Effectiveness of artificial floods for benthic community recovery after sediment flushing from a dam

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Abstract

The number of dams is predicted to increase worldwide under the current global change scenario. A major environmental problem associated with dams is the release of large quantities of fine sediment downstream. Therefore, future studies in river conservation will largely be focused on the management of sediments trapped by reservoirs. The aim of this study was to investigate the downstream ecological impacts of sediment flushing from a dam and the effectiveness of artificial flash floods as a recovery strategy. Artificial flash floods have often been employed to remove large amounts of sediment from riverbeds, but their importance in improving the biological quality of lotic environments is almost unknown. We carried out a series of quantitative macroinvertebrate samplings over a 2-year period that started before sediment release and included the artificial flushing events. We characterized the macroinvertebrate community in its structural and functional aspects, and tested the performance of two biomonitoring indexes, comparing their diagnostic ability. Our results demonstrated that sediment flushing significantly altered the structure and composition of benthic communities for more than 1 year. Flash floods exacerbated the overall biological quality, but we believe that this treatment was useful because, by removing large amounts of sediment, the biological recovery process was accelerated. Finally, regarding the water quality assessment, we found that the biomonitoring index for siltation, composed of a selection of taxonomical and functional metrics, was more reliable than the generic one.

Keywords: macroinvertebrates, siltation, dam, multi-metric index, Alpine stream, restoration.

Introduction

Dam and reservoir management is likely to represent one of the biggest challenges in river conservation over the next years. Dams are increasing both in number and in geographical extent (Fox et al. 2016), but numerous aspects of their correct management remain unresolved. In some countries there is a growing tendency to remove large dams (Gangloff 2013; Grant and Lewis 2015), while the current and anticipated changes in hydrological regimes with climate change are causing an increase in dam construction worldwide (O'Connor et al. 2015; Zarfl et al. 2015).

Dams can have a significant ecological impact on lotic ecosystems, disrupting natural flow regimes and altering naturally evolved processes (Van Cappellen and Maavara 2016; Arheimer at al. 2017). For example, dams inhibit natural dispersal pathways and movements of aquatic organisms, causing population isolation and extirpation of migratory fauna (Olden 2016). Dams transform upstream lotic habitats into lentic environments, with a strong impact on structural and functional composition of aquatic coenoses (Gray and Ward 1982; Guareschi et al. 2014; Rothenberger et al. 2017), and interrupt the natural downstream transport of sediments. Moreover, the quality of water released from reservoirs is often significantly altered during reservoir residence (WCD 2000). Dams generally trap all the bedload, i.e. gravel and coarse sand particles, and much of the suspended load, which has detrimental effects on many natural processes, both in downstream and impounded reaches (Brandt 2000). Downstream sections are usually sediment starved causing substrate armoring and increased bank erosion and riverbed incision (Kondolf 1997; Graf 2006), with many detrimental ecological effects. By contrast, in the upstream impounded reach, sediment trapping in reservoirs dramatically alters the morphological and ecological features of the systems, decreasing substrate particle size and transforming the river into a lentic body (Tang et al. 2016; Rothenberger et al. 2017). Moreover, sediment accumulation within impoundments reduces storage capacity of reservoirs, interfering with the functional purpose of the dam and reducing the efficiency of associated hydroelectric power plants (Annandale 2013).

Unfortunately, loss of storage due to sedimentation was rarely considered in the design of many dams, resulting in a widespread problem of reservoir sediment management (Morris and Fan 1998; Owens at al. 2015), with an estimated 28% of the global sediment flux being potentially trapped in reservoirs (Vorosmarty et al. 2003). There are three main strategies for managing sedimentation in reservoirs. The first - sediment yield reduction - aims to reduce sediment inflow to reservoirs by controlling erosion in the upstream river network (Quiñonero-Rubio et al. 2016). The second - sediment routing - a modern approach whereby inflow sediments are allowed to pass through or around the reservoir

(Sumi and Kantoush 2010; Auel and Boes 2011, Martín et al. 2017). Unfortunately, these approaches are not suitable for many existing dams. The last, most common strategy is sediment removal by eliminating accumulated sediment through excavation, dredging, or drawdown flushing (Espa et al. 2013; Kondolf et al. 2014).

Sediment flushing can restore or preserve reservoir storage but often has dramatic negative impacts on freshwater ecosystems. In some cases, sediment flushing causes the release of an excessive quantity of fine material, forming thick layers of sediments, which can settle in the river for years before being transported downstream by the flow, affecting the diversity and abundance of the river biota. The adverse effects associated with the unnatural deposition of fine sediment in the river bed have been documented for different aquatic groups, including macroinvertebrates (Wood and Armitage 1997; Mebane 2001; Bo et al. 2007; Bilotta and Brazier 2008; Jones et al. 2012), fish (Kemp et al. 2011; Sear et al. 2016) and plants (Izagirre et al. 2009; Jones et al. 2014). To accelerate removal of these deposits, dam managers can implement artificial flash floods, i.e. sudden releases of water from the dam for washing away and moving the settled sediments downstream. Artificial floods have been applied for achieving restoration goals, for example, to reproduce the natural flow regime in heavily regulated streams affected by multiple dams in their basins (Robinson at al. 2003, 2004, 2012).

Given the complexity and unpredictability of the flushing operations, data on their impact and effectiveness are scarce. Several operational strategies have been proposed and applied in the last years for successfully desilting reservoirs; for example, it is recommended to perform flushing operations during the period of maximum seasonal run-off to maintain the natural pattern of sediment transport (White 2001; Kondolf et al. 2014). Moreover, flushing-related sedimentation can be mitigated by providing additional water from nearby reservoirs, and stopping sediment release at night to limit the chronic stress associated with these activities (Espa et al. 2013; Quadroni et al. 2016). However, the effects of sediment flushing on aquatic communities are often underestimated due to the lack of pre-impact data and specific biomonitoring tools. We therefore aimed to analyze the impact of sediment flushing from an Alpine river dam on downstream benthic macroinvertebrate communities in terms of richness, density and structure; and to monitor the temporal recovery of these coenoses after two artificial flash floods. Finally, we compared the diagnostic power of two biomonitoring indexes based on macroinvertebrates, one generic and the other designed for detecting siltation effects.

Materials and methods

Study Area

The study was carried out in the Stura di Demonte river (hereinafter referred to as Stura), which is considered one of the least impacted rivers in the entire Italian Alpine area. In fact, the Stura hosts populations of endangered fish species, such as marble trout (*Salmo marmoratus*), grayling (*Thymallus thymallus*) and European bullhead (*Cottus gobio*). For this reason, the SCI (Sites of Community Importance) Stura di Demonte (Code IT1160036) was established according to the EU 43/1992 Directive ("Habitat Directive") to protect the most conserved river stretch, which includes four municipalities between the villages of Demonte and Roccasparvera. The study area was located downstream of the Roccasparvera Dam (44.340256°, 7.440539°; Fig. 1), which was built in 1957 and subsequently acquired by the Italian power company, Enel Green Power, in March 1963. Roccasparvera is a 13 m concrete overflow dam, with a slightly arched planimetric course and a total capacity of 0.580 x 10^6 m^3. The land use of the watershed is dominated by forests (48%), followed by pastures (39%) and rocky areas (10.8%), while the urbanized and agricultural areas account for only for 1.4% and 0.8%, respectively.

Sedimentation and dam operations

Due to a safety-related emergency, in January 2016, a sudden and rapid depletion in the volume of water in the reservoir was necessary. On 7th January at 10:15 AM, the sluice gates on the dam were opened, allowing the water to flush at 5.4 $m^3\ s^{-1}$; the discharge was progressively increased to 13.7 $m^3\ s^{-1}$ (at 02:45 PM), and was successively reduced until the reservoir was empty at the end of the flushing event on 8th January 2016 at 02:00 AM (4.5 $m^3 s^{-1}$). During flushing, water turbidity, suspended solids concentration (SSC), water temperature, dissolved oxygen (DO), pH, and electric conductivity were monitored continually by Enel Green Power at the station ROC1 (44.338820°, 7.448782°), located 0.7 km downstream. This event produced a significant environmental impact, with a massive layer of sediments being accumulated for almost 8 km downstream of the dam (personal observations). These sediments filled the river bed of the Stura and numerous irrigation canals, completely transforming their morphology. Enel Green Power and the local environmental authority (Province of Cuneo) agreed on a recovery strategy aimed at accelerating the removal of sediment from the impacted reach. Since such a large amount of sediment would have taken many years to disperse with normal flows, sudden water releases (artificial flash floods) were implemented on two occasions (18th and 25th May 2016; Table 1), which increased erosion and transport in the downstream section. In both circumstances, they started at 10:30 AM and consisted of a progressive increase of released water every 5 minutes: from 10 $m^3 s^{-1}$ to 41 $m^3 s^{-1}$. This latter condition was kept constant until 01:00 PM and then the volume of sluiced water was progressively reduced.

Sampling design

Benthic macroinvertebrate samples were collected in ten surveys from December 2015 (i.e., pre-impact) to December 2017 (Table 1). Ten quantitative samples were collected randomly in each survey, using a Surber sampler (250 μm mesh size; 0.062 m^2 area). Organisms were counted in the laboratory and identified to the family level using taxonomic keys (Campaioli et al. 1994; 1999).

Statistical analysis

Statistical analysis was performed using R (R Development Core Team 2017). Changes in the composition and structure of benthic invertebrate communities throughout the study were visually evaluated using Principal Coordinate Analysis (PCoA). A Bray-Curtis similarity matrix was used for multivariate ordination of the samples, while a Permutational Analysis of Variance (PERMANOVA) was applied to test significant effects of the "sampling session" on the composition and structure of benthic invertebrate communities. These analyses were carried out with the functions "vegdist", "betadisper" and "adonis" of the R package *vegan* (Oksanen et al. 2015). Univariate analysis was then performed to detect the specific impacts of the sediment release and the temporal recovery of macroinvertebrate communities. Four different community metrics were considered, namely total taxa richness, total density of macroinvertebrates (Ind m^{-2}), EPT (Ephemeroptera, Plecoptera and Trichoptera) richness and EPT density. Each Surber sample was considered as a replicate, and prior to performing regression models outliers were removed using the method of Zuur et al. (2010) for data exploration. Generalized Additive Models (GAMs) were employed to test the non-linear response of the selected community compared to time in days from the sediment release; for this operation, negative values were assigned to the pre-impact data (9th December 2015). All the GAMs were carried out using the "gam" function of the *mgcv* R package (Wood and Wood 2015) and applying a Poisson distribution. A negative binomial distribution was then applied in case of overdispersion.

Finally, our data were analyzed in the context of river quality assessment, comparing the performance of two selected indexes, namely the STAR_ICMi and the multi-metric index proposed by Doretto et al. (2018). The former was originally developed as a calibration tool in the context of the Water

Framework Directive (EU 60/2000 Directive), and is currently the official biomonitoring index provided by Italian regulations to classify the ecological status of running waters. STAR_ICMi is a multi-metric index composed of six community metrics, belonging to three categories (i.e. diversity, abundance, and sensitivity/tolerance). The weighted values of these metrics are aggregated in the STAR_ICMi index, and the final score is obtained by the ratio between the observed and the reference value (for details see Bo et al. 2017). The latter instead represents a recently proposed multi-metric index using benthic invertebrates designed to detect the impacts of river siltation. This index includes two diversity metrics (i.e., total taxa and EPT richness) and a functional metric (i.e., the abundance of taxa preferring shallow, fast-flowing water with coarse substrates: Ecological Group A *sensu* Usseglio-Polatera et al. 2000). Compared to the STAR_ICMi, which is generic, the multi-metric index is a stressor-specific tool designed to assess the effects of siltation (for further details see Doretto et al. 2018). These indexes were calculated for each sampling occasion, pooling together the ten macroinvertebrate samples collected on that date. The temporal trends of the selected indexes were thereby compared to evaluate which method provided the best performance in relation to the sediment release.

Results

Sediment and water chemistry parameters

The flushing operations dramatically altered the physical and chemical parameters of the water, with the greatest changes occurring during the night between 7th and 8th January 2016 (Fig. 2). An exponential increase in SSC was documented; the SSC varied from 0.00 before the flushing to 10 mg L^{-1} at the end of the operation (mean = 12.07 mg L^{-1}), with intermediate peaks. The highest level of SSC occurred on 8th January from 00:00 AM to 10:00 AM; during this period the SSC was consistently higher than 20 mg L^{-1}, reaching a peak of 31.05 mg L^{-1} (Fig. 2a). A very similar trend was also observed for the turbidity, as this parameter was highly correlated with the SSC. From the onset to the end of the flushing operations, the water turbidity increased from 0.80 to 2160 NTU, reaching a maximum value of 6209 NTU (mean = 2413 NTU). During the most intense phase of flushing, the turbidity was higher than 4300 NTU (Fig. 2b). Water temperature varied from 3.80 to 5.28 °C (mean = 4.26°C) but a sharp decline was recorded between 7:15 PM and 09:15 PM, when the water temperature dropped to 0.26 °C, but then abruptly increased again (Fig. 2c) possibly due to a thermal stratification of the impounded reach. The DO concentration markedly reduced during the monitored period from 12.50 mg L^{-1} at the beginning of the operations to 5.01 mg L^{-1} at 02:00 AM (Fig. 2d). DO then slowly increased again, reaching a value of 7.09 mg L^{-1} at the end of the operations (mean = 8.90 mg L^{-1}). A similar temporal trend was also observed for the pH (Fig. 2e), which varied from 8.20 to 7.14 (mean = 7.70). Around 01:00 AM, the pH dropped to 7.17 and then remained relatively constant until the end of the flushing activity. Finally, marked oscillations were observed in relation to the electric conductivity over the monitored period (Fig. 2f). This parameter varied from 447 µS cm^{-1} at the beginning of the operations to 483 µS cm^{-1} at the end (mean = 421 µS cm-1). However, the lowest and highest values were 225 and 502 µS cm^{-1}, respectively.

Benthic invertebrates

We collected 7,860 organisms belonging to 38 families from nine different systematic groups. Diptera and Trichoptera were the orders with the highest number of families, nine and eight, respectively, whereas Gastropoda, Crustacea and Triclada were each represented by only one family. The number of taxa per sample ranged from 1 to 21 (mean = 9), while the total density of benthic invertebrates ranged from 40 to 8440 Ind m^{-2} (mean = 1638 Ind m^{-2}). Mayflies - Heptageniidae and Baetidae - as well as dipteran larvae of the families Chironomidae and Simuliidae were the most abundant taxa.

PCoA ordination of the samples depicted a clear temporal shift in the composition and structure of benthic invertebrate communities (Fig. 3) and PERMANOVA results showed a significant effect of the sampling occasion ($F_{9,86} = 9.9$; $P < 0.001$). Compared to the pre-impact situation (i.e. 2015.12.09), the communities just after the sediment release were significantly different in terms of composition and structure, and oriented at the right side of the plot. The macroinvertebrate communities slowly approached the pre-impact community levels only after both of the flash floods (from 2016.07.22 onward), reaching full overlap with the pre-flushing community on the last sampling occasion (i.e. 2017.12.12, nearly 2 years later).

Results of the multivariate analysis were confirmed and further strengthened by those of the GAMs (Table 2). The sediment release and consequent flash floods significantly reduced the total taxa richness, which showed a unimodal pattern with a negative peak around 150 days after the sediment release (Fig. 4a). Taxa richness then progressively recovered over time, reaching values comparable to pre-impact levels after 600 days. The combined effects of sedimentation and flash floods also resulted in a similar temporal pattern for the EPT richness (Fig. 4b). Prior to impact, the average number of EPT families was ten, which subsequently dropped to three after the sedimentation event and to one after the flash floods. Again, the results of the model showed a progressive increase in the

EPT richness over time, with an almost complete recovery after 600 days. Conversely, GAMs illustrated very fluctuating patterns in the density of benthic invertebrates. Both the total (Fig. 4c) and the EPT density (Fig. 4d) showed a dramatic depletion after the sediment release as well as the flash floods. However, compared to the taxa and EPT richness, both these two events were followed by peaks in density. A less pronounced decrease was also observed after 400 days, and then both the total and EPT density increased again to nearly pre-impact values. It is interesting to note that these marked temporal fluctuations were mainly driven by the extremely high abundance of Baetidae (Fig. 5a), Chironomidae (Fig. 5b) and Simuliidae (Fig. 5c) specimens during the first post-release sampling occasions (Table 2). Thus, the specific response of these few taxa accounted for the observed results for the total and EPT density.

Biomonitoring indexes

The two biomonitoring indexes for assessing the impact of the sediment release from the Roccaspervera dam gave different responses (Fig. 6). The STAR_IMCi (Fig. 6a) on the pre-impact sampling occasion was 1.1 and then decreased as a consequence of the sedimentation event and flash floods. However, these changes only corresponded to a slight deterioration in the water quality classification, which shifted from High to Good and only dropped to Scarce on the sampling occasion 2016.06.06 (i.e., after the second flash flood, STAR_ICMi = 0.25). After the artificial floods, the index increased, ranging from 0.80 to 1.15, and it reached the High quality class on the last two sampling occasions (i.e. after 475 days). Overall, the STAR_ICMi detected a change in the water quality class, but this only resulted in a shift from High to Good for most of the recovery period, with the exception of the sampling occasion 2016.06.06. Conversely, the temporal variation of the multi-metric index for siltation proposed by Doretto et al. (2018) depicted a different temporal pattern (Fig. 6b). From the pre-impact value of 0.83, this index dropped markedly after the sediment release (0.33) and the flash floods, achieving a minimum value of 0 on the sampling occasion 2016.06.06. The index then increased over time, reaching full recovery on the last sample date. Compared to the STAR_ICMi, this stressor-specific index showed a more pronounced variation between the pre-impact value and that recorded just after the sediment release. Moreover, the temporal pattern of this index was more similar to the community metrics, especially taxa richness.

Discussion

Although several studies have documented the effects of sediment releases from dams, long-term series encompassing pre-impact data and multi-year sampling campaigns, like this study, are rare. Usually, flushing operations are programmed over long periods of time, and carried out by alternating sediment and water pulses between the day-time and night-time, or by providing volumes of diluting water from adjacent reservoirs (Crosa et al. 2010; Espa et al. 2013; 2015). These practices allow minimization of the ecological impacts of excessive releases of sediments and facilitate the recovery of aquatic communities.

Unfortunately, adopting these approaches is not always possible according to the specific features of the dam and the magnitude of the related impacts. The sediment flushing from the Roccasparvera dam, documented in this study, was concentrated in a short period (from 7th January at 10:15 AM to 8th January at 02:00 AM). In just a few hours, drastic alterations of all the main physical and chemical attributes of water were detected; for example, the SSC and turbidity were in the range of three orders of magnitude greater than before the sediment release. At the same time, a dramatic reduction in the concentration of DO and pH was observed. Similar changes have been documented by other authors in relation to flushing events (Gray and Ward 1982; Peter et al. 2014; Khakzad and Elfimov 2015, Espa et al. 2016) and have led to a strong impact on benthic invertebrate assemblages. We found a temporal shift in the composition of communities; pre-impact macroinvertebrate communities were mainly composed of EPT taxa, while after the sediment release and flash floods, the number of taxa belonging to these orders was significantly reduced. Our results corroborate other studies carried out in the Alpine area (Espa et al. 2013; Peter et al. 2014; Martín et al. 2017). More profound effects were evident when macroinvertebrate density was considered, with a collapse of the total and EPT density due to the sediment release, which was exacerbated with the flash floods. Similar results were also observed by other authors (Doeg and Khoen 1994; Quadroni et al. 2016); for example, Crosa et al. (2010) studied the ecological consequences of a reservoir flushing in an Italian alpine stream. They found the density of invertebrates dropped, on average, from 2000 to 100 Ind m^{-2}. In our study, 2 years were necessary to achieve full recovery of the benthic communities to their pre-impact taxa richness and density of macroinvertebrates. However, taxa richness recovered slowly and progressively, while there was a fluctuating response of density. The extremely high abundance and dominance of few generalist taxa, such as Baetidae, Chiromomidae and Simuliidae, may account for the temporal pattern observed for density. Several authors have demonstrated that these families are early colonists and tend to be dominant during the first phases of post-disturbance recolonization, including experimental floods from reservoirs (Otermin et al. 2002; Robinson at al. 2004). Compared to our results, a faster post-flushing recovery of macroinvertebrate communities has been documented in other studies (Crosa et al. 2010; Espa et al. 2013; 2016). This delayed recovery pattern could be explained by the fact that the flash floods were performed 5 months after the sedimentation event. However, we postulate that, in the long-term, these artificial treatments contribute to restoring the natural habitat for benthic invertebrates, facilitating the recovery of communities.

Another possible factor that could account for the full community recovery is the overall pristine conditions of the investigated stream, which was reflected by the excellent pre-impact ecological status of the sampled reach. As expected, the sedimentation process produced a deterioration of the water quality, but this mainly consisted in a shift from High to Good, even when the river bottom was buried by a large amount of fine sediment (personal observation) and the density and richness of benthic invertebrates were highly reduced. The water quality further dropped to Scarce as an immediate consequence of the flash floods. Although these operations were successful in removing the fine sediment and promoting long-term recovery, they initially induced stressful conditions for benthic invertebrates, dislodging them from the substrate. This may account for the low water quality on that occasion, after which the water quality quickly re-established to Good and then High. In particular, despite the temporary worsening of the biological quality after the flash floods, we believe that this treatment had a considerable importance overall, because large amounts of sediment were removed, which would have taken much longer to disperse naturally, thus the biological recovery process was accelerated. The limited performance of STAR_ICMi was not completely unexpected since it is a generic multi-metric index, as confirmed by other authors who obtained similar results when assessing the impacts of sedimentation from dams (Espa et al. 2015; 2016). On the contrary, the performance of the index for siltation recently proposed by Doretto et al. (2018) offered interesting results, although no ecological classification has been provided yet. Nevertheless, when looking at the temporal trend of the index, it was more reliable than the STAR_ICMi in detecting the effects of disturbance in relation to both the sedimentation release and the flash floods. Quadroni at al. (2016) evaluated the consequences of sediment flushing in an Alpine stream and stated that the STAR_ICMi can underestimate impacts, resulting in an imprecise water quality assessment. This is the first time that our stressor-specific index has been tested using an independent dataset. Overall, its performance offered promising applications and confirmed the useful role of macroinvertebrates as bioindicators for running waters (Rosenberg and Resh 1993; Buss et al. 2015). Moreover, the combination of taxonomical and functional metrics considered here appears very successful for detecting the specific effects of siltation, and this supports the adoption of functional groups for river biomonitoring (Merritt et al. 2015).

While several solutions have been proposed in relation to the timing of the sluiced water and the amount of flushed fine sediment, such as thresholds in the SSC and discharge, the development of stressor-specific methods based on the biota response still remains unexplored (Crosa et al. 2010;

Quadroni et al. 2016). Our report therefore offers the application of a diagnostic biomonitoring tool designed to evaluate the specific impacts associated with siltation. Since the number of reservoirs and dams is predicted to increase in the near future (Fox et al. 2016; Van Cappellen and Maavara 2016), developing science-based biomonitoring and management tools is an important challenge for river ecologists and managers.

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Tables

Table 1 Sampling scheme: Date = date, Label = labels used in the statistical analysis (year.month.day), Activity = data and information associated to each date

Date	Label	Activity
9th December 2015	2015.12.09	Pre-impact data
7th - 8th January 2016	-	Sediment flushing
23rd March 2016	2016.03.23	Benthic invertebrates sampling
17th May 2016	2017.05.17	Benthic invertebrates sampling
18th May 2016	-	First flash flood event
24th May 2016	2016.05.24	Benthic invertebrates sampling
25th May 2016	-	Second flash flood event
06th June 2016	2016.06.06	Benthic invertebrates sampling
22nd July 2016	2016.07.22	Benthic invertebrates sampling
09th December 2016	2016.12.09	Benthic invertebrates sampling
15th March 2017	2017.03.15	Benthic invertebrates sampling
27th April 2017	2017.04.27	Benthic invertebrates sampling
12th December 2017	2017.12.12	Benthic invertebrates sampling

Table 2 Statistics of the Generalized Additive Models: Est = estimate, SE = standard error, z = z-value, P = p-value, X^2 = Chi-square

Metric	Intercept				Smooth	
	Est	SE	z	P	X^2	P
Taxa richness	2.124	0.037	57.080	< 0.001	146.900	< 0.001
Total density	6.721	0.047	143.900	< 0.001	698.100	< 0.001
EPT richness	1.426	0.054	26.530	< 0.001	122.900	< 0.001
EPT density	5.781	0.047	121.900	< 0.001	893.600	< 0.001
Baetidae density	4.118	0.054	76.180	< 0.001	821	< 0.001
Chironomidae density	4.980	0.048	102.900	< 0.001	915	< 0.001
Simuliidae density	3.667	0.053	68.790	< 0.001	1173	< 0.001

Figure captions

Fig. 1 Area of study: the Stura di Demonte watershed and the investigated stream reach

Fig. 2 Values of: **a** concentration of suspended solids (SSC), **b** turbidity, **c** water temperature, **d** concentration of dissolved oxygen (DO), **e** pH and **f** conductivity measured at the ROC1 station during the sedimentation release from the Roccasparvera dam

Fig. 3 Principal Coordinate Analysis (PCoA) ordination plots based on the dissimilarity matrices (Bray-Curtis). Symbols represent the benthic community samples collected in each sampling occasion (year.month.day). The grey lines link each sample with its corresponding centroid (black dots)

Fig. 4 Representation of the GAM smoother applied to time interacting with the community metrics: **a** total taxa richness, **b** EPT richness, **c** total density and **d** EPT density. The black line represents the smoothed function (i.e. s(Time, degrees of freedom)), while the grey area indicates the 95% confidence interval. Black vertical line indicates the sediment flushing, while the dotted vertical lines represent the two artificial floods. Grey tics on the x-axis represent the sampling occasions

Fig. 5 Representation of the GAM smoother applied to time interacting with the density of: **a** Baetidae, **b** Chironomidae and **c** Simuliidae. The black line represents the smoothed function (i.e. s(Time, degrees of freedom)), while the grey area indicates the 95% confidence interval. Black vertical line indicates the sediment flushing, while the dotted vertical lines represent the two artificial floods. Grey tics on the x-axis represent the sampling occasions

Fig. 6 Bars indicate the values of **a** the STAR_ICM index and **b** the sedimentation index (Doretto et al. 2018) calculated for each sampling occasion (year.month.day)