

Are protected areas effective in preserving alpine stream morphology and biodiversity? An experience in the first Italian National Park

Francesca Bona¹, Tiziano Bo², Alberto Doretto², Elisa Falasco¹, Marta Zoppi¹, and Stefano Fenoglio¹

¹University of Turin

²ALPSTREAM Alpine Stream Research Center/Parco del Monviso

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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 alpine headwaters, where the effects of global change are dramatically evident and often accompanied by alterations in river hydro-morphology. In mountainous river stretches, regulation and morphological alterations such as bank reinforcement, water abstractions, dams, and weirs are increasing. In the Alps, 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 samplings and hydro-morphological evaluation in twelve high-altitude streams within the oldest Italian National Park, the Gran Paradiso Park, located in the heart of the Western Alps, and we compared results with a previous survey performed in 2005, keeping the same experimental design. Our results detected minimal changes in the hydro-morphology of the studied watercourses. Biomonitoring indices associated with benthic communities likewise do not evidence significant differences. Concerning diatom flora, we found however in 2020 a greater uniformity in species composition compared to communities of 2005, and a slight turnover between species. In conclusion, 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.

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Running title: Stream morphology and aquatic biodiversity in a protected area

Bona F.*^{1,4}, Bo T.^{2,4}, Doretto A.^{3,4}, Falasco E.^{1,4}, Zoppi M.^{1,4}, Fenoglio S.^{1,4}

¹DBIOS, University of Turin, Via Accademia Albertina, 13, I-10123, Turin, Italy

²NaturaStaff, Via Lunga 11, Mongardino, Italy

³DISSTE, University of Piemonte Orientale, Piazza S. Eusebio 5, Vercelli, Italy

⁴ALPSTREAM, Alpine Stream Research Center/Parco del Monviso, Ostana, Italy

*Corresponding author: Francesca Bona francesca.bona@unito.it

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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 alpine headwaters, where the effects of global change are dramatically evident and often accompanied by alterations in river hydro-morphology. In mountainous river stretches, regulation and morphological alterations such as bank reinforcement, water abstractions, dams, and weirs are increasing. In the Alps, 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 samplings and hydro-morphological evaluation in twelve high-altitude streams within the oldest Italian National Park, the Gran Paradiso Park, located in the heart of the Western Alps, and we compared results with a previous survey performed in 2005, keeping the same experimental design. Our results detected minimal changes in the hydro-morphology of the studied watercourses. Biomonitoring indices associated with benthic communities likewise do not evidence significant differences. Concerning diatom flora, we found however in 2020 a greater uniformity in species composition compared to communities of 2005, and a slight turnover between species. In conclusion, 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: alpine rivers, β -diversity, diatoms, macroinvertebrates, hydromorphology, RHS

Introduction

Habitat loss and fragmentation, over-exploitation of natural resources, climate change, and introduction of invasive species are listed among the main causes of the current environmental crisis, characterized by an impressive extinction rate (Dirzo, Ceballos, & Ehrlich, 2022). Among the various strategies implemented to contrast this negative trend, the increase and extension of protected areas are considered one of the most useful and immediate (Lovejoy, 2006; Acreman, Hughes, Arthington, Tickner, & Dueñas, 2020). Unfortunately, however, while the creation of 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 functioning 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 section 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 paradoxically are among the most difficult to protect, unless the protected area is located at the head of the drainage area.

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

About 90% of all alpine water courses are used and capitalized on for producing electricity through hydro-power (CIPRA, 2014), also in protected areas. Regardless of the plant size and typology, water storage and

abstraction imply morphological alterations of river bottoms and banks which can threaten aquatic biodiversity. In recent years, the estimation of local and species contributions to β -diversity is receiving increasing scientific interest (Vilmi, Karjalainen, & Heino, 2017; Bo, Doretto, Levrino, & Fenoglio, 2020). Indeed, β -diversity plays a key role in conservation and bioassessment viewpoints because 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 a study that was the first evaluation of high-altitude river morphological alterations in an Alpine protected area, 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, Falasco, Fenoglio, Iorio, & Badino, 2008, in this journal).

In this study, we sought to assess if the presence of the protected area prevented significant local alteration of the stream habitat and if the aquatic biodiversity and ecological quality of the sampled stretches were comparable after 15 years or underwent significant changes, measured as biotic indices and β -diversity-related metrics. With this aim, we repeated the same samplings, on the same sites and with the same team and methods.

Materials and methods

Study area and data collection

The study was conducted in the Gran Paradiso National Park (hereinafter GPNP). This park is the oldest protected area in Italy, born in 1856 as a royal hunting reserve for Alpine Ibex and then established as Natural Park in 1922. GPNP is located in North-Western Italy, in the Graian Alps region between Piemonte and Aosta Valley, encompassing 703 km² and ranging from 800 to 4061 m a.s.l. The land use 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.

Fifteen years after the sampling campaign of 2005, we sampled the same 12 (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.

FIG1

In the present paper, we describe methods in brief, while more detailed information is reported in Bona et al. (2008). All sampled stretches are characterized by a prevailing siliceous substrate, altitude ranging from 1200 and 1900 m a.s.l., and similar human activities nearby (hiking, summer tourism, pasture) but different levels of morphological alterations. To quantify them, we applied the South European River Habitat Survey (hereafter RHS) and related indices (Raven, Holmes, Dawson, & Everard, 1998; Buffagni & Kemp, 2002; Buffagni, 2004): Habitat Quality Assessment (HQA), Habitat Modification Score (HMS) and Lentic-Lotic River Descriptor (LRD). We examined also the available climatic data in correspondence with 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 those 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 the number of rainy days).

All the aquatic samples were taken in spot-check 2 of the RHS procedure. Water physical and chemical parameters were measured following the same methods as 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). Concerning diatoms, we computed the following indices: IPS (Coste, 1982), TI (Rott et al., 1999), and ICM (Mancini & Sollazzo, 2009), which replaced the EPI_D in the Italian application of the Water Framework Directive. In addition, we also compared the two years in terms of richness (number of species) and diversity (Shannon Index and Evenness). Furthermore, we analyzed diatom communities

in terms of ecological preferences (Carayon, Tison-Rosebery, & Delmas, 2019), guilds (Passy, 2007; Rimet & Bouchez, 2012) and by classifying each taxon according to the most recent diatom Red List (Hofmann, Lange-Bertalot, Werum, & Klee, 2018). Benthic macroinvertebrates were sampled using a 250 μm mesh kick net on diverse habitats and substrate grain size and typology within each sampling station. The taxonomy level went from species to family (for some Trichoptera, Coleoptera, and Diptera). The following indices were calculated: BMWP, ASPT, IBE, N families, N taxa, N EPT families, and % stenocious 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.

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 RHS (i.e., HQA, HMS, and LRD) we applied the Spearman test. For community analyses, we ran an NMDS on taxa composition and performed the PERMANOVA test to detect possible significant differences between years. To test biotic homogenization in both taxonomic diatom and macroinvertebrate communities, we performed the test of homogeneity for multivariate dispersion (Anderson, 2006). These analyses were based on the Bray-Curtis dissimilarity metric for diatoms, while the Jaccard metric for macroinvertebrates.

We followed the approach of Legendre & 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 the site uniqueness, 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 is SCBD for a single species, the higher its contribution to the variance among sites. We calculated LCBD and SCBD Hellinger-transformed abundance data with the “beta.div” function of the values using *adespatial* R package (Legendre & De Cáceres, 2013). All the statistical analyses were performed in R (R Core Team, 2020).

and has been used as a tool to identify sites to prioritize in conservation efforts (Ruhí, Datry, & Sabo, 2017).

Results

River Habitat Survey 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 reported the boxplots of the three indices derived from the RHS application on the 12 river stretches.

FIG2

For all three indices the difference between the two years is not significant (Wilcoxon Signed Test $p > 0.05$). The median value of HQA is slightly lower in 2020, whereas HMS remains almost the same and the LRD highlights an even more lotic character in 2020. Overall, from 2005 to 2020 the hydro-morphology of the examined stretches did not highlight any evident change. Analyzing the individual sites, Leviona has suffered some deterioration in the HQA, while Valmontey is the site that has improved the most. In terms of HMS, Forzo, Roc and Nampio indicate the most marked increases in morphological alterations while Entrelor, Vaudaletta and Noaschetta are the most improved.

Concerning the physical and chemical water parameters, Table 1 reports the summary for the two years.

TAB1

The only significant differences concern nitrate, 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 regards also SRP and conductivity.

Climatic data

The most visible difference between the two periods considered is in the maximum values of the temperature in the summer and winter months (Figure 3): in both seasons temperatures reach values higher in 2020 than in 2005. Also, the annual trend of the average monthly rainfall shows clearly visible differences, with higher values characterizing the years preceding the 2005 samplings.

Benthic communities: comparison among ecological indices

Concerning diatoms, the comparison 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 4 a,b,c). Diatom indices also (Fig 4 d,e,f,g) depict a substantial unchanging situation, except for the Trophic Index TI, which is significantly higher in 2020 (Wilcoxon paired test $V = 11.5$, $p < 0.05$). An increase in TI corresponds to a decrease in the environmental quality as this index increases along a gradient of human impact. The worsening in TI concerns, in particular, Valnontey, Orco, Noaschetta and Nampio.

Fig4

In both years, diatom communities were mainly composed of alcalobiontic and alcalophilous taxa, adapted to mean conductivity and very high oxygen saturation levels (99.5%). Concerning tolerance to organic matter and nutrients, both diatom communities were dominated by oligosaprobous taxa with a small proportion of mesosaprobous ones, which slightly increased in 2020; oligotrophic (more than 90%) and oligo-mesotrophic taxa. In terms of ecological guilds, diatom assemblages of 2005 were in mean dominated by low-profile species (76.5%), which are favored in nutrient-poor and high-disturbance habitats; 13.4% of the taxa belonged to the high-profile guild, developing in nutrient-rich sites, more lentic flows and low disturbance conditions; while only 1.9% was represented by motile species, mainly epipellic taxa favored by high nutrient content and able to move within periphyton and river bed toward suitable habitats. 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%.

In both campaigns, the genus *Achnantheidium* is by far the dominant one in all the sites, with the three species *Achnantheidium lineare*, *Achnantheidium pyrenaicum* and *Achnantheidium minutissimum*, very common in the Alpine rivers. *Achnantheidium* species are low profile, resistant to current velocity thanks to physiological adaptation mechanisms which allow them to tightly adhere to the substrate. Some other interesting species reached consistent relative abundances, at least in some sites. This is the case of *Fragilaria arcus*, a rheophilic species, very sensitive to pollution and typical of low mineralized freshwaters, found in high percentages in Forzo and Orco in 2005, well distributed but with lower abundances in the 2020 campaign. *Achnantheidium thienemannii* was found in Campiglia in 2020 with a 10.7% relative abundance. *Epi-themia goeppertiana*, a large motile species, usually epiphytic, was exclusively found in Nampio in 2020. *Cymbella excisiformis*, widely spread in headwater is the third species 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. In this sense, the sites sheltering the highest number of endangered species were in 2005 Noaschetta and in 2020 Leviona and Vaudaletta. On average, 2005 samples hosted a lower abundance (39.5%) of individuals belonging to endangered species than 2020 ones (50.2%).

For macroinvertebrates 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). On average, macroinvertebrate community composition is characterized in both sampling campaigns by a relatively small number of well-adapted, orophilic and rheophilic taxa, with a mean number of taxa equal to 16, 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 the stenoeccious taxa account for at least 50% of the total taxa (average 64%, minimum 47% in Campiglia 2020 and maximum 87.5% in Gran Nomenon 2005). Indeed, these communities are mainly composed of orophilic and stenoxymbiont organisms strictly adapted to cold and harsh environments, such as highly sensitive Diptera (namely *Haplotrix lugubris*) Ephemeroptera Heptageniidae (in particular, *Epeorus alpicola*) and Plecoptera Systellognatha, such as *Dictyogenus alpinus*, *Perlodes intricatus* and *Siphonoperla montana*.

No noticeable differences can be evidenced by comparing the two campaigns. For example, the occurrence of stenoeccious Plecoptera remains constant, as in the case of Perlidae (58.3 % of stations on both occasions), Chloroperlidae (41.7 %) and Perlodidae (66.7 % in 2005, 58.3 in 2020) were found. The same pattern of occurrence is reported for Blephariceriidae, found in 75% of samples in 2005 and 83.3 % of samples in 2020.

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

Benthic communities: multivariate analysis

For diatoms, we applied the NMDS to community species abundances. Results are shown in Figure 6, where the 2020 samples outline an inner ellipsoid nested in a wider ellipsoid including the 2005 data. This distinction is confirmed by the PERMANOVA test which 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 ($F_{1,22} = 8.953$; $p = 0.007$). The 2005 communities showed a higher heterogeneity among sites in the taxonomic composition than the 2020 communities, as proved by a higher (and statistically significant) distance from the centroid in 2005, while 2020 sites appear more homogenous and therefore graphically closer to each other.

Fig 6

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 (graphic not shown).

Figures 7 and 8 report the results concerning LCBD and SCBD applied to diatom and macroinvertebrate data respectively. The two years did not show a significant difference in terms of LCBD (Figure 7a and Figure 8a). For this reason, Figures 7b,7c, 8b and 8c report the single LCBD and SCBD calculated for the most recent campaign.

Fig 7

Fig 8

A. minutissimum and *A. lineare* got the highest SCBD score in 2020 (0.08 for both species). The other species contributing most to β -diversity were *C. excisiformis*, *Achnantheidium thienemani*, *E. silesiacum* and *Fragilaria pectinalis*.

Concerning macroinvertebrates, *S. montana* among others got the highest SCBD score, followed by several other taxa attaining similar scores (e.g. Blephariceridae, Elmidae, *Nemoura* sp., Limoniidae). Chironomidae got by far the lowest score.

Regarding the contribution of sites to local diversity, sites displaying the highest LCBD in 2020 based on diatom community composition were Leviona, Roc and Gran Nomenon; Roc Entrelor, Savara and Campiglia

according to macroinvertebrate data. Averaging the LCBD values obtained on the two communities, the sites that contribute most to local diversity are Roc, Entrelor and Leviona.

Discussion

In freshwater environments, biodiversity loss is dangerously menaced by several anthropogenic pressures acting on streams and river ecosystems both at local and global scales. 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, and the strong lotic character which in some cases reduces the habitat diversification. The slight decrease in HQA or increase in HMS recorded in a few sites is 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, HMS improved thanks to the removal of artificial structures such as small bridges or weirs. The stronger loticity highlighted by 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 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 that still corresponds to a first quality class according to the Italian legislation (DM 260/2010) and that 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 atmospheric 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 reach values close to 26 °C, and also in the winter months the values are always higher than in the 1999-2005 period. Also, the annual trend of the average monthly rainfall shows clearly visible 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 the Alps, which are identified as an area particularly affected by climate change (Doretto et al., 2018; Gobiet et al., 2014).

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,

How the stream benthic communities, and especially diatoms, respond to these changes is evidenced by the multivariate analysis of the community composition rather than by the 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 heterogeneous. The species turnover between the two 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: the 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 notice that in 2005 species classified as “early alert” in the diatom Red List represented on average 6.6% of the total abundance, while 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 and disappeared in 2020 inventories. The results obtained in this research confirm the need to preserve habitat conditions favoring the recovery of these species. 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 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. Being both very sensitive species, particular at-

tention 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 habitat modification score, which means that poor communities are typical of the most impacted sites. Moreover, according to Ossyssek, Hofmann, Geist, & Raeder (2022), we found a good correlation between total diatom richness and endangered species.

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

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 notice 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 which 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 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 role, 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 collected in 2005 were expressed as presence/absence. For diatoms, the metrics used are 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, Datry, & Sabo, 2017).

Moreover, this research demonstrates that the creation of 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 supporting decision-making strategies also 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, the conservation of aquatic ecosystems is one of the main objectives of protected areas and requires harmonization actions among protected areas managers and stakeholders, who should be driven toward biodiversity protection. In protected areas, water use for human purposes (such as recreational or energetical) is generally minimal, and catchment integrity is mostly maintained, so as 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 (Falasco, Piano, Doretto, Fenoglio, & Bona, 2018; Falasco, Doretto, Fenoglio, Piano, & Bona 2020; Falasco, Bona, Risso, & Piano, 2021; B-Béres, Stenger-Kovács, Török, & Török-Krasznai, 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 (Mason, Mouillot, Lee, & Wilson, 2005; Doretto et al., 2020; B-Béres et al., 2021). However, not all protected areas can play this role with the same effectiveness with rivers. In fact, the GPNP covers the entire headwater and therefore there are no impacts located upstream.

In conclusion, this study once again underlines the importance of maintaining and, if necessary, enlarging the alpine protected areas facing 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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Table 1 - Summary data for water chemistry and temperature. Mean and standard deviations are reported.

Year	Water t °C	Conductivity $\mu\Sigma \text{ } \zeta\mu^{-1}$	pH	DO %	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)

Figure 1 - Study area with sampling sites

Figure 2- Boxplots of the indices derived from the RHS application in 2005 and 2020. The bottom and top of each box are the 25th and 75th percentiles, the line in the middle is the median, whiskers go from the end of the interquartile range to the furthest observation within 1.5 times the interquartile range. HQA = Habitat Quality Assessment HMS = Habitat Modification Score LRD= Lentic Lotic River Descriptor.

Figure 3 – Comparison between climatic data in the years 1999-2005 and 2014-2020. Maximum air temperature a) summer monthly data b) winter monthly data; c) average monthly rainfall (data source: Arpa Piemonte www.arpa.piemonte.it). Errors bars are standard deviations.

Figure 4 – Comparison between 2005 and 2020 diatom community indices. Boxplots from d) to g) show the most common European diatom indices: IPS= Indice de Polluosensibilité Spécifique (Coste, 1982); ICMi= Intercalibration Common Metric Index (Mancini and Sollazzo, 2009); TI= Trophic Index (Rott et al., 1999). Boxplot statistical limits as in Figure 2.

Figure 5 – Comparison between 2005 and 2020 macroinvertebrate community indices. IBE= Italian Biotic Index (Ghetti, 1997). Boxplot statistical limits as in Figure 2 BMWP, ASPT, IBE, N families, N taxa, N EPT families, % stenocious taxa

Figure 6 – Graphical result of the NMDS displaying samples based on the diatom matrix.

Figure 7 - Contribution to diatom community β -diversity. a) LCBD 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 as in Figure 2. LCBD= Local Contribution to β Diversity; SCBD= Species Contribution to β Diversity.

Figure 8 - Contribution to macroinvertebrate community β -diversity. a) LCBD for 2005 and 2020; b) SCBD of each taxon; c) LCBD of sampling sites. Boxplot statistical limits as in Figure 2. LCBD= Local Contribution to β Diversity; SCBD= Species Contribution to β Diversity.

Figure 1

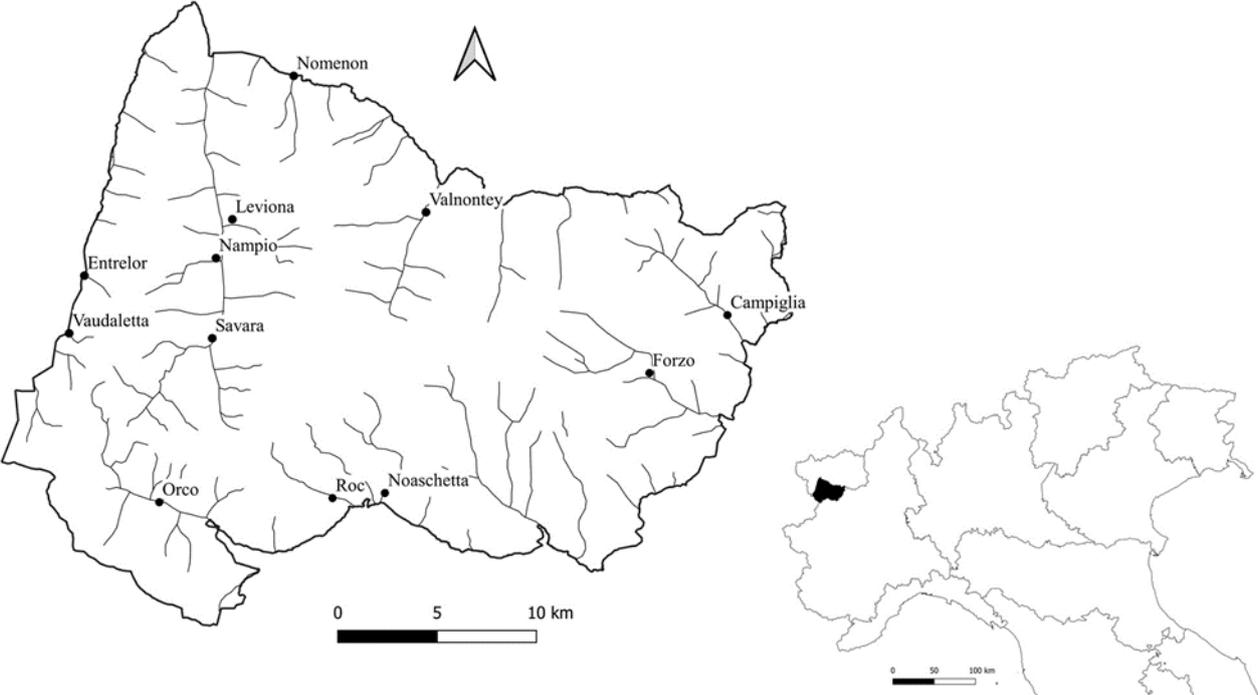


Figure 2

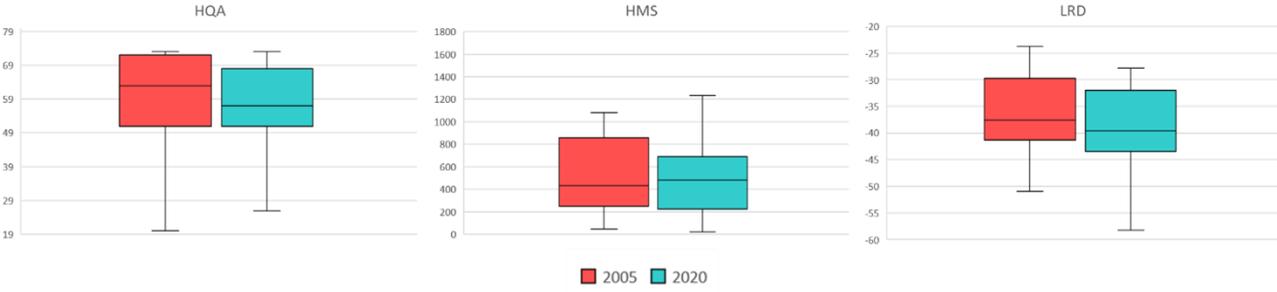


Figure 3

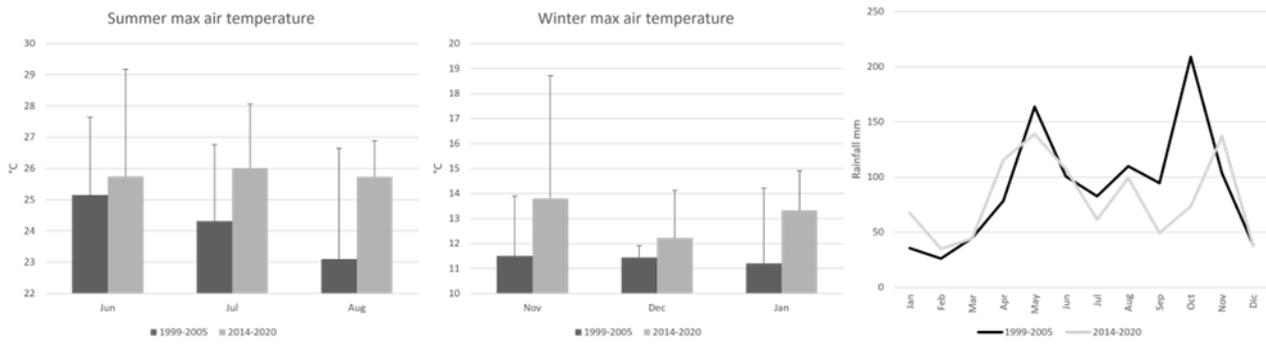


Figure 4

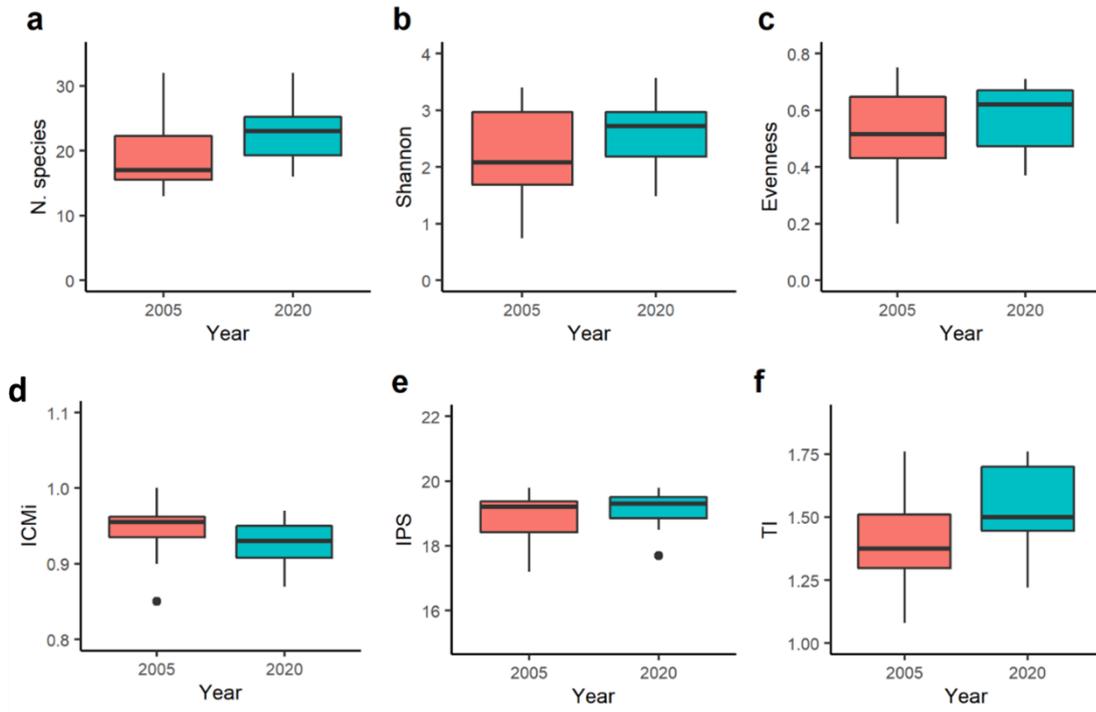


Figure 5

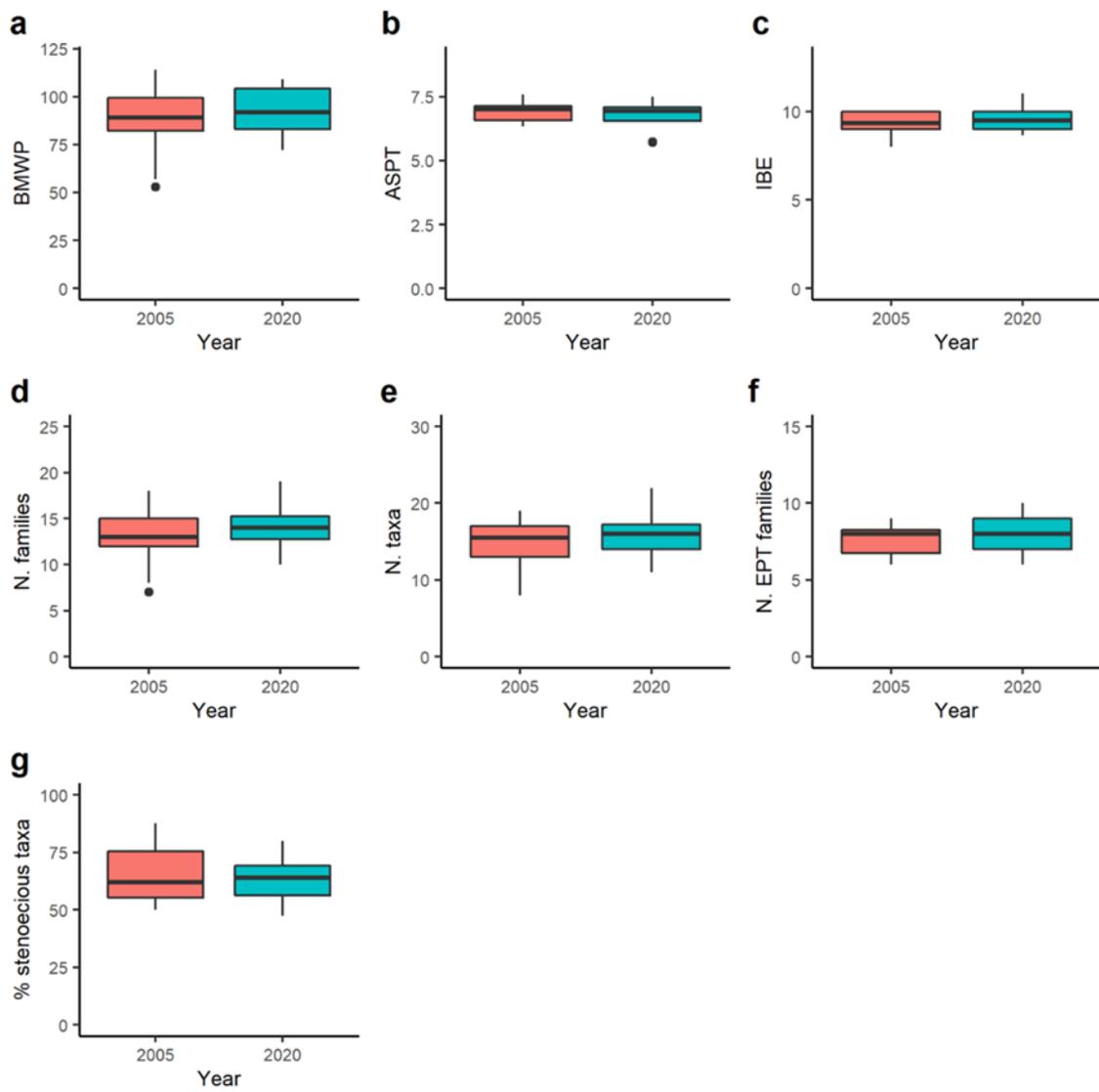


Figure 6

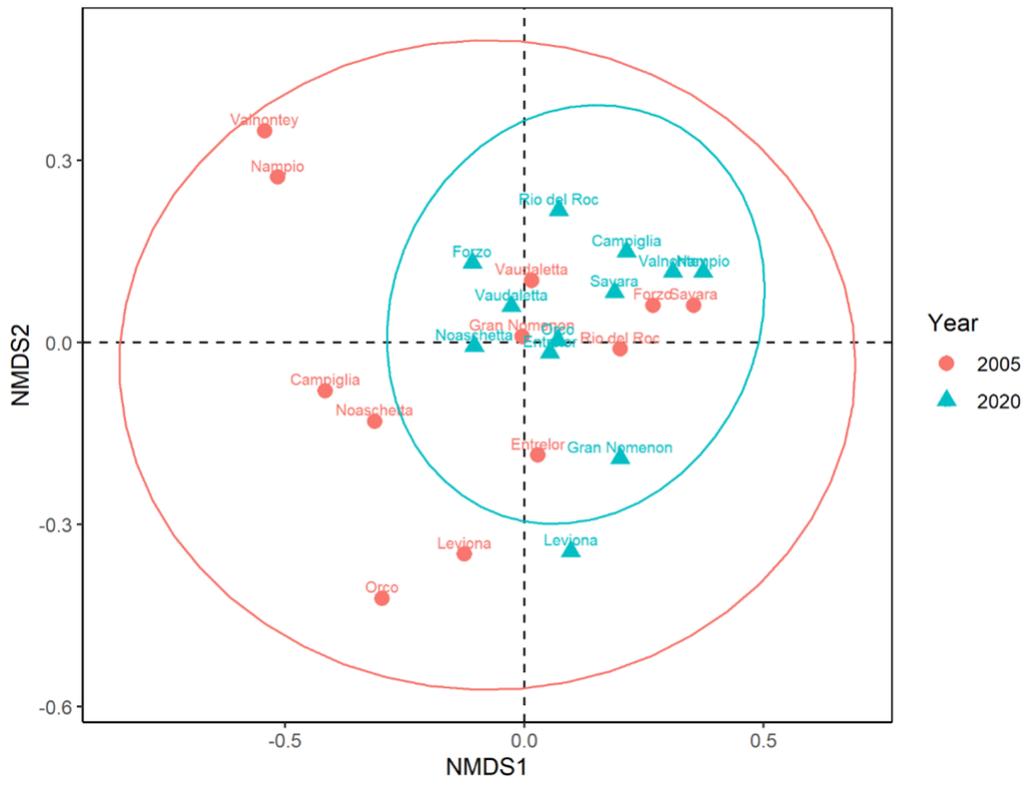


Figure 7

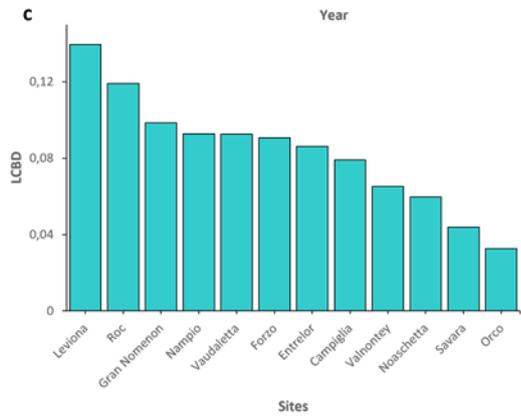
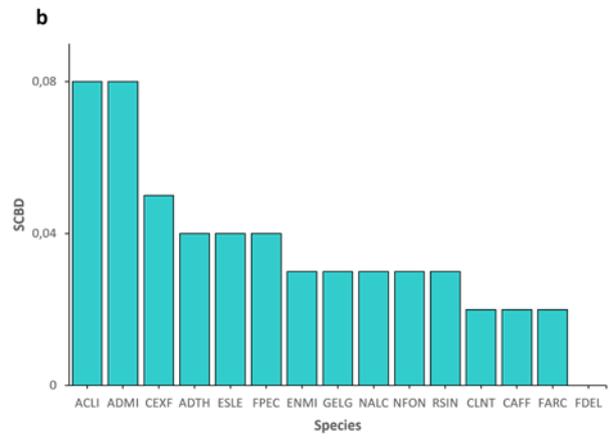
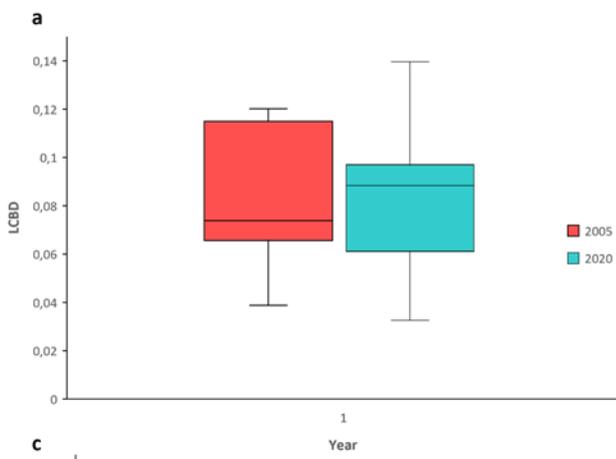


Figure 8

