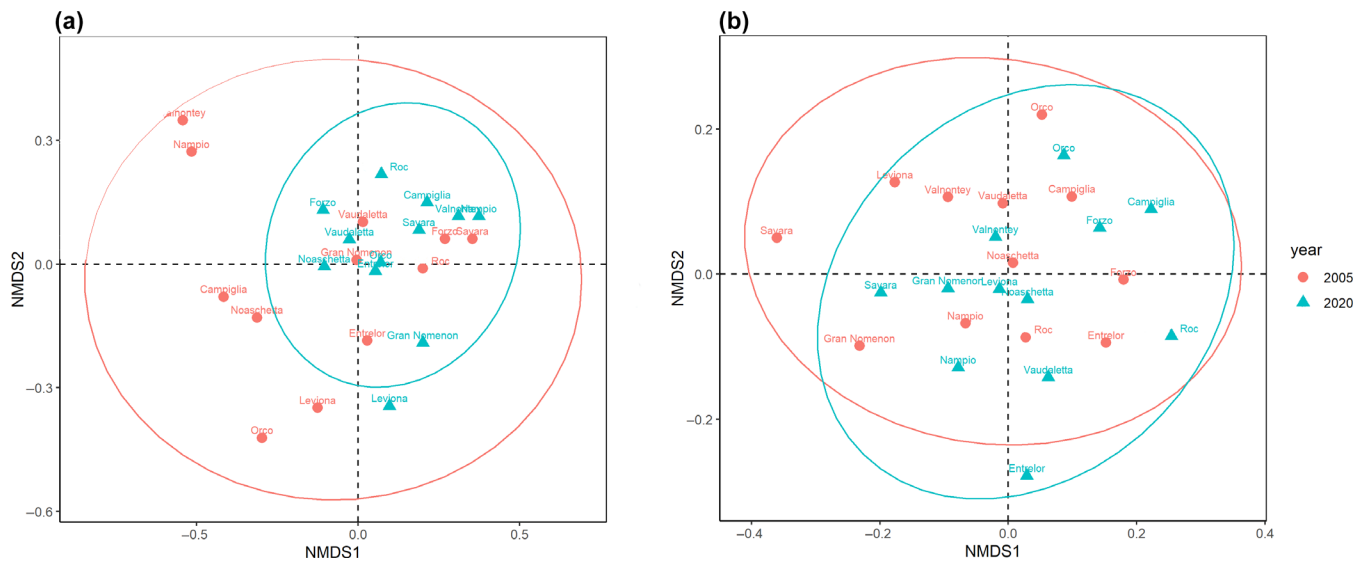


FIGURE 6 Graphical result of the non-metric multidimensional scaling displaying samples based on the diatom matrix. [Color figure can be viewed at [wileyonlinelibrary.com](https://onlinelibrary.wiley.com/doi/10.1002/nla.124)]

HMS improved thanks to the removal of artificial structures such as small bridges or weirs. The stronger loticity highlighted by the LRD Descriptor in 2020 confirms the highest percentage of erosive habitats detected during the last survey. Indeed, only 28% out of 194 monitored spot-checks showed erosive bank features in 2005 (such as local erosion, eroding cliff, or eroding bank), while there were 66% in 2020.

Water quality highlighted a small increase in the nitrate concentration in 2020 with a mean value of 0.760 mg L^{-1} , which still corresponds to a first quality class according to the Italian legislation (D.M. 260/2010) and is in line with nitrate concentrations in streams originating from ablating glacier (Brighenti et al., 2019). This increase in nitrates could be related to the ice melting as glaciers accumulated several atmospherically delivered chemical compounds, and nitrate is easily eluted from melting glacier ice and snowpack (Saros et al., 2010). Indeed, the climatic data, although limited as temporal series, seem to indicate a trend towards an increase in maximum temperature: in recent years, the summer temperature reaches values close to 26°C , and, also in the winter months, values are always higher than in the 1999–2005 period. In addition, the annual trend of the average monthly rainfall shows clear differences, with higher values almost always in the years preceding the 2005 samplings. These data are in line with what has been highlighted by studies on climatic trends in high mountain streams of southern Europe, which is identified as an area particularly affected by climate change, as highlighted by Doretto et al. (2018) and Gobiet et al. (2014) for the Alps, and by Sáinz-Bariáin et al. (2016) and Alba-Tercedor et al. (2017) for the Iberian Peninsula.

Overall, our study shows a picture of few changes in local hydro-morphological impacts, accompanied by slight variations in water quality in a period affected by visible consequences of climate change.

This is probably due to the environmental protection measures related to the presence of a protected area.

How the stream benthic communities, and especially diatoms, respond to these changes is shown by the multivariate analysis of the community composition rather than by traditional indices. The NMDS ordination technique performed on diatom community composition showed that the 2020 samples are nested in a wider ellipsoid, including the 2005 data, and resulted in significantly less heterogeneity. The species turnover between the 2 years mainly concerns co-generic species, mostly within the genera *Navicula* and *Nitzschia*. *A. lineare*, which showed a positive trend from 2005 to 2020, and *A. minutissimum* have the highest SCBD scores. These two species, however, have different ecological preferences: *A. lineare* is more sensitive to anthropic pressures, while *A. minutissimum*, despite being very common in mountain streams, has a wide tolerance range towards both water pollution and physical alterations (Brighenti et al., 2019). It is important to note that, in 2005, species classified as ‘early alert’ in the diatom Red List represented, on average, 6.6% of the total abundance, compared with only 3.1% in 2020. Of the 11 taxa belonging to this category, *F. arcus* showed the most significant decrease from a mean abundance of 5.8% in 2005 to 2.2% in 2020. Another species deserving attention is *Psammothidium subatomoides*, recorded in 8 samples out of 12 in 2005, but disappeared in 2020 inventories. The results obtained in this research confirm the need to preserve habitat conditions favoring the recovery of these species. Among the restoration measures that have proved to be more efficient to preserve healthy diatom communities, we cite the maintenance of natural flows (Falasco et al., 2016) and of riparian vegetation that enhances the water temperature stability (Bowler et al., 2012). When looking at species not yet classified in the Red List because of insufficient data, we focused on those species following a decreasing trend over the

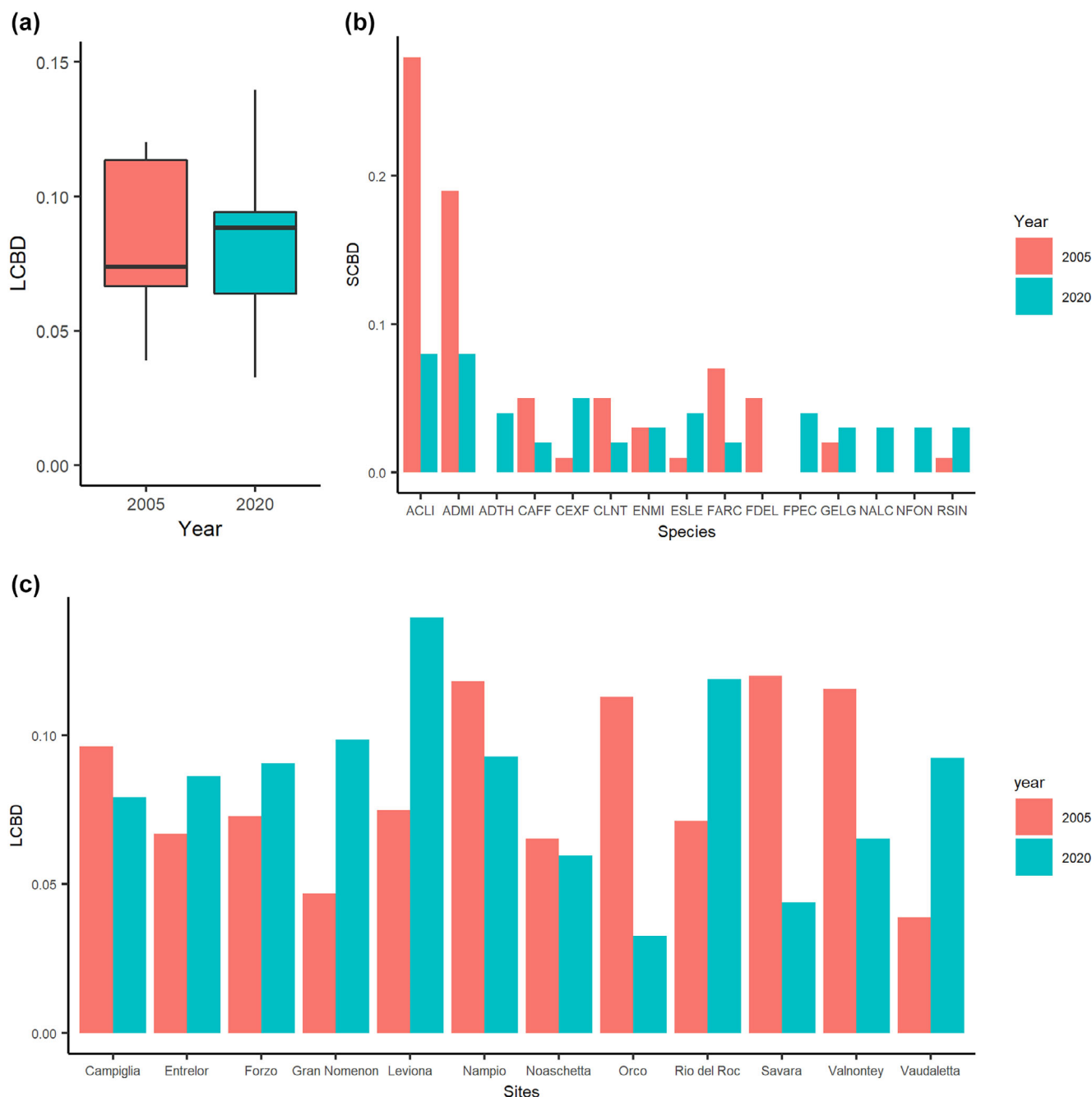


FIGURE 7 Contribution to diatom community β -diversity. (a) LCBD boxplots for 2005 and 2020; (b) SCBD of each diatom species; (c) LCBD of sampling sites. Species with SCBD < 0.02 were not included. Boxplot statistical limits are as in Figure 2. LCBD, local contribution to β -diversity; SCBD, species contribution to β -diversity. [Color figure can be viewed at [wileyonlinelibrary.com](https://onlinelibrary.wiley.com/doi/10.1111/jbr.12541)]

two sampling periods. Among them, *Fragilaria delicatissima*, detected in 2005, was not present in any of the 2020 inventories, while *Encyonopsis minuta* frequency showed a decrease. As these are both very sensitive species, particular attention to their temporal and spatial distribution pattern should be paid. We found a general positive trend between the increasing habitat quality (HQA), diatom species richness, and Shannon diversity, confirming that habitat heterogeneity is one of the main drivers for taxa richness (Bona et al., 2008). At the same time, this trend was negative (although not significant) considering

HMS, which means that poor communities are typical of the most impacted sites.

Among the biomonitoring indices, a significant worsening of the ecological status was only detected by the TI index (Wilcoxon test, $p = 0.034$), which on average indicated oligotrophy in 2005 and mesotrophy in 2020. TI is the most sensitive index to nutrient levels of natural origin (Rott et al., 1999) and is known for its suitability in Alpine rivers. The IPS remains in the class of oligotrophy, and the ICMi indicates a high ecological quality for both years.

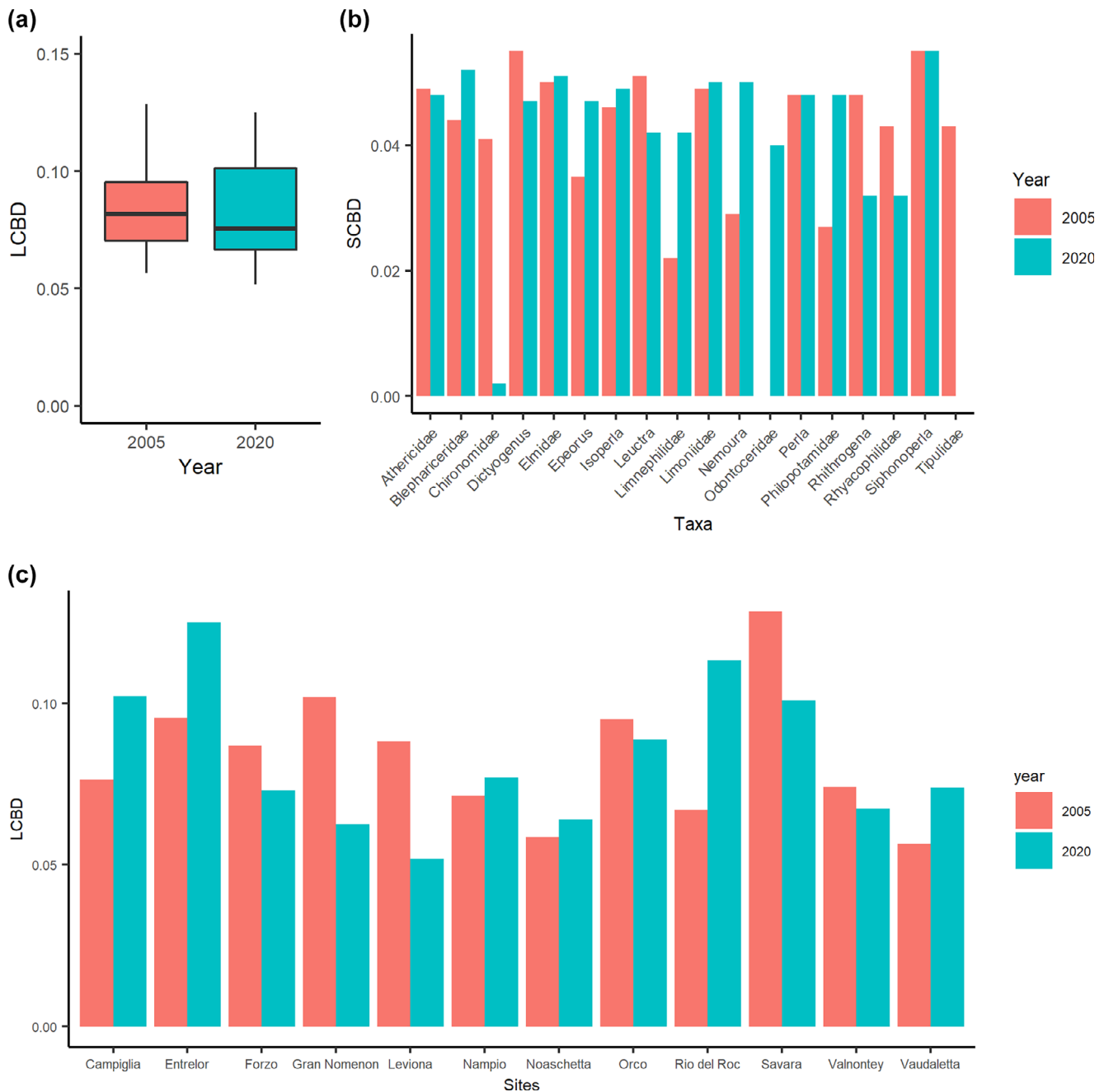


FIGURE 8 Contribution to macroinvertebrate community β -diversity. (a) LCBD boxplots for 2005 and 2020; (b) SCBD of each taxon; (c) LCBD of sampling sites. Boxplot statistical limits are as in Figure 2. LCBD, local contribution to β -diversity; SCBD, species contribution to β -diversity. [Color figure can be viewed at wileyonlinelibrary.com]

Macroinvertebrate communities showed no significant variations between the two campaigns. Orophilic and stenoxymbiont organisms, strictly adapted to cold and harsh environments, characterized these communities on both occasions. It is interesting to note that, regardless of the sampling year, macroinvertebrate biomonitoring indices showed a greater correlation with hydro-morphological alteration than those based on diatoms.

These differences in the response to anthropogenic stress confirm what has been reported by previous studies that compared the two indicators (Bona et al., 2008; Mykrä et al., 2012; Piano

et al., 2019): diatoms respond to water quality and habitat features at different scales with respect to macroinvertebrates, discouraging the use of one group as a substitute for the other. Their different responses are undoubtedly linked to bio-ecological factors, such as their different trophic roles, and the difference in size and length of the life cycle. Metrics related to diversity are certainly influenced by the taxonomic level and the qualitative or quantitative nature of the data collected as well. Indeed, indices currently used for macroinvertebrates require the genus as the finest taxonomic level, and data are expressed as presence/absence. For diatoms, the metrics used were

based on the taxonomic level of species and their relative abundance, and this could have contributed to their more complex response highlighted by the multivariate analysis.

Our study highlights the role of a large, protected area in preserving freshwater habitats and biological communities from human impacts. The β -diversity partitioning identified sites to prioritize in protection efforts, confirming its role in conservation planning (Ruhí et al., 2017). Moreover, this research demonstrates that creating databases relating to the ecological quality of rivers is crucial for tracking the environmental evolution of lotic systems, creating historical series of considerable management interest, and also supporting decision-making strategies in rural or mountainous areas. In this context, the presence of a natural park is essential for its role as a promoter of research and data collection.

Indeed, conservation of aquatic ecosystems is a pivotal mission of protected areas and requires harmonization actions among natural park managers and stakeholders, who should be driven toward biodiversity protection. In protected areas, water use for human purposes (such as recreational or for energy production) is generally minimal, and catchment integrity is mostly maintained, as are spatial and temporal river connectivity and habitat heterogeneity. All these features significantly contribute to the maintenance of a high functional richness of biological communities, as demonstrated by recent research (B-Béres et al., 2021; Falasco et al., 2018, 2020, 2021). It has been recognized that high functional richness increases community resilience against extreme events, which is expected to significantly increase under a global climate change scenario (B-Béres et al., 2021; Doretto et al., 2020; Mason et al., 2005). In this context, the GPNP, which includes the head of the drainage area, can effectively exert its protective action on river communities by preserving its functional richness. Moreover, headwaters will likely represent a possible refuge for taxa in a scenario of climatic warming (Sáinz-Bariáin et al., 2016).

In conclusion, this study once again underlines the importance of maintaining and, if necessary, enlarging Alpine protected areas that face an era of rapid climate change (Finlayson & Pittock, 2018), as they play a fundamental role in the preservation of the water resource and the ecosystems that guarantee its quality.

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DATA AVAILABILITY STATEMENT

The data that support the findings of this study are available from the corresponding author upon reasonable request.

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SUPPORTING INFORMATION

Additional supporting information can be found online in the Supporting Information section at the end of this article.

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