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Sustainable management of hydraulic diversions in fluvial ecosystem

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It is one thing to find fault with an existing system.
It is another thing altogether, a more difficult task,
to replace it with another approach that is better.
-Nelson Mandela, 16 November 2000
(speaking of water resource management)

Abstract

This thesis focuses on the study of the negative effects induced on stream ecosystems by hydropower schemes in an Alpine river system. The downstream ecological consequences of 4 controlled free-flow flushing operations designed to remove sediments accumulated in an Alpine reservoir are described. Sediment removals were carried out for analogous duration and in the same period of the year from 2006 to 2009. Brown trout populations were remarkably reduced after the beginning of flushing operations, but appeared to reach equilibrium with the new environmental conditions. Macroinvertebrate fauna, despite exhibiting a significant reduction in abundance and biomass following each flushing, showed a quick recovery within 3 months. Nevertheless, the yearly occurrence of sediment flushing changed the taxa composition, allowing species with fast life cycle and good colonizing ability to become dominant. Particle size analysis of core samples collected in riffle habitats provide evidence of a significant increase in interstitial fine sediment, that could adversely affect the recovery processes. Maximum allowable SSCs of 10 g L^{-1}(daily average) and 5 g L^{-1}(overall average) for flushing operations carried out in small to medium high-gradient Alpine streams are recommend. In the same river network, a data set (n=30) of free-flowing and altered flow streams was used to detect the effects of water diversion on macroinvertebrate communities using single and multimetric indices as well multivariate approaches. Although macroinvertebrate diversity decreased significantly in response to reduced flows, the results suggest that invertebrate-based metrics may be poor descriptors of the magnitude of flow reduction. A consistent pattern in the dataset was demonstrated through multivariate analysis by groups of taxa that were respectively reduced or increased in abundance at the impaired sites, confirming that diversions are potentially responsible for a change in macroinvertebrate composition.

The findings of this research showed how both activities have the potential to alter biological communities, but even that could be managed together to minimize risks. Considering obtained results this project will help support current and future research of Alpine stream ecosystems and will provide information for a management of freshwater resource balancing economical, technical and environmental issues.

Contents

CHAPTER 1

Introduction

1.1 General introduction

Existing and projected future increases in water demands have resulted in an intensification of the complex conflict between the exploitation of rivers as water and energy sources and their conservation as biologically, integrated ecosystem (Tharme, 2003). Hydropower represents one of the most important energy resources in Italy, contributing about to 16% of total gross electricity production in the country. Although the numerous benefits of hydropower generation as renewable energy source (i.e. no direct waste and CO_2 emission), at the local scale its impact on freshwater ecosystems may be severe (Fette et al., 2007). Dams and diversions, in fact, severely change the natural flow regime of watercourses, mainly reducing the downstream discharge and eliminating the natural flood peaks. Management of hydropower is particularly relevant in the Alps, since the abundance of hydropower facilities has already affected most of Alpine watersheds (Truffer et al., 2001; Meile et al., 2010). Sediment management practices, aimed at preservation of storage volumes, further increase human pressures on Alpine river ecosystems. Indeed, sustainability of Alpine reservoirs is severely threatened by sedimentation resulting from natural geomorphological processes (Hartmann, 2004) and measures to remove accumulated material are often necessary to maintain the storage capacity and/or grant the efficiency of the installation (Brandt, 2000a). Failing to do so, it results in the loss of storage capacity, consequently the loss of flood protection abilities, water supply reliability and hydropower generation potential (Hartmann 2004). Knowledge of the impacts resulting from the alteration of sediment and flow regime is thus fundamental for an integrated and sustainable management of water resources in Alpine regions. Due to their relevance and close connection (i.e. watercourses downstream a dam are subjected to the joint effect of flow regulation and sediment release operations), this thesis discuss both the arguments. In particular, chapter 2 analyzes the effects of sediment removal operations from reservoirs on instream habitat and biota, whilst in chapter 3 the effects on macroinvertebrate community of reduced flow downstream diversions are investigated. The present study has been conducted in the province of Sondrio (Nothern Italy), a mountainous area of the Central Alps largely exploited for hydropower production, as part of the regional project *Sediment Removal from Artificial Reservoir: Impact on Fish Fauna and Evaluation of Protection Strategies*, concluded in 2007, and of the EU Interregional project ECOIDRO, *Water Use and Safeguard of Environment and Biodiversity in the River Basins of Adda, Mera, Poschiavino and Inn*.

1.2 Mountain stream ecosystems

Mountains are a key source of water for human consumption and economic use (irrigation, hydropower and industry) both within mountain regions and in downstream lowlands. Although mountain streams drain en estimated 20% of global land area, they provide water to about 50% of the world's population (Bandyopadhyay et al., 1997) and contribute nearly 50% of the sediment that river carry to the global oceans (Milliman and Syvitsky, 1992). Furthermore, mountains are home to a substantial portion of the planet's biodiversity of species and ecosystems (Price, 1999).

Stream ecosystems are regulated by features and processes occurring at a range of spatial scales. At the largest scale, climate, geomorphology and land use control, through top-down processes, channel morphology, stream hydrology, thermal regime, water chemistry and biotic community structure. At finer scales, the availability of suitable habitat and food resources, as well as species interactions regulate organism populations by means of bottom-up processes. Few other ecosystems possess either the frequency or intensity of environmental changes that are observed in running waters, which makes disturbance, mainly in the form of flood, a dominant factor of community organization in streams (e.g. Power and Stewart, 1987; Resh et al., 1988; Death and Winterbourn, 1995; Death, 1996; Lake, 2000; Death, 2002).

Stream character in high mountain catchment is strongly influenced by the prevailing water sources (glacial, non-glacial) and the origin of major tributaries (Füreder, 2007). Although highland streams share common features such as steep gradient, high turbulence and dissolved oxygen concentration, different water sources (glacial, snowmelt, rain and groundwater fed) produce a characteristic discharge regime and a distinctive suite of physical and chemical characteristics (Brown et al., 2003). Accordingly, between the permanent snowline and the treeline, three general types of stream ecosystems have been identified (Milner and Petts, 1994; Ward, 1994; Malard et al., 1999; Füreder et al., 2001): the kryal, or glacier-melt dominated system; the rhithral, or seasonal snowmelt-dominated system; and the krenal, or spring-fed system. Although useful to characterize the stream stretches draining alpine horizon, this classification loses importance far downstream, where streams become more comparable to rain-fed water courses.

In contrast to terrestrial ecosystems, where in-situ primary production is the dominant energy source for consumers, biologically available energy in lotic ecosystems originates from two sources: primary production occurring within the wetted channel (autochthonous energy) and terrestrial organic matter entering the stream by aerial litterfall, lateral transport, bank erosion, or exfiltrating groundwater (allochthonous energy). The relative importance of autochthonous and allochthonous production changes along the river continuum depending on stream size and riparian vegetation (Vannote et al., 1980; Minshall et al., 1985) and across biomes (e.g. Bott et al., 1985; Mulholland et al., 2001). The energy base of headwater streams in forested catchments is mainly leaf litter from the woody riparian vegetation (Cummins et al., 1973; Graça, 2001), whereas autochthonous production is expected to be the dominant energy source of most alpine streams (Ward, 1994). Riparian vegetation above the tree line or in proglacial areas is typically sparse. Therefore, direct and lateral inputs of terrestrial organic matter is expected to be very small (Ward, 1994). If slopes are steep, the riparian vegetation changes within short distances from “rock desert” to subalpine forests (Ormerod et al., 1994). This change in terrestrial vegetation is assumed to be paralleled by an increasing input of allochthonous organic matter and thus, reduced autochthonous dominance in ecosystem energetics.

1.3 Impacts of hydropower production

Although the type and severity of human-generated stressors affecting freshwater ecosystem differ on a worldwide scale, alteration of physical habitat associated with generating hydroelectric power is one of the major drivers affecting the integrity of aquatic ecosystems in Alpine mountain areas. At present no single major river in the Alps flows for its entire course in a natural condition and fewer than ten alpine rivers exist whose courses are uninterrupted for more than 15 km (Baetzing and Messerli, 1992). Despite the numerous benefits of hydropower production, this renewable energy source can have serious negative consequences on the ecological integrity of freshwater ecosystems, essentially connected to the combined actions of sediment trapping and alteration of natural flow patterns (particularly flood). In many developed countries river damming is now recognized to be excessive (Graf, 1999).

A vast body of scientific research has accumulated supporting a natural flow paradigm (*sensu* Poff et al., 1997), where the flow regime of a river, comprising the five key components of variability, magnitude, frequency, duration, timing and rate of change, is recognized as central to sustaining biodiversity and ecosystem integrity (Poff and Ward, 1989; Karr, 1991; Richter et al., 1997; Rapport et al., 1998; Rosenberg et al., 2000). Indeed, flow is a major determinant of physical habitat in streams which in turn is a major determinant of biotic composition (Bunn and Arthington, 2002). Changes in stream depth and width, current speed, sediment load, temperature and water quality connected to the alterations of streamflow, can in turn influence availability and suitability of habitat for aquatic biota (Stanford and Ward, 1979; Armitage, 1984; Baran et al., 1995). The magnitude of impact on aquatic fauna depends on the type of flow management and the extent of departure from the natural flow regime. Impacts range from low for run-of-the-river operations to high for hydropeaking operations. Peaking hydropower meet daily variations in electricity demand by allowing water to flow though turbines only at certain times, usually from mid-morning through early evening. Altering water level with increased frequency and rates of change that differ markedly from the natural flow regime, create unstable habitat conditions that can be especially disruptive to juvenile fish and limit spawning opportunities for adults (Freeman et al., 2001). Macroinvertebrate too are vulnerable to rapid diurnal changes in flow (Moog, 1993; Valentin et al., 1995; Bosco Imbert and Perry, 2000; Bruno et al., 2009) and regulated reaches below hydroelectric dams with erratic flow pattern are typically characterized by species-poor macroinvertebrate communities (Munn and Brusven, 1991). Just as increased instability can reduce the functioning of mountain streams, increased stability can also be detrimental to physical and biological diversity (Reice,1994). Loss of seasonal flood peaks, as a result of the regulation of flow, for example, removes cues for spawning or release of plant seeds and may favor the proliferation of specific taxa (Munn and Brusven, 1991). Dams may disrupt the longitudinal continuity of physical and biological features predicted by the river continuum concept (*sensu* Vannote et al., 1980). Numerous species of fish and aquatic invertebrates rely on ability to move up- and downstream to complete their lifecycle (Stanford and Ward, 1993; Gomi et al., 2002), and longitudinal connectivity is important to nutrient cycling (Newbold et al., 1982). Similarly, many species of riparian plants rely on water transport of propagules (Nilsson et al., 1991; Nilsson and Jansson, 1995; Merritt and Wohl, 2006).

Sediment transported from mountains headwater exerts an important control on downstream channel stability and increases or decreases in sediment supply can destabilized downstream channel segments (Wohl, 2006). Because of the loss of the river normal sediment load, the result of deposition in the slow waters of the impoundment, the discharge immediately below a dam is "sediment- starved". If flows have a greater transport capacity than the amount of sediment being supplied, release discharge may cause scouring of fine material and armoring (coarsening) of the stream bed, a process in which the surface substrate becomes tightly compacted. This can lead to substantial channel and bank erosion and down-cutting of the streambed as the river adjust to the altered balance between the amount of water and sediments that is transporting (Williams and Wolman, 1985; Parker and Sutherland, 1990; Richards and Clifford, 1991; Kondolf, 1997; Brandt, 2000b; Vericat et al., 2006). Where fine sediment is available, the absence of flushing flows can result in their accumulation within the streambed (Sear, 1995; Stevens et al., 1995), which in turn affect the quality of stream habitat for fish and invertebrates. A number of negative effects are reported in literature connected to excessive fine sediment in streambed or clogging. In general, the deposition of fine sediment in stream ecosystems is detrimental to aquatic organisms because of reductions in streambed substrate composition, permeability, and stability (Young et al., 1991; Cobb et al., 1992; Haschenburger and Roest, 2009). These alterations in the physical environment can decrease egg-to-fry survival rates in fish (McNeil, 1966; Buermann et al., 1995; Argent and Flebbe, 1999; Soulsby et al., 2001; Haschenburger and Roest, 2009;) and can affect stream and benthic macroinvertebrate production and periphyton communities (Erman and Erman, 1984; Noel et al., 1986; Waters, 1995). Alteration in the quality and quantity of deposited sediments can affect the structure and function of benthic macrofaunal communities determining for example the loss of microhabitat (Rae, 1987), the loss of access to trophic resources (Lenat et al., 1981), the damage to respiratory systems of organisms (Lemly, 1982), or changing the ecological processes that are on the basis of autochthonous and allochthonous sources of energy of the system (Quinn et al., 1992).

1.4 Sediment management of Alpine reservoirs

Sediment deposition in reservoirs reduces storage capacity, and poses risks of blockage of intake structures as well as sediment entrainment in hydropower schemes. The use for which a reservoir was built can be sustainable or represent a renewable source of energy only where sedimentation is controlled by adequate management, for which suitable measures should be devised. Accepted practice has been to design and operate reservoirs to fill with sediment, generating benefits from remaining storage over a finite period of time. The consequences of sedimentation and project abandonment are “left” to the future. These consequences can be summarized as: sediments reaching intakes and greatly accelerating abrasion of hydraulic machinery, decreasing their efficiency and increasing maintenance costs; blockage of intake and bottom outlet structures or damage to gates that are not designed for sediment passage. This ‘future’ has already arrived for many existing reservoirs and most others will eventually experience a similar fate, thereby imposing substantial costs on society (Palmieri et al., 2001).

Options to manage sedimentation can divided in three common ways (White, 2001): minimizing sediment loads entering reservoirs, minimizing deposition of sediments in reservoirs and removing accumulated sediments from reservoirs. Methods to reduce sediment input include catchment conservation programmes and engineering measures to control erosion, upstream trapping of sediments and bypassing of high sediment loads. For alpine ranges these methods are limited. Conservation programmes by cultivating vegetation can only be applied in lower elevation areas with moderate temperatures while in upper parts changes of the slopes by forming terraces may reduce soil erosion. Upstream trapping faces the problem that the sediment supply from the mountain slopes is not limited and fills such constructions in the tributaries shortly. As a consequences they have to be managed, too, to preserve storage volume for high flow situations with the risk of sudden sediment peaks. The third category of measures to reduce sediment input focuses on high flow situations with high sediment content and their bypassing around the reservoir using channel or tunnels. The first two options disturb the natural sediment transfer system if they are not just counterbalancing human impacts. The possible lack of sediments in downstream sections of the river basin may cause erosion problems. Bypassing is limited to short term events and simulates natural conditions in the downstream reach avoiding severe disturbances of the ecosystem.

Reduction of sediment deposition by facilitating flood sediment-laden flows to pass through reservoirs before settlement, deposition and consolidation of sediments have chance to occur, is one of the most effective and economic in some cases ways to preserve the storage capacity. Singular events in the alpine environment may carry extreme amount of sediment in a very short time causing remarkable loss of storage capacity. The sediment sluicing and the venting of turbidity currents (venting) are the two main techniques that can be classified as such methods. Sluicing consists on lowering the water level during flood events and opening the lower outlet at the dam wall. In this way the velocity of the water will increase and the transported material will not deposit. In some cases it is possible to use the phenomena of turbidity currents to avoid sedimentation during flood events. Once a density currents occur in a reservoir they move gravitationally along the basin toward the dam site and can be effectively vented through the lower outlets. This mechanism is one of the major driving forces for sediment movement in alpine lakes as most conditions for turbidity currents are fulfilled (i.e. high concentration of suspended material entering the reservoir, significant water depth at the inlet of the lake, marginal flow velocities in the water body, steep bottom slope of the reservoir, narrow geometry of the lake) (Hartmann, 2004).

Methods to remobilize already deposited material can be divided into operational options and technical measures. Operational measures focus on the relocation of sediments from the reservoir into the downstream reach as suspended load. In most mountain

hydropower reservoirs the cheapest and the easiest way (from a technical point of view) seems to be flushing operation of the reservoir (Gvelesiani and Shmal'tzel, 1971).
Flushing is a technique which, by using a suitable combination of the drawdown (water level lowering) and increased flow in the reservoir, allow previously deposited sediments to be discharges from the reservoir basin into the downstream river. Flushing is undertaken over a relatively short period – usually a few days or weeks and would typically be annual. Flushing may be undertaken with the reservoir effectively empty (empty or free-flow flushing) so that the riverine conditions are established or with the reservoir partially drawn down (pressure flushing) (White, 2001). To be able to erode depositions, especially if consolidated, more time and water in needed compared to sluicing operation. While sluicing and other 'routing techniques' results in the seasonal pattern of sediment outflows largely following the pattern of sediment inflows, flushing typically compresses the annual sediment load which may occur over two or three months into a few days or weeks. Additionally long lasting flushing operations with high sediment concentrations affects the ecosystem downstream. A more moderate method to transfer deposited sediments downstream uses artificial remobilization and discharge of the sediment laden flow through the turbines of the power plant. The concentration of the sediments in the downstream section can be limited by the intensity of remobilization to meet ecological requirements and interest of stake-holders. However some modifications at the turbine runners and sealings are necessary to avoid damages caused by abrasion. Technical measures are mostly using different dredging techniques to either relocate the material into the downstream section or deposit it outside the water body. All option, especially technical measures encounter additional problems in the alpine environment compared to lowland applications. Reservoirs at high elevations are often situated in remote areas with limited access making it difficult or even impossible to transport large equipment. Described method to evacuated sediment from reservoirs affects the ecosystem in different ways. Channel adjustment to increased sediment influx depends on the magnitude, frequency, duration and grain-size distribution of the sediment released, and on the downstream channel characteristics (Wohl and Rathburn, 2003). Adverse biological effects can be related both to the direct effects of increased suspended sediments during the removal operation and to the morphological modification of river habitats following sedimentation of the removed material (Crosa et al., 2010). Sediment

management practices aimed at preservation of reservoir storage should therefore consider safeguard of riverine ecosystem, possibly against affordable financial costs (including loss of regulated water and hydropower, engineering works, monitoring programs and eventual research projects).

1.5 Objectives of the study

The primary objective of this thesis is the study of ecological criteria and biological tools to monitor and minimize the negative effects induce on stream ecosystems by hydropower schemes in an Alpine area. In particular, this work focuses on the two major topics which are related to the sustainable use of Alpine water courses, organized as follows:

1) CHAPTER 2. The downstream ecological effects of 4 controlled free-flowing flushing operations designed to remove sediments accumulated in an Alpine reservoir are investigated. Specific objectives are:
    a. to quantify the extent to which the SSC in the receiving water course can be controlled by flushing operations;
    b. to test the hypotheses of damage to invertebrates and fish which have led to the proposed regulatory framework;
    c. to propose guidelines on SSC for similar Alpine water courses subject to flushing operations;
    d. to document biological recovery capacity and identify the temporal extent of recovery;
    e. to verify long-term biological sustainability of yearly sediment removals.
2) CHAPTER 3. The consequences of discharge reduction (minimum flow releases) on macroinvertebrate communities in a data set of 30 Alpine streams are analyzed. Specific objectives are:
    a. to study the response of macroinvertebrate assemblages to a gradient of hydrological alteration corresponding to increasing water diversion;
    b. to verify at what extent invertebrate-based metrics respond to flow reduction;
    c. to evaluate the use of macroinvertebrate community for detecting the effects of flow reduction in Alpine stream ecosystems.

CHAPTER 2

Effects of yearly experimental sediment releases on instream habitat and biota

2.1 Introduction

Worldwide annual loss of reservoir storage by sedimentation is estimated around 0.5% (more than 33,000 Mm^3) while demand for regulated water and renewable electricity is still increasing (White, 2010). Reservoir functionality is severely impacted when less than 50% of storage has been sedimented and if silting is not controlled, reservoir is simply lost: nevertheless, old structures cannot be simply replaced by new ones as economically exploitable sites for new reservoirs are shrinking. Consequently as pointed out since the nineties (Morris and Fan, 1997) approach in design and operation of reservoir should deeply change: storage should be considered as a non-renewable resource and the concept of replaceable engineering infrastructure with limited economic life (generally establish by sedimentation) should be substituted by sustained long-term utilization.

Sediment flushing may offer a viable means for recovering and maintaining storage capacity of small to medium sized mountain hydropower reservoirs. Satisfactory results may be achieved with "free-flow" flushing, i.e. by letting inflow water run over the bottom of the empty reservoir and to outflow through bottom sluices (Morris and Fan, 1997; Brandt, 2000a). Although this operation does not involve excessive technical difficulties, and water loss may be acceptable for reservoirs with small storage volumes, concerns arise over the ecological consequences for the river receiving the removed sediments. If the amount of evacuated sediment is relevant (i.e. of the order of annual deposition) and the duration of the works ranges as usual between a few to a tenth of days, the ecological impact of a flushing can be severe (Gerster and Rey, 1994; Rathburn and Wohl, 2001). Early observations document significant environmental damage downstream of the dam due to sediment release (Kanthak, 1924; cited by Brown, 1943). More recent case studies have quantified specific biological and physical alterations: variation of the water chemistry (Buermann et al., 1995; Hesse and Newcomb, 1982; Garric et al., 1990), reductions of benthic organisms due to bed siltation (Chutter, 1969; Rabení et al., 2005) or following the increase in suspended solid concentration (SSC) (Gammon, 1970; Gray and Ward, 1982), severe fish mortality (Hesse and Newcomb, 1982; Buermann et al., 1995), as well as the sedimentological and geomorphological consequences of flushing (Brandt and Swenning, 1996; Wohl and Cenderelli, 2000). Environmental effects of flushing are reviewed by Brandt (2000a). These adverse ecological consequences can be connected to the direct effects (lethal, sublethal and behavioral effects) of the suspended solids on living organisms and to the long lasting morphological modification of river habitats following sedimentation of the flushed material.

In spite of its key role in the field of reservoirs management, this subject is poorly documented and technical guidelines for sediment release and for related monitoring activities represent still controversial issues. In particular, prescriptions on the very basic quantities necessary to plan and operate a flushing (threshold quantities, peak and average allowable SSCs, peak and average allowable flow rates, duration of the event, frequency of flood peaks) balancing technical, ecological and economical aspects, are not so far provided. The ecological response to such sediment removal activities is in fact strongly site-specific and the effort related to appropriate field surveying is huge.

A number of publications recommend maximum levels of turbidity or SS for the protection of fish (McLeay et al., 1984; Loyd, 1987), but their wider application in other settings is still questionable (Doisy and Rabení, 2004). Other examples point out that the maximum SS concentrations during flushing operations encompass a wide range of values, in some cases expressed in terms of vol/vol, or indirectly by nephelometric turbidity units (NTU). Authors suggested that a threshold limit of 50 mg L^{-1} may provide an adequate margin of safety for aquatic life (Pruitt et al., 2001; Petz-Glechner et al., 2003). In 1996, a panel of scientists recommended to the Georgia Department of Natural Resources (DNR) proposed establishing a turbidity standard of 25 NTU (Kundell and Rasmussen, 1995). Gerster and

Rey (1994) report for the Ticino (CH) Cantonal normative limits ranging from 5 ml L^{-1} to 10 ml L^{-1}. The Austrian legislations prescribes a maximum of 4.5 g L^{-1} (Schneider et al., 2006). Examples from Italy show maximum hourly concentrations of 100 g L^{-1} (Vitali et al., 1995), an average concentration of 6 g L^{-1} during the entire operation (Regional Government of Veneto, 2006) or maximum values of 40 g L^{-1} with a maximum duration of 0.5 hours (Regional Government of Piemonte, 2008).

Table 1. Selected proposed Suspended Solids Concentration criteria and guidelines for protection of aquatic life.

Country	Suspended sediment concentration	References
Europe	Maximum of 25 mg/l measured as annual mean (for high level of protection).	Alabaster and Lloyd, 1982; EIFAC,1964.
British Columbia	Maximum increase of 25 mg/l from background level in 24 hours, and a maximum mean increase of 5 mg/l from background in 30 days (when background is less than or equal to 25 mg/l).	British Columbia Ministry of Environment, Lands and Parks, 1998.
Austria	Maximum of 4,500 mg/l for flushing operations.	Schneider *et al.*, 2006.
Switzerland	Limits for flushing ranging from 5ml/l to 10 ml/l.	Gerster and Ray, 1994.
Italy	Maximum of 100,000 mg/l for 1 hour and a maximum daily mean of 6,500 mg/l (for flushing operations).	Vitali *et al.*, 1995.
Northeast Italy	Maximum of 30,000 mg/l for 2 hours, and a mean of 6,000 mg/l for the entire duration of flushing.	Regional Government of Veneto, 2006.
Northwest Italy	Maximum of 40,000 mg/l for 0.5 hours and less than 5,000 mg/l for the entire duration of flushing (equal or less than 1 week).	Regional Government of Piemonte, 2008.

Examining the literature, it became clear that problems arise with respect to the temporal definition of sediment thresholds and for the identification of their “numerical” descriptors. This aspect is shown in Table 1 which summarize some of the different criteria present in the international scenario. For example, the European Freshwater Fish Directive (FFD) currently stipulates a guideline annual mean suspended sediment concentration of 25 mg L^{-1} based on ecological impacts on salmon as a key indicator species. However, any attempt to characterize a sediment target in terms of an annual mean clearly fails to represent the highly episodic nature of sediment transport. Sediment targets for flushing clearly require a different temporal basis than those for general/natural sediment transport. These studies outline that there is still uncertainty concerning key aspects of this complex subject, which has hindered the delineation of technical guidelines for sediment release management and for monitoring activities. As a result, reservoir operators usually undertake the flushing operation by experience rather than according to a definite, evidence-based operating procedure (Chang et al., 2003). This chapter presents the results of a study of consecutive sediment free-flow flushing events from an alpine reservoir, in order to provide more precise quantitative clarification of the effects of reservoir flushing. Specific objectives, leading to the final management objectives, are :

a) to quantify the extent to which the SSC in the receiving water course could be controlled by flushing operations;
b) to test the hypotheses of damage to invertebrates and fish outlined above, which have led to the proposed regulatory framework;
c) to propose guidelines on SSC for similar Alpine water courses subject to flushing operations;
d) to document biological recovery capacity and identify the temporal extent of recovery;
e) to verify long-term biological sustainability of yearly sediment removals.

Understanding temporal recovery patterns will aid managers in making better decision about the timing of, and recovery from, future sediment releases.

2.2 Study site

Valgrosina reservoir is located in Northern Italy, in the alpine valley of Valtellina (Rhaetian Alps) (Figure 1). The reservoir, placed approximately at an altitude of 1,200 m a.s.l., is fed directly by the Roasco d'Eita stream (eastern branch of the Roasco stream) and, through a short canal, by the Roasco di Sacco stream (western branch of the Roasco stream); both streams may by-pass, partly or completely, the reservoir. Hydraulic works to regulate reservoir inflow are equipped with gauging weirs.

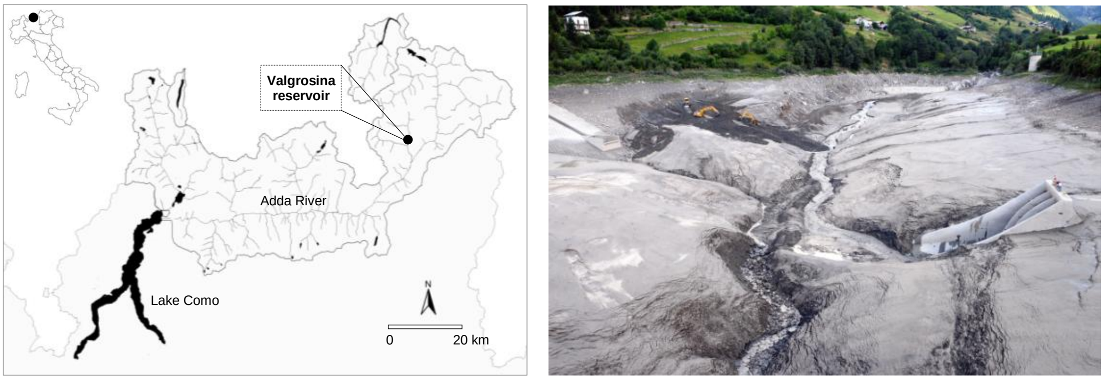


Figure 1. Location of the research site in the Northern Italy and reservoir bottom during 2008 flushing operation (third day).

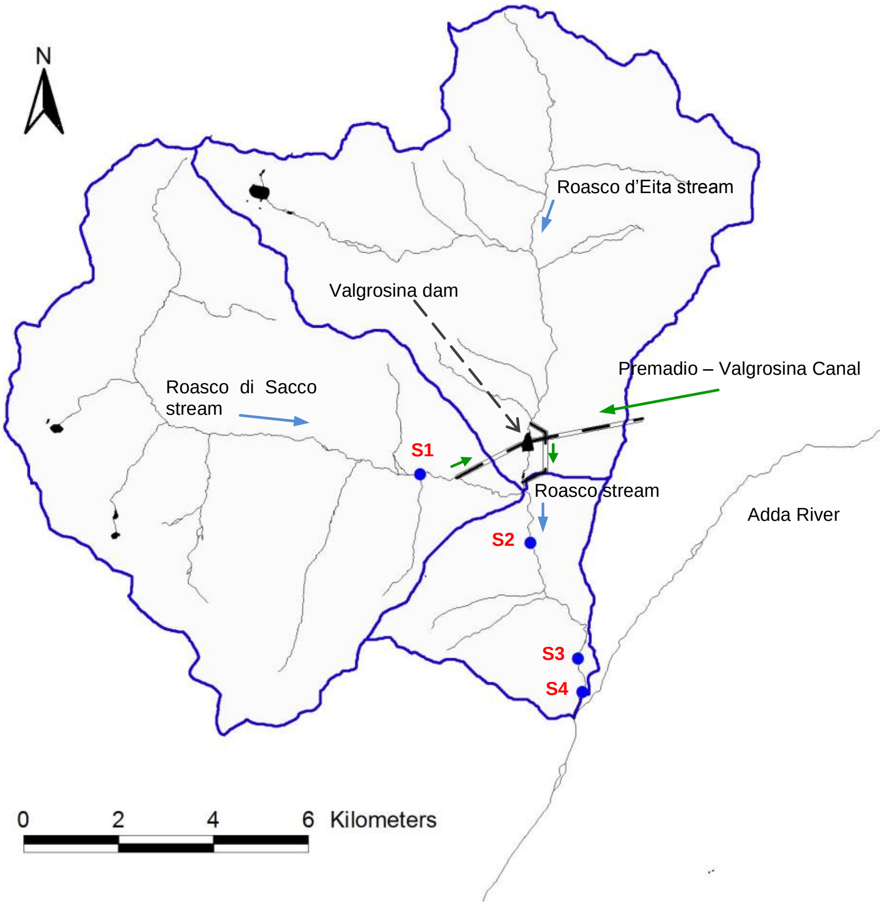


Figure 2. Roasco stream catchment and measurement stations. Hydraulic diversions and bypass systems are also drawn.

The principal water supply reaches the reservoir through the Premadio-Valgrosina tunnel-canal connecting the powerhouse of Premadio to Valgrosina reservoir (Figure 2). The effluent, the Roasco stream, flows for about 7.8 km, with average slope of 8%, mainly in a deeply incised canyon in predominantly gneissic rocks, before flowing into the Adda river at 585 m a.s.l.. The Roasco stream is a high-gradient, narrow, confined step-pool stream where the substrate is dominate by large and stable boulders. It shows a typical nival stream flow regime with natural mean annual discharge (downstream the confluence of western and eastern branches) of about 4.5 $m^3 s^{-1}$ and a monthly maximum in June (2.5÷3 fold the annual mean). From 2009, a minimum flow target of 0.2 $m^3 s^{-1}$ between November and April and of 0.4 $m^3 s^{-1}$ between May and October is maintained with the release of water from the diversion work on the Roasco di Sacco stream. Previously, seasonal modulation of the released water wasn't scheduled and a target flow of 0.2 $m^3 s^{-1}$ was maintained all over the year.

Drainage area at the dam section (including the basin of the western tributary) is approximately 120 km^2; the catchment mainly develops in a mountainous area lacking significant human activities, characterized by rocky and stony surfaces, poorly vegetated (<45% of the total area) or with mixed vegetation, primarily alpine vegetation (28%) and coniferous woodland (15%). The hydrographic basin linked by Premadio-Valgrosina canal is considerably larger than the natural one (about 650 km^2).

Valgrosina reservoir has a relatively small storage capacity (1.3×10^6 m^3) but plays a key role in the local hydroelectric system by managing the daily regulation of water volumes coming from the Premadio-Valgrosina canal. Its small storage volume, coupled with relatively large inflows, results in a short water renewal time (one day on average). Valgrosina reservoir feeds Grosio power-plant (428 MW installed capacity, 598 m average effective head, 80 $m^3 s^{-1}$ maximum turbine flow rate).

Inflows may by-pass Valgrosina reservoir during the flushing operations and the Premadio-Valgrosina canal can provide additional water when the inflow from the natural tributaries is not sufficient for sediment removal. The reservoir is, therefore, particularly suitable for free-flow flushing and control of SSC in the outflow waters.

High flow velocity and turbulence combined with low particle size (<0.1 mm) of the removed material facilitate the sediment transport by the flushing discharge. This pattern is quite common in the Alps where erosion rates are generally moderate (Cerdan et al.,

2010) and glacial silt may predominate in reservoirs deposits. A fluvial barrage, located in the Adda River 3 km below the confluence with the Roasco stream, reduce the downstream transport of the flushed sediments.

2.3 Methods

Four sediment removals by free-flow flushing were carried out for analogous duration (12-13 consecutive days) and in the same period of the year (Aug. – Sept.) from 2006 to 2009: from Aug. 28 to Sep. 09 in 2006 (Castelli, 2007), from Aug. 20 to Aug. 31 in 2007, from Aug. 25 to Sep. 06 in 2008, from Aug. 24 to Sep. 5 in 2009. A Suspended Solid Concentration (SSC) threshold of 5 g L^{-1} (overall average) was planned according to the Newcombe and Jensen (1996) concentration-duration response model in order to limit environmental damage in the effluent stream (Crosa et al., 2010). Except during the first days, alternation between daytime sediment release and nighttime discharge of clear water was performed by keeping the dam lower outlets close. During these periods the inflow waters bypassed the reservoir and provided clear water for the night flush. Safety of dislodging works on the bottom of the dried reservoir ruled out the possibility of continuing flushing all day long, but also an attempt to temporarily lighten the stress on the downstream biological communities was tried. Moreover, halving the flushing duration with double suspended solids concentration (as to maintain target evacuated volume) does not change significantly SEV, as promptly verified by Newcombe and Jensen (1996) equation. At the end of each flushing event discharges of 3-4 $m^3 s^{-1}$ of clear water for 24-48 hours, followed by a peak discharge of ~10 $m^3 s^{-1}$ were provided in order to flush the accumulated sediment from the riverbed.

Sampling activities were carried out at the following sites (Figure 2): Roasco di Sacco – S1 (1,300 m a.s.l), Sopiane – S2 (990 m a.s.l., 2.4 km downstream the dam), Selve del Dom – S3 (630 m a.s.l., 6.0 km downstream the dam), Grosotto gauging weir – S4 (about 700 m downstream of S3).

2.3.1 Suspended solids and flow rate measures

A Controller Lange SC100 with a probe for continuous oxygen (Lange LDO TM) and suspended solids (Solitax ts-line sc) measurement was installed at sites S2 and S3. Measurements at S2 were carried out only during 2008 and 2009 flushing. Since flushing operations were managed on the base of real-time (i.e. not calibrated) data of the turbidimeter, volumetric measures of setteable solids were performed by using Imhoff cones. One-litre samples were collected by hand and poured into an Imhoff cone for setteable solids analysis every 15 minutes. The volume (ml L^{-1}) of deposited sediment on the cone bottom was recorded after 10 minutes. In order to post-calibrate turbidity and setteable solids data, at least 30 random one-litre samples (for each flushing and monitored station) were analyzed in the laboratory for Suspended Solid Concentration (SSC) using Standard Method 2540 D-F (APHA et al., 2005) as follow. The whole sample was poured in an Imhoff cone and allowed to settle for 30 minutes. The fraction still containing suspended solids (non-settleable solids) was siphoned and filtered through a 0.45 μm filter with a vacuum filtering apparatus; the residue retained on the filter was dried to a constant weight at 105°C, cooled in a desiccator and weighted. The increase in weight of the filter represents the non-settleable solids. The settleable solids were removed from the bottom of the cone, put into a pre-weighted crugible, dried at 105°C, cooled and weighted. The sum of settleable solids and nonsettleable solids represents the total suspended solids in the water sample.

During 2008 and 2009 monitoring campaign, suspended material was collected on alternate days in a bucket and kept quiet till it was completely settled, then it was rescued from the bottom of the bucket and brought to the laboratory for particle size analysis by hydrometer analysis (ASTM D422-63). Thanks to this test, the size of the particles passing 0.063 mm (silt and clay) is indirectly computed by measuring the time of sedimentation of grains remaining in suspension in distilled water. For computation of diameter of grains Stoke's law, which relates the terminal velocity of a free-falling sphere in a liquid to its diameter, is used. During the test , a series of density measurements at known depth of suspension (using a hydrometer) and at known times of settlement gives the percentages of particles finer than the diameters given by Stokes' law.

Flow-rate was recorded at section S4, where a gauging weir was equipped with an ultrasound continuous level measurement probe (between S2 and S3 there are no significant tributaries). Daily suspended solid load (kg) was calculated multiplying daily average CSS and discharge for the hours of measurement.

2.3.2 Effects on streambed morphology

Surveys of channel geometry and measurements of streambed composition were carried out in the final reach of Roasco stream to quantify the effect of sediment deposition. Four transects, two in riffle units (r1 and r2) and two in a pool (p2 and p3, respectively at the upstream and downstream end of a pool), were established to assess differences between pre and post flushing conditions of river bed. Figure 3 shows the position of the cross sections respect to the other measurement sites. Cross section r2 is located in the same point as site S3, while cross section r1 and the pool are located respectively 200 m and 400 m upstream site S3. The average gradient in this segment of the stream is 3.91%..

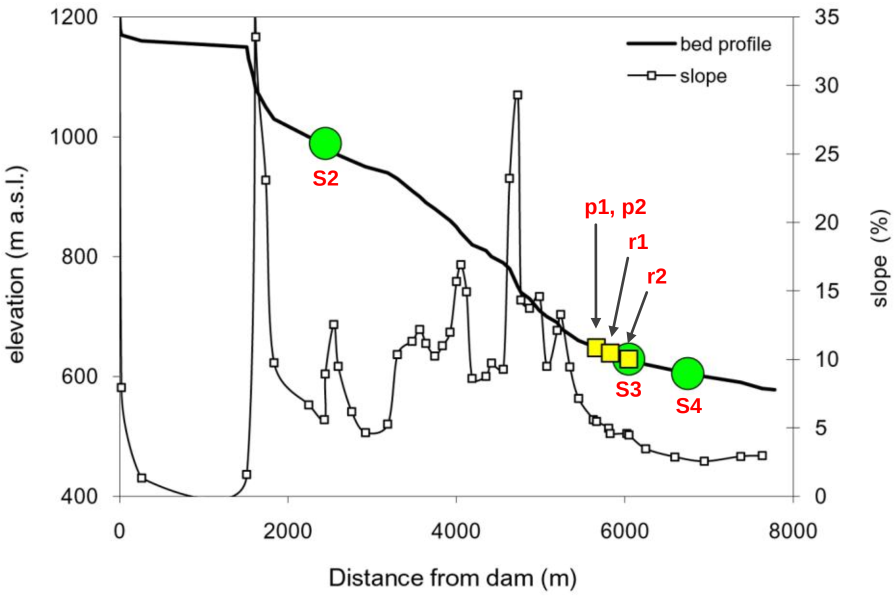


Figure 3. Slope, bed profile and position of Roasco Stream sampling sites.

Each cross-section was surveyed before (7 August 2008) and after (22 September 2008) the 2008 sediment removal. Cross section profiles of the river bed were realized taking evenly-spaced measurements (step of 25-50 cm) of water depth across each transect. In addition core sediment samples from the bed surface were collected to determine fine sediment concentration by particle-size analysis using a McNeil corer (McNeil and Ahnell, 1964) with a core tube diameter of 0.14 m. Three bulk samples equally spaced across the channel's cross section were collected at section r1 and r2. In order to analyze temporal variations and increase confidence in collected data, McNeil samples were collected approximately in the same points before and after the flushing. The depth of the water in the pool prevented the use of the McNeil sampler, hence only cross section showing pool geometry before and after 2008 flushing are available hereCollected material was dry-sieved through a standard sieve series and the portions retained on each sieve were weighted. For a comparison of the percent fines over space and time and to reduce the problem of non discriminate volume sample (Church et al., 1987; Rood and Church, 1994) core samples were truncated at 25 mm. This is to ensure that the percent fines is not affected by the presence of a few large particles. During the removal of particles, fine sediment is suspended within the core tube and since this suspended sediment can be a large fraction of the total fine particle mass (Rex and Carmicheal, 2002), one litre inner core water sample was collected and analyzed for SSC. After removing by hand bed material, the water's height in the core tube was measured in differ points to calculated the total sample volume, then the water inside the inner core was stirred by hand to resuspend settled fines and immediately one litre subsample was collected. The product of SSC and the total volume of turbid water inside the McNeil corer produced an approximation of the dry weight for the suspended material. The estimate dry weight of the suspended material was added to the weight observed in the core samples, yielding the percent of material < 0.1 mm. Percent materials less than 0.1 mm, 0.3 mm, 0.85 mm, 1.2 mm, 2.4 mm and 9.5 mm were calculated directly from the sieving results plus filtration. US Geological Survey's SEDSIZE program was utilized to calculate percentiles D_{25}, D_{50}, D_{75}.

The final reach of Roasco stream was also surveyed by visual inspection to check the patterns of sediment deposition after each release. In particular after 2009 flushing a more extensive visual assessment of the river condition was conducted by walking along the stream between sites S3 (from here to about 100 m downstream) and S4 (from here to 0,5 km upstream) and measuring the thickness of fine-sediment deposits.

2.3.3 Effects on fish populations

The biological effects of flushing operations were evaluated for fish and zoobenthic fauna. Fish populations were quantitatively sampled by electrofishing surveys (removal method with two passes) both at section S3 and section S4 using a backpack electrofishing device (mod. ELT60-IIGI 1.3 kW DC, 400/600V). The latter site is located downstream of the mentioned weir for flow-rate measurement and fishing activities are forbidden. Sampling was performed immediately before and after the flushing events (8 August and 27 September 2006; 7 August and 19 September 2007; 19 August and 23 September 2008; 18 August and 16 September 2009). Fish were identified to species level, counted, and measured for total length (TL) and body weight (W). Brown trout (*Salmo (trutta) trutta*) was the dominant species and was thus selected as an indicator species for this study. The quantitative surveys were used to estimate density (in terms of individuals and biomass) using removal method with two passes (De Lury, 1947). Population densities and biomasses were calculated taking into consideration the areas sampled, equal to 0.038 ha and 0.071 ha at site S3 and S4 respectively. In order to compare the body condition of fish before and after the flushing, Fulton's body condition factor K (Fulton, 1904) was estimated as:

$$K = \frac{100{,}000 \times W}{LT^3}$$

where W is the weight of the fish in grams (g) and LT is the length of the fish in millimeters (mm). During the 2008 pre-flushing campaign, specimens of brown trout caught at site S4 were measured and ink-marked with panjet marks (except of young of the year fish) to assess the influence of fish movements on the registered effects on population.

Cumulative length frequencies distributions were compared between pre and post flushing samples with a Kolmogorov-Smirnov (K-S) two sample test. Chi-square test was used to compare frequency of individuals in three different size classes (LT≤120 mm, 200<LT≤ 200 mm and >200 mm TL) among years. The effect on the reproductive success, possibly connected to sedimentation in spawning areas, was performed by comparing the recruitment of successive years.

For predicting the effects of flushing on the fish population, the Newcombe and Jensen (1996) concentration-duration response model was used. It allows to compute Severity Effect Value (SEV) according to the following equation:

$$\text{SEV} = A + B \times \ln(ED) + C \times \ln(SSC)$$

where *A*, *B, C* are the regression coefficients, *ED* is the duration of the exposure (in hours), *SSC* is the concentration of the suspended solids (in mg L^{-1}), SEV (Severity Effect Value) is an index ranging between 0 and 14, and represents the effects on the fish community. Lethal effects are expected if SEV is greater or equal to 10; in particular SEV values equalling 10, 11, 12, 13, 14 correspond to mortality ranges of: 0 - 20%, 20 - 40%, 40 - 60%, 60 - 80%, 80 - 100%, respectively.

2.3.4 Effects on macroinvertebrate community

Benthic invertebrates were sampled at site S1 (from May 2009) and S3. S1 was the chosen reference site for macrobenthos community assessment and was located in the occidental tributary of the Roasco stream. For the first event, a pre-flushing survey was carried out in November 2005 and March, August 2006, while the post-flushing survey took place during October 2006. In order to verify the recovery capacity of the community, a subsequent sampling campaign was executed in March 2007. For the second event, pre-flushing analysis was carried out in August 2007, whilst post-flushing campaigns were executed in September and December 2007. In 2008, pre flushing surveys were carried out in June and August, post flushing in September and December.

Benthic invertebrates were collected semi-quantitatively using a standard kick net sampler along cross-sections representative of different microhabitat. Qualitative analysis allowed estimation of the Extended Biotic Index - EBI (Woodiwiss, 1978; Ghetti, 1997). The index scores range from 0 to 14, and are ranked in 5 quality classes, from I (excellent) to V (poor). For evaluation of invertebrate densities, quantitative surveys were carried out in riffle habitats using a Surber sampler (area 0.0841 m^2; 500 µm mesh). Three replicate samples were collected on each sampling date. Moreover, 3 Surber samples were collected monthly from September 2008 to December 2008 to study with more detail recolonization processes. From May 2009 to March 2010 a monthly sampling was carried out at Site S1(reference site) and S3. A quantitative multihabitat sampling approach was used in accomplishment of Water Framework Directive (2000/60/EC) following the procedure describe in Buffagni and Erba (2007). Ten quantitative sample units (area 0.1 m^2; 500 µm mesh) were proportionately allocated in relation to the occurrence of microhabitats in the studied reaches. Collected macro-invertebrates were preserved in formalin (4%), identified to genus (Plecoptera, Ephemeroptera) or family level and, in the case of Surber samples counted. The structure and function of the benthic community were evaluated using mean density (individuals m^2), taxa richness (taxa m^2), number of Ephemeroptera Plecoptera Tricopthera taxa (EPT m^2), Margalef's richness, Shannon–Wiener diversity (H), Simpson's Dominance (C) and Evenness indices. Individuals were assigned to functional feeding guilds and then the percentage of each guild in each sample was calculated. The effects of yearly flushing operations was evaluated comparing data collected before and after the 2006 sediment removal using one-way ANOVA (date as effect) following data transformation (log(x+1)) to increase normality. Dates influenced by recent sediment release were excluded from this analysis. The short-term impacts of flushing (comparison of samples collected immediately before and after each flushing) was evaluated using one-way ANOVA for 2006, 2007, 2008 operations and a two-way ANOVA for 2009 flushing (site and date as effects), following log(x+1) data transformation.

Samples collected with the quantitative multihabitat approach were classified into five classes (from high to bad status) using the STAR_ICM index, i.e. the new official Italian method for classification based on macroinvertebrate communities, which was developed for WFD Intercalibration purposes (Buffagni et al., 2007b; EC, 2008). The STAR_ICM index is a multi-metric index based on six different metrics (ASPT index, Log_{10}(sel_EPTD+1), 1-GOLD, Number of families of EPT, Total number of families and Shannon-Weiner diversity index). The identification level required for calculation is family. After normalization by the median value of reference sites' samples, these metrics are combined into the STAR_ICM index. To classify samples, as far as the Biological Quality Element of macroinvertebrates is concerned, the official Italian boundaries were used (EC, 2008). Metric and index calculations were run with ICMeasy software, version 1.2 (Buffagni and Belfiore, 2007).

Biological data were analyzed by multivariate analyses after average densities by taxa and station had been transformed to the square root to reduce the importance of abundande taxa. Ordination analysis was run by means of the XLSTAT version 7.5.3 Copyright Addinsoft 1995-2005 software.

2.4 Results

2.4.1 Flushing operations

During the sediment removals, dissolved oxygen (DO) was always near the saturation level. In particular, during the first SSC peak in 2006 (peak 1 in Figure 6) DO concentrations remained comparable to unimpaired values, ranging from 10.8 mg L^{-1} (before the dam outlet opening) to 10.3 mg L^{-1} (after the SSC peak), with saturation values always higher than 90%. Flushed material was predominantly silt; particle size analysis of suspended sediment showed only very slight difference between various days and between the two analyzed events (Figure 4 and 5). Sediment size at site S2 and S3 was essentially the same: mean D_{10}, D_{50} and D_{90} were respectively 3.2 μm, 16 μm and 49.9 μm at site S2 and 2.8 μm, 14.5 μm and 43.6 μm at site S3. These differences were not significant (One-way ANOVA, $p<0.05$). For both stations, considering all days together, 81% of material was included between coarse silt (0.0625mm) and very fine silt (0.004 mm), while only a little parts was very fine sand having diameter bigger than 0.0625 (5%) and clay with diameter less than 0.004 mm (13%).

SSC measured in the field by turbidimeter and Imhoff cones were corrected performing a posteriori calibration through analysis of collected samples. Comparison between continuous probe data and laboratory samples data obtained by filtration and drying showed a good concordance between the two datasets both during 2006 ($CSS_{lab}= 1.06 \times CSS_{probe}$, $r^2=0.84$, n=150) and 2007 ($CSS_{lab}= 0.97 \times CSS_{probe}$, $r^2=0.94$, n=54), therefore raw SS concentration data (i.e. with no calibration of the probe data against laboratory values) are presented (Castelli, 2007; Espa et al., 2007; Crosa et al., 2010). The disagreement between the continuous probe measurement and the laboratory values observed during the first day of 2007 flushing (Figure 6) was due to the high sand content in the analyzed samples resulting from the erosion of a large landslide that in December 2006 partly occluded the Roasco stream just below the junction of the western tributary forming a pond of several thousand cubic meter. During 2008 and 2009 probe data were corrected through analyzed samples. Detailed description of the calibration procedure can be find in Folini (2009) and Previde Prato (2009) respectively for 2008 and 2009 flushing.

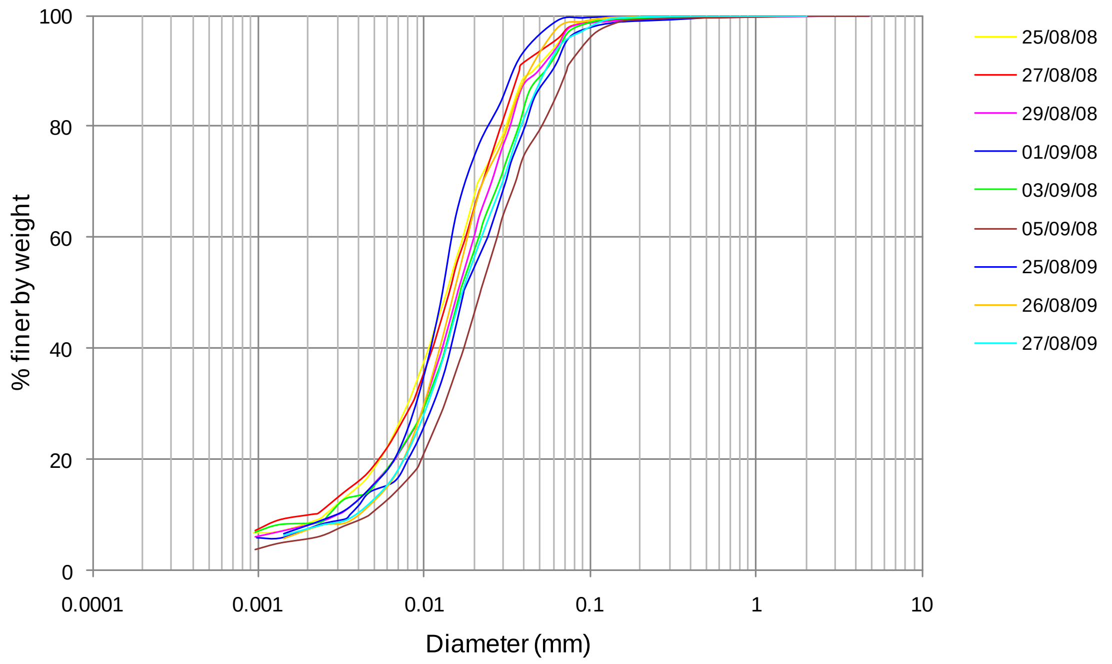


Figure 4. Particle size distribution of suspended sediment collected at site S2

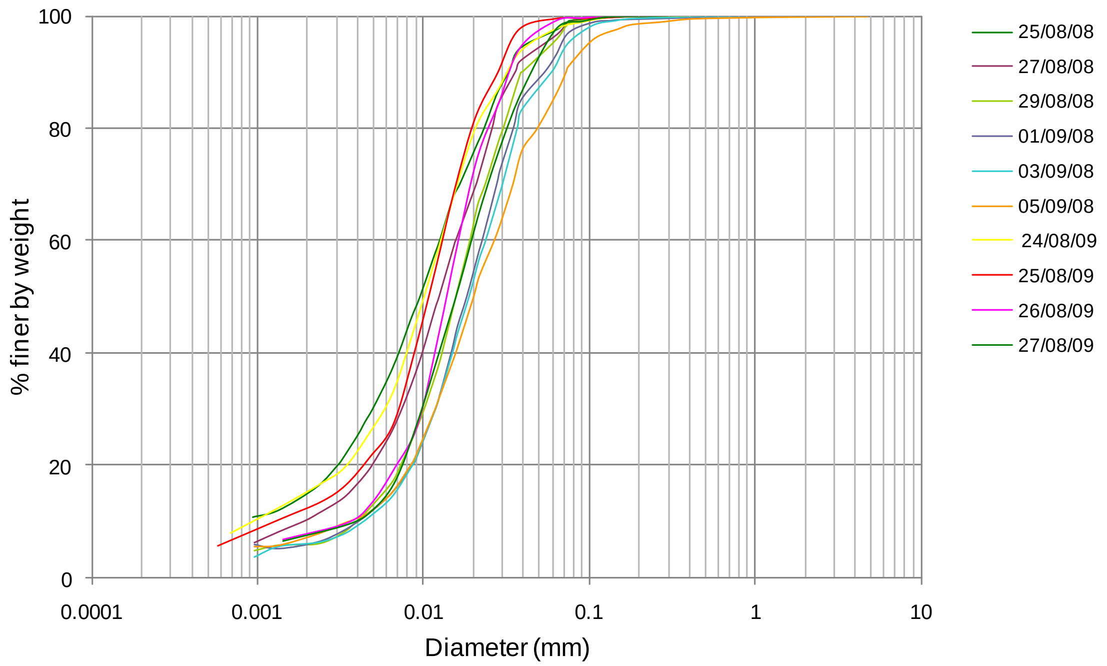


Figure 5. Particle size distribution of suspended sediment collected at site S3

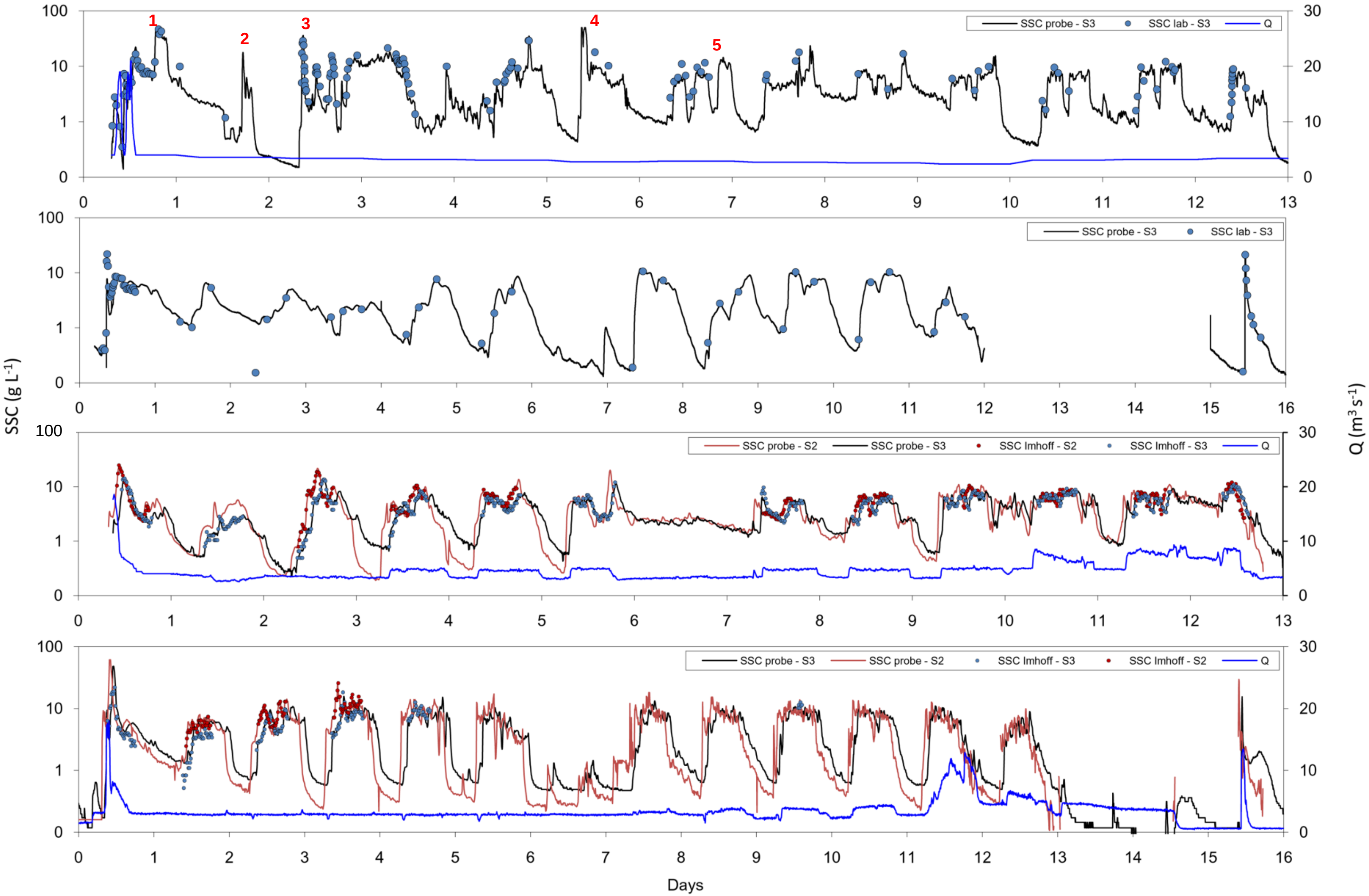


Figure 6. Time series of SSC (g L^{-1}) and Q ($m^3\ s^{-1}$) recorded in the Roasco Stream at site S2 and S3. (a) 2006. For the explanation of the numbers see the text; (b) 2007; (c) 2008 and (d) 2009.

Computed mass of suspended solid overall flowed through the measurement sections (M_{TOT}), suspended solid concentration averaged over the entire flushing duration (SSC_{AVE}) as well mean value of the flow rate (Q_{AVE}) are listed in Table 2. Quantitative descriptions of the flushing operations are shown in terms of the complete SSC time series recorded (Figure 6) and the SS duration data provided in Table 3.

The four sediment removals from the Valgrosina reservoir were quite similar. Except for the first one or two days of each flushing work, daytime sediment release was alternated to nighttime clear water release, keeping bottom outlet open but making tributary discharges bypass the reservoir. This alternation of sediment releases (day) and clear water (night) determined the pulsing pattern shown by the SSC in Figure 6. After the first daytime.

Table 2. Main hydraulic parameters of the discussed flushing events.

Site	Event	M_{TOT} (10^3 kg)	SSC_{AVE} (g L^{-1})	Q_{AVE} (m^3 s^{-1})
S2	2008	~ 21000	4.0	4.6
	2009	~ 18100	4.1	3.5
S3	2006	~ 17000	4.7	3.2
	2007	~ 14000	3.0	4.4
	2008	~ 18700	3.5	4.6
	2009	~ 17600	4.0	3.5

Table 3. Duration of SSC in the Roasco stream based on site S2 and S3 measurements. Duration is expressed as percentage of the overall operation time.

		SSC (g L^{-1})											
Site	Year	50	40	30	20	10	8	6	4	2	1	0.5	0.2
S2	2008	-	-	-	0.6	3.1	8.1	24.0	43.8	70.6	84.2	91.0	99.8
	2009	0.2	0.2	0.3	0.5	7.4	18.5	33.5	44.0	52.0	63.9	75.1	96.7
S3	2006	0.3	0.6	1.5	2.1	9.3	16.1	29.3	41.5	61.1	80.3	91.9	98.1
	2007	-	-	-	-	0.8	6.5	16.6	29.3	52	72.4	83.5	97.3
	2008	-	-	-	-	1.2	3.7	14.8	39.8	70.1	85.6	97.6	100
	2009	-	0.2	0.3	0.4	3.7	15.6	30.4	43.8	56	67.4	92.8	98.5

The time series of the SSC measured during 2006 at site S3 were characterized by several short, sharp peaks, occasionally exceeding 30 g L^{-1}, a value corresponding to 1.5% of the overall duration (Table 3). The maximum peak was measured at the end of the day 1 (peak

1 in Figure 6) and was a consequence of 0.5 $m^3 s^{-1}$ water inflow into the already emptied reservoir lasting about 4 h. Although a lower discharge would have been more appropriate to maintain the sediment load at more prudent levels (at least during the first inflow operation), the relatively large value was chosen because accumulated sediments threatened to obstruct the already opened lower outlet of the dam. After the first day, several flushing strategies were tested, including release under pressure (peaks 2 and 3 in Figure 6) and different inflows in the empty reservoir, which caused the irregular variation of the SSC measured at section S3.

On day 6 the second lower outlet was opened and the abrupt removal of sediment occluding this outlet gave rise to peak number 4 in Figure 6. The analogous operation carried out during the first day of the 2006 flushing on the first lower outlet did not have the same consequences because the clear water from the small Fusino reservoir (storing about 70×10^3 m^3) was used to increase dilution during the initial phase of outlet opening. After the first week, the system became more stable, reliable and predictable as shown, for example, by the SSCs series measured during the seventh day. In this case, with full-opened lower outlets and using the available fraction of flow for dilution (2.9 $m^3 s^{-1}$), the following operations were carried out. From 7 a.m. to 11 a.m., a constant inflow of 0.3 $m^3 s^{-1}$ was maintained through the reservoir; in the following two hours (from 11 a.m. to 1 p.m.) this inflow was interrupted; the same schedule was repeated twice but increasing the discharge of Roasco d'Eita flowing through the reservoir (0.5 and 0.7 $m^3 s^{-1}$, respectively). SSC measured at site S3 were closely associated with increases in water inflows (0.3, 0.5, 0.7 $m^3 s^{-1}$, peak 5 in Figure 6).

During the final days of the 2006 flush, a satisfactory control of the out-flowing SS loads through reservoir operations was achieved. For example, on day 12 the water flowing into the reservoir (1 $m^3 s^{-1}$) from the Premadio-Valgrosina canal and the entire discharges of the two natural tributaries (Roasco d'Eita and Roasco di Sacco) by-passing the reservoir maintained an average discharge of 3.2 $m^3 s^{-1}$ in the Roasco stream. Furthermore, an excavator was used from 8 to 12 a.m. and from 1.30 to 6.30 p.m. to dislodge sediments from the banks of the main channel formed on the reservoir bottom. The SSC was strongly correlated with excavator activity, with concentrations ranging between 8 and 10 g L^{-1} during digging and a noticeable drop during work breaks (Figure 6).

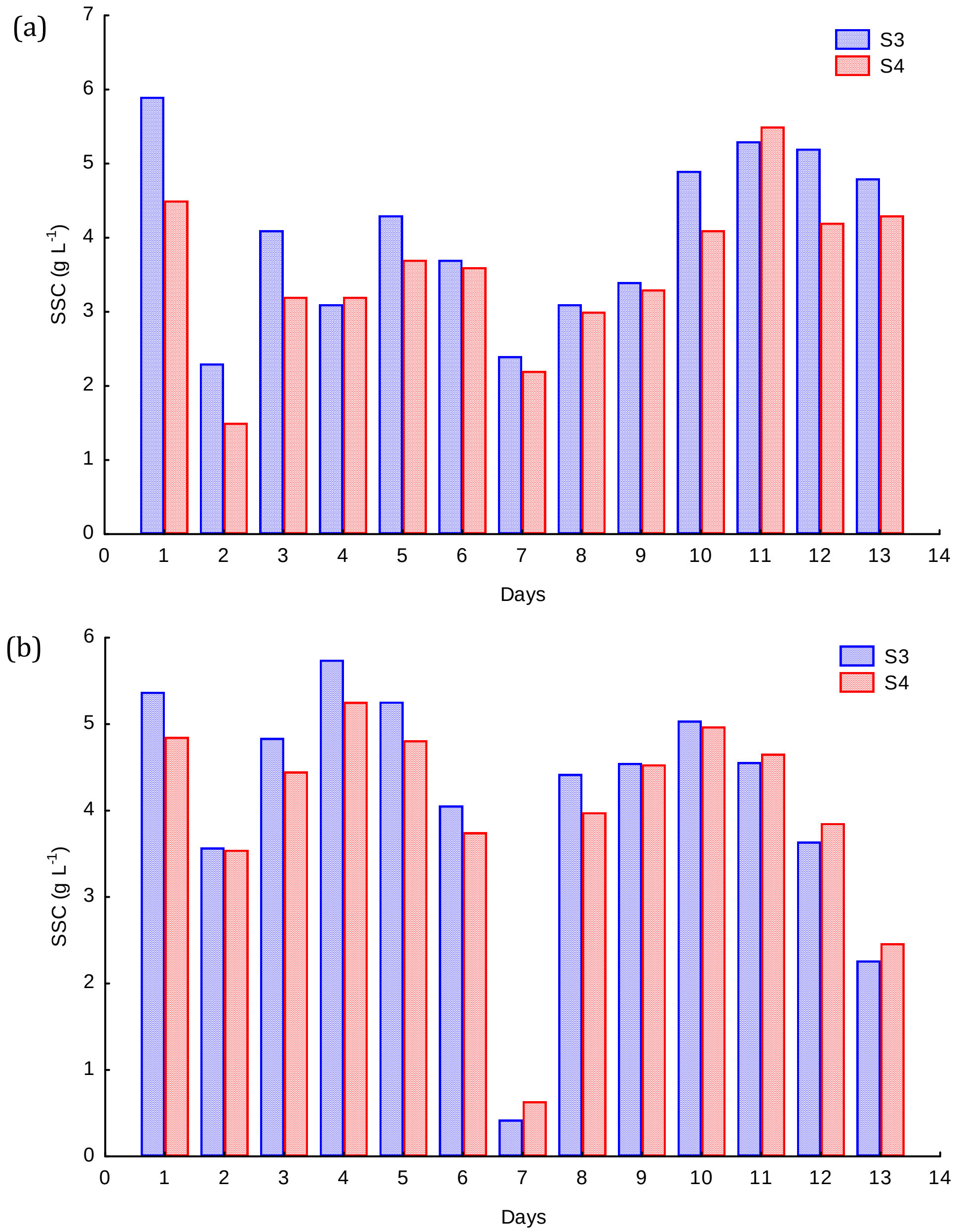


Figure 7. Daily mean SSC measured at site S2 and S3 during 2008 (a) and 2009 (b).

On the whole, if compared to 2006 flushing, these events are characterized by lower SSC, both in terms of averaged values (Table 2) and peak values (Table 3). Alternation between daytime sediment release and nighttime discharge of clear water was performed, as for the previous event, but the maximum SSC was lower and the SSC series was more regular compared to 2006. In fact, the efficiency of lower outlets was restored during the

2006 flush, and the experience acquired in 2006 allowed a better use of scouring and dilution waters. Nevertheless, SSC measured during the first day of flushing confirmed the difficulties in system control on the early flushing phase, connected with the drainage of the reservoir and with the contemporary decrease of the flow rate available for dilution.

As it can be observed in Figure 6 (c an d) both measurement sites had the same SSC pattern, with a 1-2 hours temporal shift as a function of the discharge released in the channel. Higher SSCs were found at site S2, especially during high sediment load events. In particular, SSC >10 g L^{-1} were measured for 3% and 1% of overall duration respectively at site S2 and S3 in 2008, while in 2009 approximately for 7% and 4% of overall duration (Table 3). Although peak concentration was lower at site S3, average SSCs were not markedly reduced in the final reach measurement station (Figure 7). Even considering the mass of suspended solid overall flowed in each section (Table 2), we didn't find remarkable difference between the estimated mass flowed at site S2 and S3.

Moreover particle-size of flushed material did not differ significantly from site S2 and S3, thus excluding important deposition phenomena in the Roasco reach downstream of western tributary confluence. Accordingly, it can be assumed that the greater part of the fine sediment scoured during flushing was carried along the stream (>90%) and that SS measurements carried out in the downstream site were representatives of the suspended solid dynamics in the studied stream segment.

2.4.2 Effects on streambed morphology

Sketches of the cross-sections with measurement points for water depth, velocity and particle size analysis (Mcneil samples) are presented in Appendix A. Discharge calculated from depth and velocity measurements during the pre flushing survey was 0.4-0.6 m s^{-1}. Changes in streambed elevation resulted from both scour and fill processes. In particular in proximity to the thalweg, high discharge during flushing scoured the channel, deepening the bed (i.e. pool left side) while near the streambanks sedimentation of fine material was more evident. Anyway, comparison of the cross section morphology before and after the flushing didn't provide evidence of an high deposition event (i.e. pool filling) in the wetted channel. A total of 12 bulk samples were analyzed for particle size analysis, 3 for each riffle zone cross section and condition (before – after). Sample weights ranged from 0,75 to 1,38 kg (particles less than 25 mm in diameter). The results are presented as percent finer by weight curves in Figure 8 and 9.

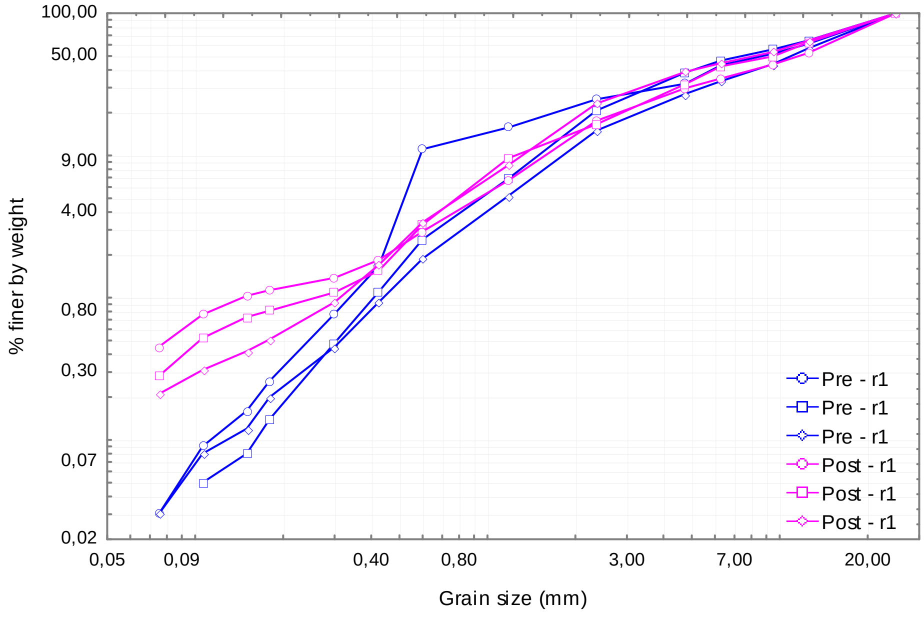


Figure 8. Particle size distribution curves of McNeil samples collected at cross section r1.

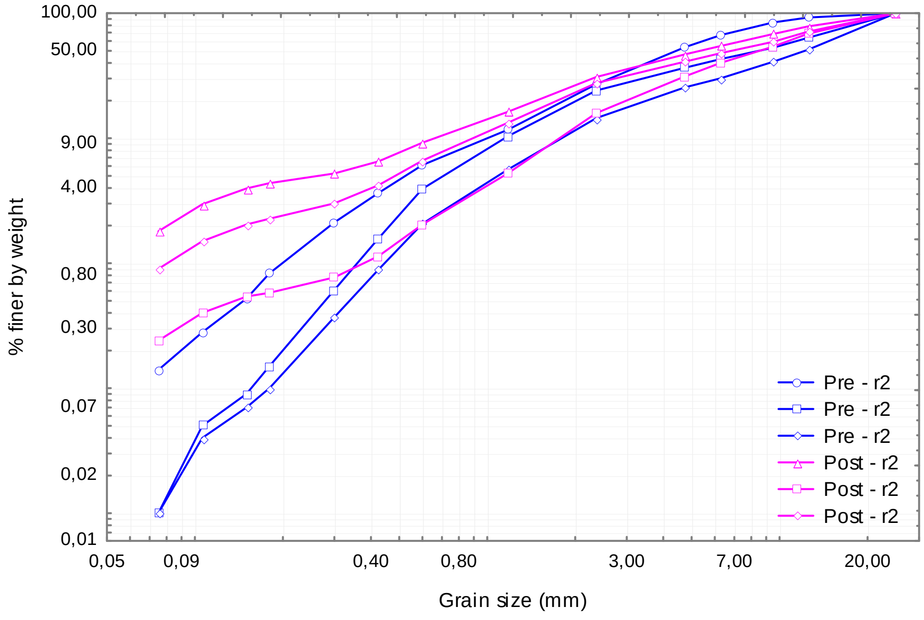


Figure 9. Particle size distribution curves of McNeil samples collected at cross section r2.

The same pattern was observed between the measurement sections with a visible increase of the percentage of fines at the end of sediment flushing and bed cleaning operations. In transect r1 the percentage of material <0,1 mm ranged from 0.05 to 0.09% (CV = 0.29) before the flushing and from 0.31 to 0.76% (CV = 0.42) at the end of operations. In transect r2 the same fraction passed from 0.04 – 0.028% (CV = 1.13) to 0.4 – 2.96% (CV=0.79). No significant difference was found between the median percentage of material <0,1 mm observed in section r1 and r2 (Mann-Whitney U Test, $p>0.05$). Considering the average of all bulk samples, the cumulative percentage of fine sediment less than 0.1 mm in diameter increased 11-fold, from 0,10% (CV=0.95) to 1,08% (CV=0.95), while the material passing 0.075 mm sieve increased 18-fold passing from 0.04% (CV=1.42) to 0.65% (CV=0.96) and these differences were both significant (Wilcoxon Matched Pairs Test, $p<0.05$).

Table 4. Summary table of particle size distribution indices.

Sample	Date	% fine sediment (mm)								
#	#	<9.5	<2.4	<1.2	<0.85*	<0.3	<0.1	D_{25}	D_{50}	D_{75}
Section r1										
1	Before	52.1	25.0	15.9	13.3	1.0	0.3	2.3	8.5	13.4
2	Before	55.8	20.8	7.0	4.6	0.6	0.2	2.8	7.4	13.1
3	Before	44.1	15.2	5.5	3.6	0.8	0.4	4.1	10.5	13.7
4	After	44.4	18.5	7.8	5.7	2.6	2.0	3.4	11.0	13.9
5	After	51.0	18.6	11.7	8.2	3.5	2.9	3.2	8.8	13.2
6	After	53.9	23.9	9.5	6.6	1.9	1.3	2.4	7.9	13.1
Section r2										
7	Before	84.3	27.3	12.0	8.8	2.5	0.6	2.1	4.2	7.5
8	Before	53.1	24.4	10.8	7.1	1.0	0.4	2.4	8.2	13.2
9	Before	41.0	14.7	5.8	3.8	0.6	0.3	4.4	11.8	14.1
10	After	69.5	33.6	19.6	15.8	9.0	6.9	1.5	4.8	10.7
11	After	54.8	17.3	6.8	5.0	2.4	2.1	3.4	8.1	12.7
12	After	60.7	30.7	17.0	13.4	7.1	5.7	1.8	6.4	12.4
Before	*Mean*	55.1	21.2	9.5	6.9	1.1	0.4	3.0	8.4	12.5
	SD	15.4	5.3	4.1	3.7	0.7	0.2	1.0	2.6	2.5
After	*Mean*	55.7	23.8	12.1	9.1	4.4	3.5	2.6	7.8	12.7
	SD	8.6	7.0	5.2	4.4	2.9	2.3	0.8	2.1	1.1

*Interpolated

The dry weight for the suspended material measured in inner core water samples increased significantly after the flushing (Mann-Whitney U Test, $p<0.05$): from an average of 2.2 g (CV = 0.4) to 16.5 g (CV=0.2) in transect r1 and from an average of 3.0 g (CV=0.7) to 38.1 g (CV=0.6) in transect r2. The estimate dry weight was added to fraction <0.075 mm weight of core samples and resulting particle size distribution indexes are summarized in Table 4. Considering the average of all samples, the increase of cumulative percentage of finer than 9.5 mm, 2.4 mm, 1.2 mm and 0.85 mm (McNeil and Ahnell, 1964) was respectively 0.6% (CV=20.1), 2,5% (CV=3.7), 2.6% (CV=2.8) and 2.2% (CV=2.8); these differences (after-before comparison) were not significant (Wilcoxon Matched Pairs Test, $p>0.05$). On the contrary, the fraction less than 0.3 mm and 0.1 mm in diameter were significantly greater (Wilcoxon Matched Pairs Test, $p<0.05$) in samples collected after the flushing respect to pre flushing conditions, representing respectively about 1.1% (CV= 0.66) and 0.4% (CV=0.43) in weight before and 4.4% (CV=0.66) and 3.5% (CV=1.00) after. Mean D_{25} and D_{50} slightly decreased in post flushing samples, but this difference was not significant (Wilcoxon Matched Pairs Test, $p>0.05$).

Visual assessment of the riverbed condition indicated that the majority of the sediment deposition occurred in the areas outside the wetted channel at minimum flow (i.e. bank areas affected by flushing discharges). Fine sediment thickness of streambank deposits ranged from 1 to 10 cm (Figure 11 and Figure 12). Minimum flow riverbed resulted generally covered by a thin (few mm) silt veneer (Figure 10). Thicknesses of a few centimeters were mainly observed in the marginal zones, in the areas characterized by lower velocity (Figure 13), or downstream of big boulders. Deposits of 20 cm and more in depth were confined to limited situations with peculiar channel morphology which allow the water to stagnate as discharge decreased (i.e. pools outside the wetted channel at the baseflow). Moreover, limited segments of the river with a higher gradient, often between big boulders, didn't show significant sedimentation (Figure 13). On the whole, study reaches didn't seem to be excessively impaired by sediment deposition and it may be assumed an average accumulation in the investigated reaches of about 0.20 $m^3\ m^{-1}$. This estimate is based on the areal extension and depth of the measurable deposits, hence it doesn't take into account the interstitial sedimentation (clogging), that however is quantitatively unimportant on the overall quantities.

Figure 10. Veneer of fine sediment covering cobbles in the wetted channel at site S2 (sx) and at cross section r1 (dx).

Figure 11. Pool (cross section p1 and p2) (dx) and detail of the right bank (sx). Fine-sediment deposit on the right bank: length 10m, wide 2 m thickness 5-10 cm.

Figure 12. Site S3 (cross section r2), left bank (sx) and detail of the deposit on the right bank (dx).

Figure 13. Lower velocity areas in the wetted channel were the most affected by deposition (sx); high gradient segment with no significant sedimentation phenomena (dx).

2.4.3 Effects on fish populations

Total body lengths (TL) and weights (W) of fish caught at site S3 and S4 along the studied period are presented in the form of the boxplot diagrams in Figure 14. The plots indicate the median, interquartile range, outliers, and extreme values within each data subset. Estimates were calculated separately for three different size classes (≤120 mm, 120<LT≤ 200 mm and >200 mm TL). Individuals less than 120 mm in total length were defined as young of the year (YOY) using length-frequency distributions, since the youngest cohort could be visually distinguished relatively clearly. Length frequency distributions of brown trout captured in the two sampling stations from 2006 to 2009 are shown in Figure 15. Estimated (± 95% CL) brown trout population size and biomass are given in Table 5.

Total body length of fish captured at site S3 (n = 929) ranged from 43 to 363 mm and averaged 152.9 mm (SE = 1.8 mm). Mean length of each size class varied between 2006 and 2009, from 70 to 87 mm for YOY, from 157 to 181 mm for size class 1 (LT ≤ 200mm) and from 226 to 245 mm for size class 2 (LT>200mm). Weights ranged from 0,73 g to 456 g and averaged 49.2 g (SE = 1.7g). Total length of brown trout capture at site S4 (n = 1,062) ranged from 43 to 382 mm and averaged 183.9 mm (SE = 2.6 mm). Median LT value observed at site S4 (185 mm) was significantly greater than at site S3 (159 mm) (Mann-Whitney U Test, $p<0.05$).

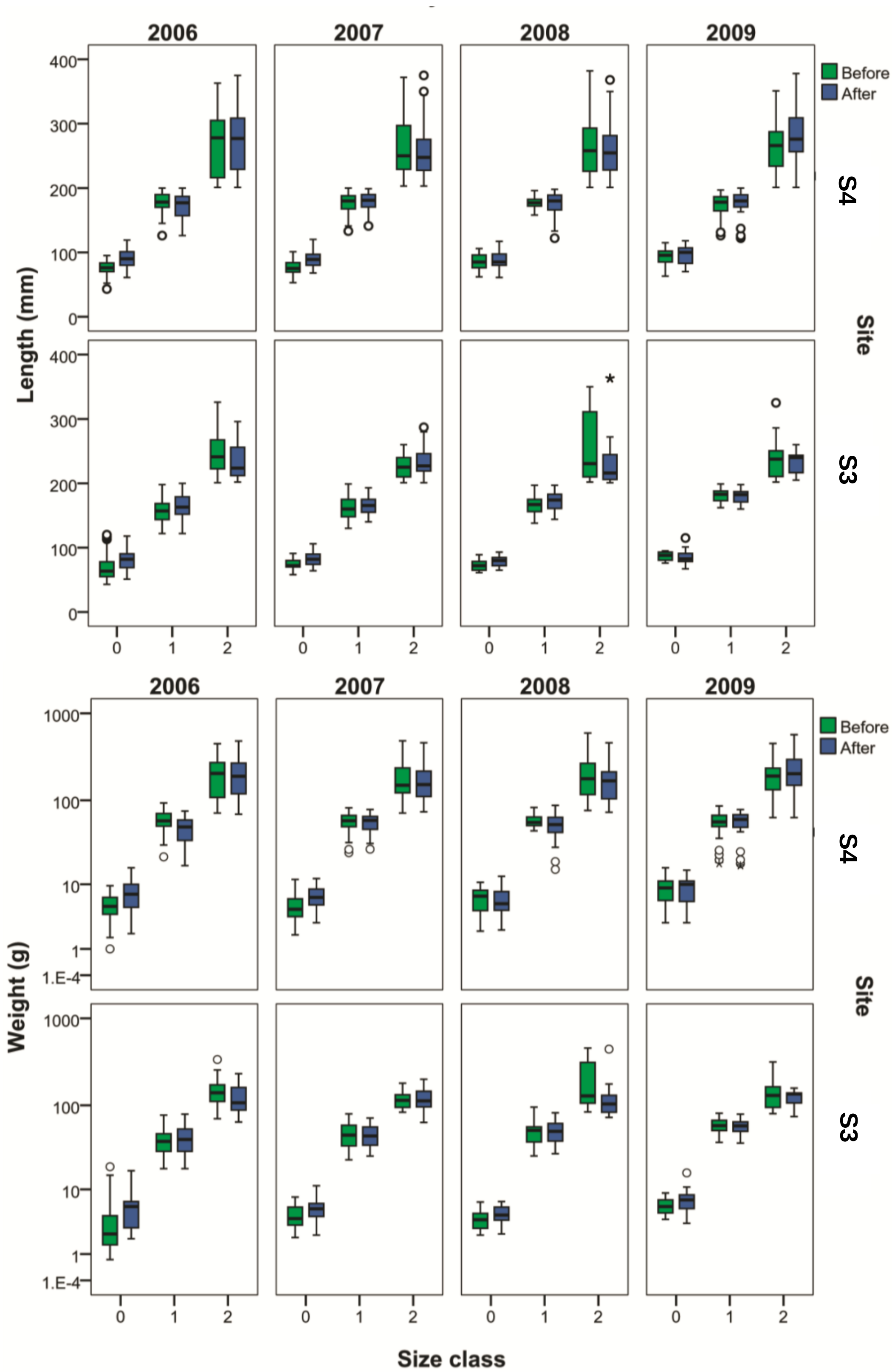


Figure 14. Boxplot diagrams depicting distribution of length (mm) and weight (g) within sites and length groups (0, LT≤120 mm; 1, 120<LT≤ 200 mm; 2, LT >200 mm).

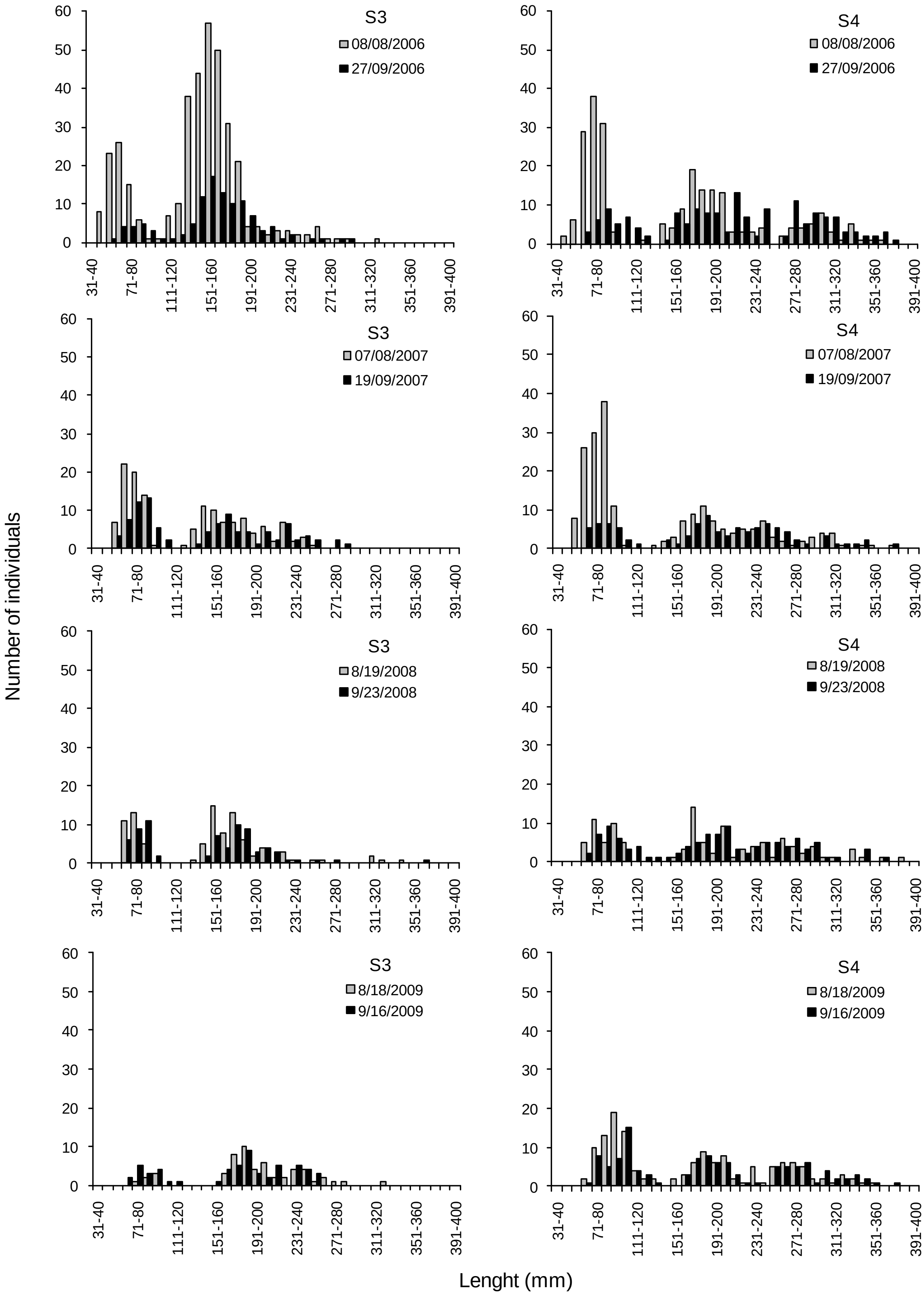


Figure 15. Population length-structure of brown trout (Salmo trutta trutta) measured at Selve del Dom (top left: 2006; top right: 2007) and Grosotto (bottom left: 2006; bottom right: 2007) during the flushing events. Grey and black bars refer to pre and post flushing respectively.

Mean length of each size class varied, between 2006 and 2009, from 75 to 96 mm for YOY, from 172 to 179 mm for size class 1 and from 259 to 276 for size class 2. Weights ranged from 1 g to 594 g and averaged 98.1 (SE = 3.3 g). At site S3 before the first flushing, the trout population was extremely abundant, although hampered by an altered age structure because of the fishing activities occurring at this locality.

Table 5. Estimated (±95% confidence limits, CL) brown trout *(Salmo (trutta) trutta)* biomass (kg ha^{-1}) and density(Ind ha^{-1}) in the two sampling sites.

Date	Site S3		Site S4	
	Ind ha^{-1}	Kg ha^{-1}	Ind ha^{-1}	Kg ha^{-1}
8-ago-06	10638.6 (± 678.0)	409.7 (± 26.1)	4026.8 (± 508.7)	291.1 (± 36.8)
27-set-06	2894.3 (± 78.2)	132.5 (± 3.6)	2373.7 (± 84.8)	294.4 (± 10.5)
7-ago-07	3833.0 (± 234.7)	147.7 (± 9.0)	2967.1 (± 98.9)	169.4 (± 5.6)
19-set-07	2425.0 (± 104.3)	110.3 (± 4.7)	1412.9 (± 127.2)	136.8 (± 12.3)
19-ago-08	2581.4 (± 208.6)	146.5 (± 11.8)	1681.4 (± 183.7)	176.4 (± 19.3)
23-set-08	2033.9 (± 156.5)	94.3 (± 7.3)	1653.1 (± 98.9)	164.6 (± 9.8)
18-ago-09	1434.1 (± 26.1)	126.9 (± 2.3)	2105.3 (± 98.9)	184.3 (± 8.7)
16-set-09	1486.3 (± 26.1)	96.8 (± 1.7)	1667.2 (± 56.5)	180.4 (± 6.1)

At site S4 a lower density was detected, but the trout population showed a good age structure as a consequence of the lack of fishing activities. For both sites natural reproduction was evident in the presence of extremely young specimens (YOY) (Figure 15). At site S3, the brown trout population after the first flush showed a severe density reduction of about 73% (from 11,000 to 3,000 individuals ha^{-1}). Reduction of total fish biomass was lower (68%), because of the selective pressure of the event on juveniles, as demonstrated by the different responses of the YOY, juveniles ($120<LT\leq 200$ mm) and adults ($LT>200$ mm), which showed density decreases of 78%, 70% and 42%, respectively (Table 6). At the second sampling site, the flushing reduced the overall populations 40% in terms of density while no remarkable alteration in terms of total biomass was measured. Again, the heavier effect was on specimens with $TL \leq 120$ mm, whose number decreased of about 70%, while no negative impact was registered on individuals greater than 200 mm in total length. The results of the fish monitoring activities for the 2007 event confirmed that the most severe effects acted on the youngest individuals (YOY decreased of about 36% and 78% respectively at site S3 and S4). The trout population had not recovered to the abundances measured in 2006 before the flushing, and a further density reductions of 37% (25% in term of biomass) and 52% (19% in term of biomass) occurred respectively at site S3 and S4 after the second flushing.

Table 6. Number, percentage composition and reduction of brown trout *(Salmo (trutta) trutta)* individuals subdivided for length class ((0, LT≤120 mm; 1, 120<LT≤ 200 mm; 2, LT >200 mm).

Site	Date	Lenght class						Reduction		
		0		1		2		0	1	2
		n°	%	n°	%	n°	%	%	%	%
S3	6-Aug	87	23.8	255	69.7	24	6.6	78	70	42
	6-Sep	19	17.3	77	70.0	14	12.7			
	7-Aug	64	46.0	53	38.1	22	15.8	36	47	
	7-Sep	41	45.1	28	30.8	22	24.2			
	8-Aug	29	31.2	50	53.8	14	15.1	3	30	14
	8-Sep	28	37.3	35	46.7	12	16.0			
	9-Aug	6	10.9	25	45.5	24	43.6		12	21
	9-Sep	16	28.1	22	38.6	19	33.3			
S4	6-Aug	109	46.2	66	28.0	61	25.8	69	38	
	6-Sep	34	20.9	41	25.2	88	54.0			
	7-Aug	114	56.2	40	19.7	49	24.1	78	40	10
	7-Sep	25	26.9	24	25.8	44	47.3			
	8-Aug	36	33.3	25	23.1	47	43.5	14		
	8-Sep	31	27.9	28	25.2	52	46.8			
	9-Aug	62	43.4	30	21.0	51	35.7	35	7	6
	9-Sep	40	34.5	28	24.1	48	41.4			

The 2008 and 2009 sediment removals determined a relative low reduction of brown trout density and biomass respect to previous ones. In 2008, at section S3 was observed a decrease of density of about 20%, while at site S4, the number of individuals estimated after the flushing was fully included in the 95% CL of those estimate before. Here, before the flushing, 108 brown trout specimens were captured, 33% YOY, 23% with 120<LT≤ 200 mm and 44% with TL ≥ 200 mm. A total of 69 individuals between 150 and 380 mm of TL were marked. After the flushing a negligible decrease in fish density and biomass was observed. Furthermore a large number of marked fish was recaptured in the same site: 52 specimens, corresponding to 75% of marked individuals (Figure 16).

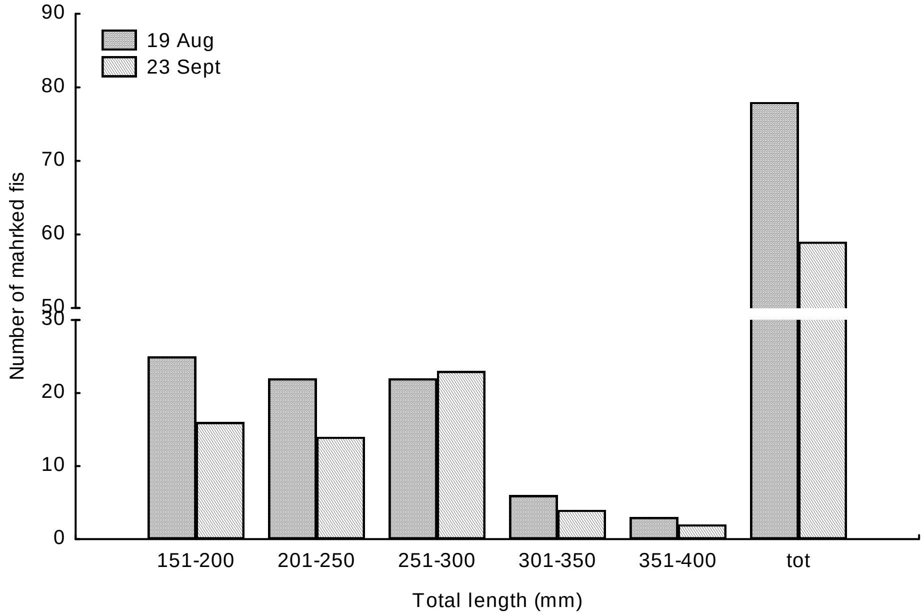


Figure 16. Number of marked and recaptured fish at site S4 in 2008.

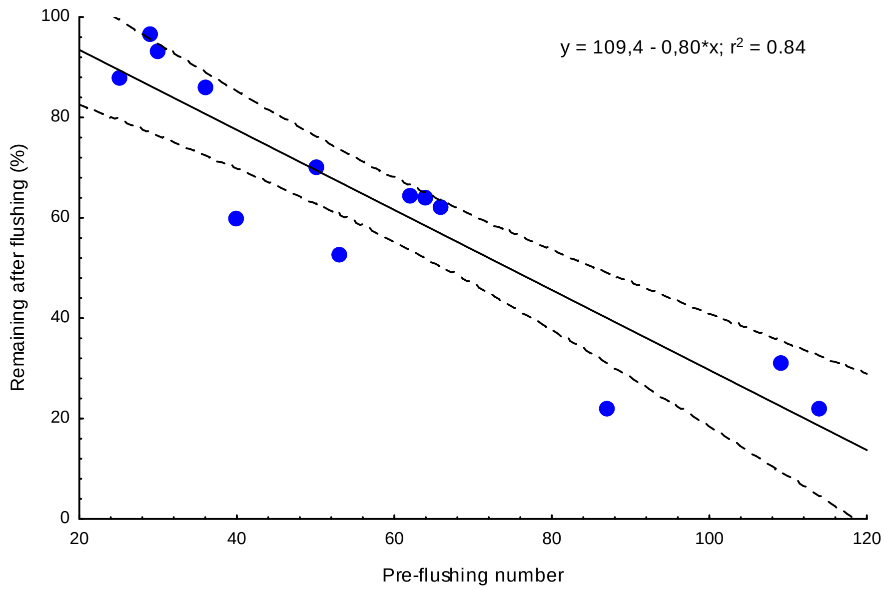
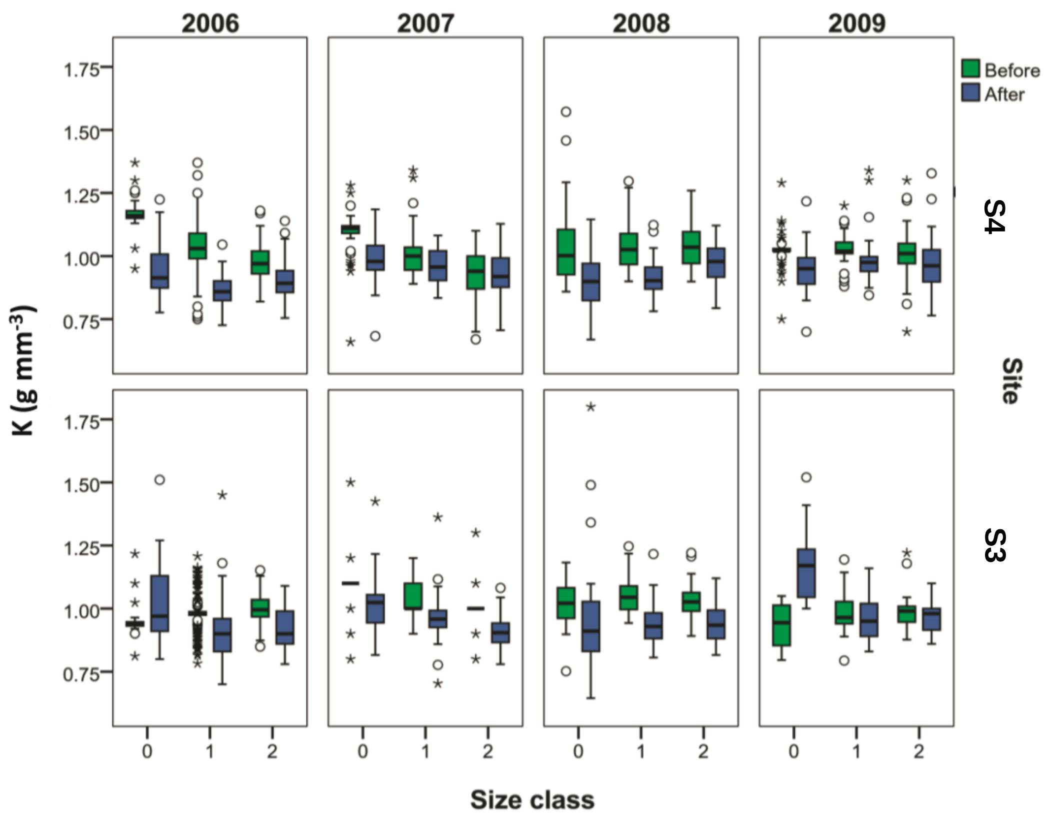


Figure 17. Relationship between the number of individuals with TL<200m caught before the flushing operations and the percentage of remaining after the flushing (Class 1 at S3 in 2006, YOY at S3 in 2009 and class 1 at S4 in 2008 have been excluded). Data refer to both site (S3 and S4).

In 2009, a density reduction of 21% was measured at site S4 while no effects occurred at site S3. However, unlike S4, the trout population had not recovered the same abundance as the previous year and a small number of YOY was caught, attesting the low reproductive success of 2008-2009 season in this station.
A significant correlation (p<0.05, test of Pearson's correlation) was found between the number of individuals caught before the flushing and the percentage remaining after the operations, for both YOY and length class 1 (120<LT≤200 mm). The relationship found (Figure 17) suggests a density-dependent survival mechanism for juveniles.
Cumulative length frequency distributions (based on 30 mm size increments) were compared with Kolmogotov-smirnov (K-S) two sample test to evaluate the effects of flushing operations on fish population structure. At site S3 no difference between pre and post samples was found in 2007, 2008, 2009, while a significant difference was observed in 2006 comparison (K-S test, p<0.05). Moreover median LT (153 mm in August 2006 and 160.5 mm in September 2006) differed significantly (Mann-Whitney U Test, n1=355 n2=110, p<0.05). At site S4 cumulative length distributions were significantly different in 2006 (K-S test, p<0.01) and 2007 (K-S test, p <0.001) comparison. Median LT (180 mm and 213 mm respectively in August and September 2006; 148 mm and 203 mm respectively in August and September 2007) differed significantly in both years (Mann-Whitney U Test, p <0.05). This confirms that the heavier impact on fish population was registered in 2006 and 2007 and the effects was greater upon the younger individuals (median LT increase in September, since fewer juveniles were caught after flushing).
Fgure 18 presents the boxplot diagrams for the coefficient of condition K calculated within size classes and sites. The plots indicate the median, interquartile range, outliers, and extreme values occurring within the data set. Significant differences between the median values measured before and after every flushing for each subset of data (site and length group) were tested using the Mann-Whitney U Test. Median K condition factor was significant lower (p<0.05) after the flushing (with the exception of 2009 at site S3, of YOY at site S3 in 2006, and class 2 at site S4 in 2007) which demonstrate a substantial decrease of fish condition after the flushing.
Recruitment of brown trout was assessed in August 2006, 2007, 2008 and 2009 as the number of YOY and the cohort's percentage contribution to the population. Numbers of YOY were markedly lower in 2008 and 2009 than in previous years.



Fgure 18. Boxplot diagrams depicting distribution of k coefficient of condition within sites and length groups (0, LT≤120 mm; 1, LT ≤ 200 mm; 2, LT >200 mm).

At site S3 more than 60 YOY fish were caught in 2006 and 2007, while less than 30 individuals in the following two years (Table 6). In the second sampling site YOY catches declined from more than 100 individuals to less than 60 in 2008 and 2009. On the contrary, number of larger individuals (LT>200 mm) remained relatively stable at both sites. Considering the percentage contributions, the proportion of YOY at site S3 steadily declined from 2007 onwards and in 2009 was significantly lower than expected (χ^2 =34.44, p<0.01; August 2006 was excluded from the analysis). At site S4 no clear pattern was observed, since percentage contribution in 2009 was roughly the same as in 2006. The proportion of YOY was significantly lower than expected only in 2008 but not in 2009 (χ^2 =23.79, p<0.01). These comparisons offer evidence for a negative influence of the flushing upon recruitment of brown trout, especially at site S3.

For evaluating the predicting capacity of the SEV model we used the duration and the SSC of the four flushing. As for the equation coefficients, we applied those defined by Newcombe and Jensen for salmonids (all ages): a = 1.0642, b = 0.6068, c = 0.7384. Fish data of Table 5 allow SEV estimation for the four investigated flushing events. The comparison is given in Table 7.

Table 7. Computed and observed SEV for the four analyzed flushing.

Event	ED (h)	SSC (mg L^{-1})	SEV computed	SEV observed	
				S3	S4
2006	300	4700	11	13	11
2007	280	3000	10	11	12
2008	300	3500	11	11	10
2009	312	4000	11	10	11

The calculated SEVs do not show substantial differences: predicted mortalities range between 0-20% (SEV = 10) in 2007 and 20–40% (SEV = 11) for the other flushing. Considering the recorded fish densities reductions, major differences between observed and predicted values were measured during 2006 and 2007 flushing, while in the following years the model appears to estimate correctly fish mortality.

2.4.4 Effects on macroinvertebrate community

Structure and function

Total biomass (g m^{-2}), mean density (individuals m^{-2}), taxa richness (T), number of EPT (EPT), Margalef's diversity (D_{Mg}), Shannon–Wiener diversity (H), Simpson's Dominance (C) and evenness (J) indices measuread are summarized in Appendix D. Density estimates of the recorded taxa are presented in Appendix B.

From November 2005 to August 2006 the macroinvertebrate community was on average 20% Plecoptera, 45% Ephemeroptera, 26% Diptera and 8% remaining taxa including Trichoptera (7%), Tricladida and Oligochaeta. Dominant taxa represented on average 29% of the community. After the beginning of sediment removals (excluding the dates influenced by recent sediment flushing, September, October and November), Plecoptera made up 9% of the community while Ephemeroptera (49%) continued to be abundant. Diptera increased to 30% and Trichoptera to 12%. Dominant taxa represented on average 63% of the community.

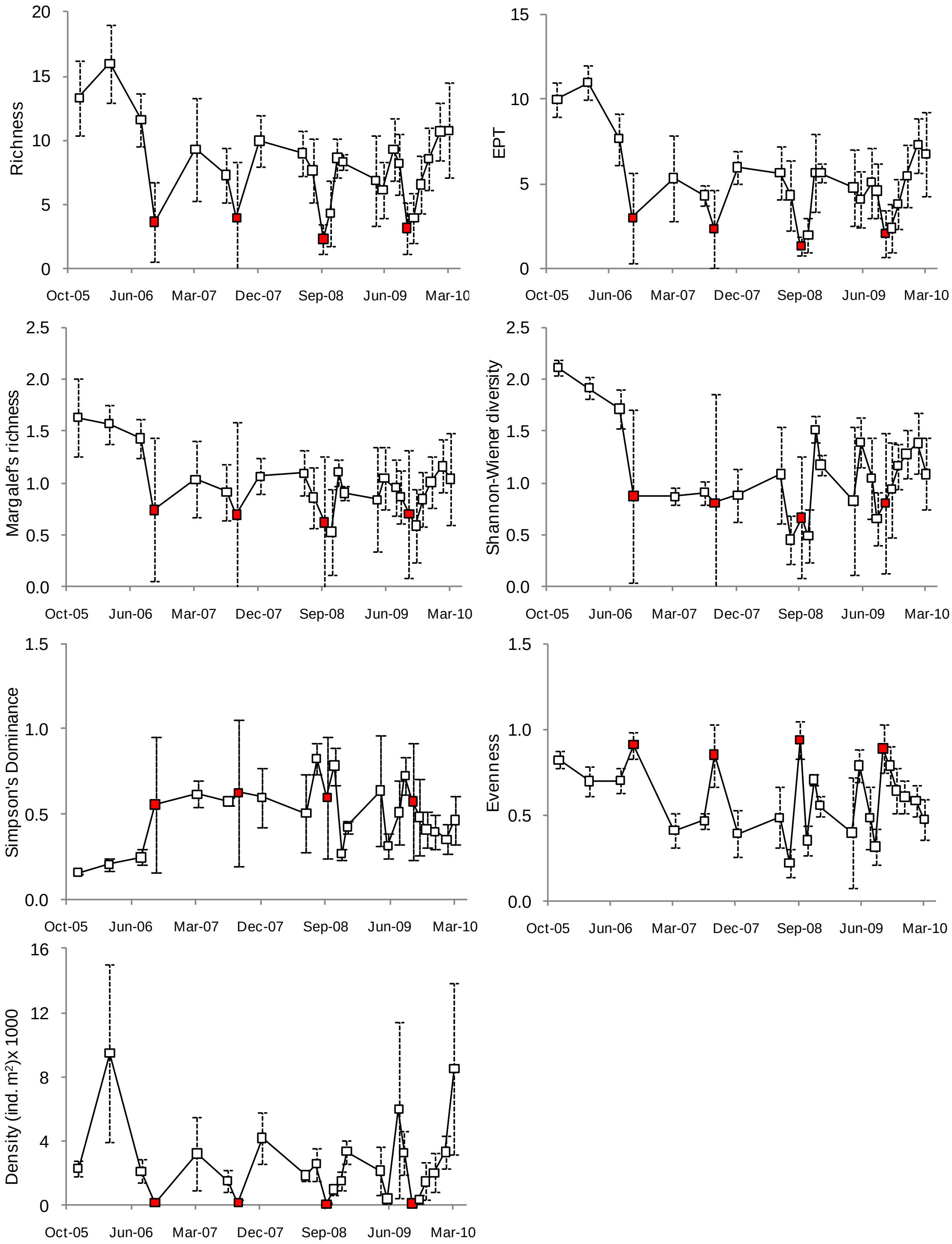


Figure 19. Mean (±1SD) macroinvertebrate density (ind. m^{-2}), taxa richness (taxa m^{-2}), EPT (taxa EPT m^{-2}), Margalef's, richness, Shannon–Wiener diversity, Simpson's dominance and evenness indices from surber samples collected on each sample date at site S3. Black symbols indicate post-flushing samples.

The reduction of Plecoptera abundance was mainly due to *Leuctra*, that decreased from 12% to 3% its density. Although mayflies continued to represent more than 40% of the community, *Baetis* became almost the dominant, representing on average 48%, while Heptageniidae that was common before representing 16%, became rarer and sporadic (~1%). Limnephilidae (shredder) changed from 1% to 9%, while Hydropsichidae (filter feeder), although sporadic even before (4%), disappeared.

Gatherer-collectors and grazer-scrapers represented (with the exclusion of the September, October and November samples) respectively 34% (SD=9) and 35% (SD=9) of the invertebrate density. Filter feeders represented 12% (SD= 14). Shredder and predators were represented by smaller percentages, respectively 7% (SD=10) and 8% (SD=6). In the samples recently affected by the flushing (September, October and November) scrapers decreased to 28% (SD=6), gatherer-collectors to 31% (SD=6) and shredder to 5% (SD=4). Consequently predators increased to 18% (SD=8). Filter feeders remained stable (13% (SD=8)). Changes that occurred in mean density, taxa richness, number of EPT taxa and other indices are described in Figure 19. Mean density values per sampling date ranged from 63.4 (2008 post flushing sampling) to 8,519.5 (march 2010). The reduction in density after each flushing ranged between 91% and 98% and averaged 95%. The values of total biomass were between 0.03 and 3.67 g m^{-2}. The lowest value was for the 2008 post flushing sample and the highest value was for March 2006. The reduction in total biomass after each flushing ranged between 85% and 96% and averaged 92%. After 3 months, recovery of the community was evident, both in terms of density and biomass, with observed values approaching pre-flushing levels.

A substantial decline of the mean number of total taxa and EPT taxa was observed after the beginning of flushing operations in August 2006: the values ranged respectively from 11.7 to 16.0 and from 7.7 to 11.0 in the samples collected before, while from 2.3 to 10.8 and from 1.3 to 7.3 in the samples collected after. Mean taxa richness and number of EPT taxa were reduced by 45 -70 % after each flushing. The highest (>0.6) and lowest (<0.4) values respectively of Simpson's Dominance and Evenness indices were measured when one taxon (*Baetis* but even Chironomidae and Limnephilidae) represented at least 75% of the community. One-way ANOVA indicated that the beginning of yearly sediment removal operations had a significant influence on T, EPT, H, D_{mg}, J and C indices (Table 8).

Table 8. Results of the one-way ANOVA testing for effects of the beginning of yearly flushing on macroinvertebrate density, T, EPT, H, D_{mg}, J and C. Values were log(x+1) transformed prior to analysis.

Variable	Effect	Sum of Squares	df	Mean Square	F	Sig.
Density	Between Groups	0.998	1	0.998	3.921	0.050
	Within Groups	30.045	118	0.255		
	Total	31.043	119			
H	Between Groups	0.171	1	0.171	18.059	0.000*
	Within Groups	1.118	118	0.009		
	Total	1.289	119			
D_{mg}	Between Groups	0.099	1	0.099	14.868	0.000*
	Within Groups	0.788	118	0.007		
	Total	0.888	119			
J	Between Groups	0.020	1	0.020	5.956	0.016*
	Within Groups	0.403	118	0.003		
	Total	0.423	119			
C	Between Groups	0.054	1	0.054	16.582	0.000*
	Within Groups	0.384	118	0.003		
	Total	0.438	119			
T	Between Groups	0.392	1	0.392	12.799	0.001*
	Within Groups	3.610	118	0.031		
	Total	4.001	119			
EPT	Between Groups	0.571	1	0.571	16.735	0.000*
	Within Groups	4.024	118	0.034		
	Total	4.595	119			

*$p<0.05$

Macroinvertebrate community after September 2006 was characterized by lower diversity, evenness and higher dominance values, confirming that some badly affected taxa had not recovered to pre-release densities (e.g. *Ecdyonurus, Rhithrogena, Crenobia alpina*) while other more resilient taxa were more abundant than before (e.g. *Baetis*, Chironomidae).

As for the short-term impact of flushing operations (comparison of 2006, 2007, 2008 pre-flushing (August) and post-flushing samples (September)) sediment removals had significant effects on density, J, T and EPT (one-way ANOVA, $p<0.01$). Evenness was significantly higher in September samples (Figure 19), since few species with few individuals were collected. Two-way ANOVA indicates significant 2009 flushing effect on macroinvertebrate density (interaction term, $F_{1,1} = 53.2$, $p<0.0001$), richness (interaction term, $F_{1,1} = 10.6$, $p<0.01$), EPT (interaction term, $F_{1,1} = 4.4$, $p<0.05$) and evenness (interaction term, $F_{1,1} = 63.0$, $p<0.0001$) (Table 10). Considering the average of all years, among the taxa still present just after (within 10-20 days) the sediment removals were *Baetis*, *Protonemura*, *Leuctra*, Rhyacophilidae and Chironomidae, while no specimen of Heptageniidae and Limnephilidae was found. Densities were very low (60- 130 individuals m^{-2}) and *Baetis* spp. made up 36-63% of the community, demonstrating high resistance to the flushing.

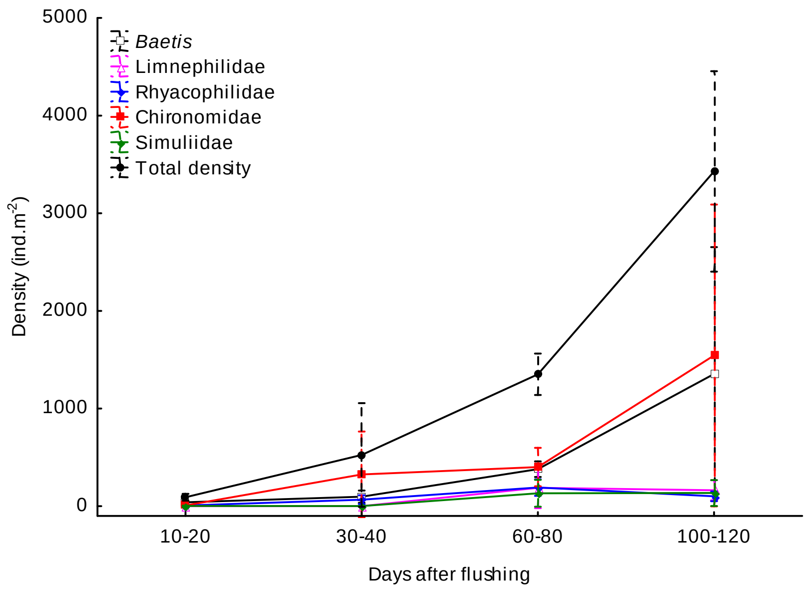


Figure 20. Increase in mean number of individuals (±SD) after the sediment removals.

Taxa that showed fast recovery responses included *Baetis*, Chironomidae, Limnephilidae, Rhyacophilidae and Simuliidae. Figure 20 presents density estimates in the post flushing samples (from september to december) for the main taxa involved in the recolonization process. Recolonization trajectories for the mean number of individuals primarily reflected recolonization by large numbers of *Baetis* and Chironomidae. Abundances of Chironomidae and Baetis increased to values >300 individuals m^{-2} respectively one month and two month after the end of sediment removals and in December they made up 65-90% of macroinvertebrate community. Limnephilidae, Rhyacophilidae and Simuliidae recolonized after 60-80 days with abundance >100 individuals m^{-2}.

Reference site (S1) was sampled monthly from May 2009 to March 2010 (except January). Hence comparison with downstream site refers exclusively to this study period. Mean density, T, EPT, DMg, H, C and J values are summarized in Figure 21.

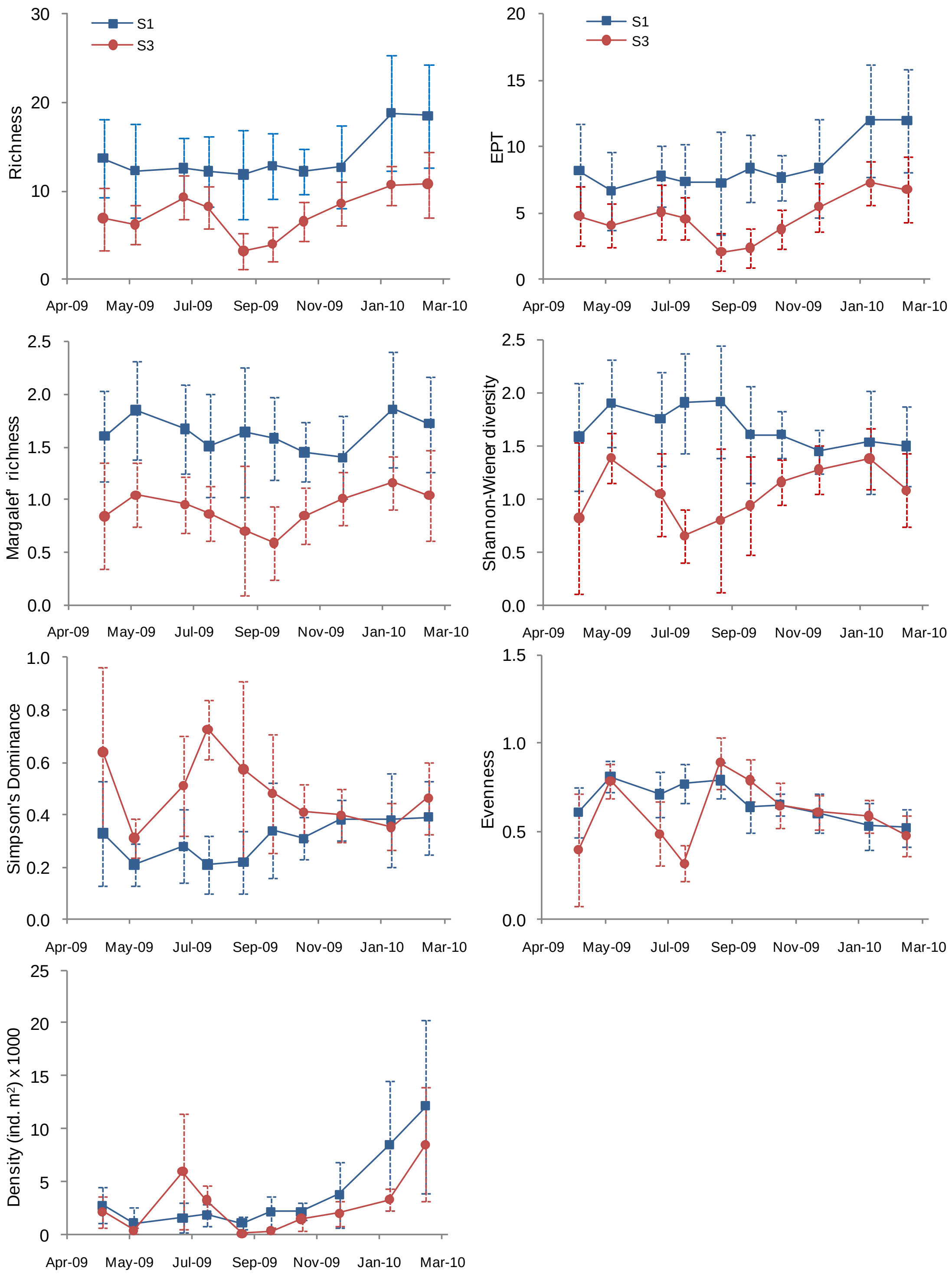


Figure 21. Mean (±1SD) macroinvertebrate taxa richness (taxa m^{-2}), EPT (taxa EPT m^{-2}), Margalef's, richness, Shannon–Wiener diversity, Simpson's dominance index and evenness from surber samples (n=10) collected at each site from May 2009 to March 2010 .

Estimates densities of the recorded taxa are presented in Appendix B. The macroinvertebrate community in the study period was 42% Plecoptera, 13% Ephemeroptera, 35% Diptera and remaining taxa included Trichoptera (4%), Tricladida (5%), Coleoptera (1%) and Oligochaeta (0.2%). Dominant taxa represented on average 34% of the community. Nemouridae and Leuctridae were the most represented families in June, July, August November and December, while Chironomidae in February, March and October. Respect to site S3, Plecoptera, with the families of Nemouridae (15% on average) and Leuctridae (25%), and not Ephemeroptera was the most abundant order. *Baetis* represented on average only 10% of community, while the family of Heptageniidae 8%. Gatherer-collectors and grazer-scrapers represented respectively 31.5% (SD=3.2) and 29.8% (SD=3.2) of the invertebrate density. Shredder and predators percentages were respectively 14.5% (SD=4.6) and 13.2% (SD=4.3). Filter feeders were represented by small percentages in all samples (mean 6.5%, SD=2.1). Little fluctuations was observed within each functional feeding guild throughout the sampling period, as demonstrated by small SD. The mean density per sample varied significantly throughout the sampling period, ranging from about 1,000 to 12,000 individuals per m^2. In particular, the abundances in June, July and September were significantly lower than those in February and March (One-way ANOVA, Scheffè test, $p<0.01$). The high flows occurring during the snowmelt determined the low densities measured in late spring. Mean taxa richness remained relatively constant between June and December (12-13 taxa per sample) and then increased in late winter-early spring, approaching a value of 19 taxa per m^2. Shannon-Wiener diversity was quite stable throughout the sampling period, ranging from 1.92 to 1.45. Similar values at site S3 were estimated for the pre-flushing period (before 2006 flushing).Late winter-early spring was the season characterized by the highest number of taxa collected, and at the same time by the highest dominance (0.4) and lowest evenness (~0.5), since *Leuctra* made up 57% of community in December and Chironomidae 40% in February and March. Evenness measured in February and March was significantly lower than in June, August and September (One-way ANOVA, Scheffè test, $p<0.01$). Mean T, EPT, D_{Mg}, H, J and C values measured at site S1 were significantly different from site S3 (One-way Anova, $p<0.001$) (Table 9). Macroinvertebrate community at the reference site was characterized by a greater number of taxa and higher diversity.

Capnia, Chloperla, Dictyogenus, Brachiptera were some of the taxa found at site S1 and not at site S3.

Table 9. Mean and SD of T, EPT, D_{Mg}, H, C and J from surber samples collected from May 2009 to March 2010 at site S1 and S3 (September, October and November were excluded from the analysis).

	S1		S3	
	Mean	SD	Mean	SD
Density	3701.17	3669.17	3652.48	2735.14
T	13.78	2.61	8.67	1.76
EPT	8.59	1.87	5.46	1.18
H	1.68	0.18	1.10	0.28
D_{Mg}	1.63	0.15	0.99	0.11
J	0.66	0.10	0.52	0.15
C	0.31	0.07	0.49	0.15

Table 10. Results of two way ANOVA testing for effects of 2009 flushing (mean comparison of August and September samples) on mean density, T, EPT, D_{Mg}, H, J and C. Values were log(x+1) transformed prior to analysis.

Variable	Effect	Sum of Squares	df	Mean Square	F	Sig.
H	Site	0.567	1	0.567	47.033	0.000*
	Date	0.000	1	0.000	0.019	0.892
	Site*date	0.000	1	0.000	0.023	0.881
D_{Mg}	Site	0.269	1	0.269	20.87	0.000*
	Date	0.005	1	0.005	0.399	0.531
	Site*date	0.017	1	0.017	1.284	0.265
J	Site	0.029	1	0.029	31.856	0.000*
	Date	0.065	1	0.065	71.276	0.000*
	Site*date	0.057	1	0.057	63.035	0.000*
C	Site	0.168	1	0.168	53.575	0.000*
	Date	0.005	1	0.005	1.617	0.212
	Site*date	0.007	1	0.007	2.094	0.157
T	Site	1.074	1	1.074	35.774	0.000*
	Date	0.399	1	0.399	13.296	0.001*
	Site*date	0.317	1	0.317	10.552	0.003*
EPT	Site	0.908	1	0.908	22.894	0.000*
	Date	0.263	1	0.263	6.635	0.014*
	Site*date	0.175	1	0.175	4.414	0.043*
Density	Site	2.644	1	2.644	23.734	0.000*
	Date	10.633	1	10.633	95.461	0.000*
	Site*date	5.925	1	5.925	53.196	0.000*

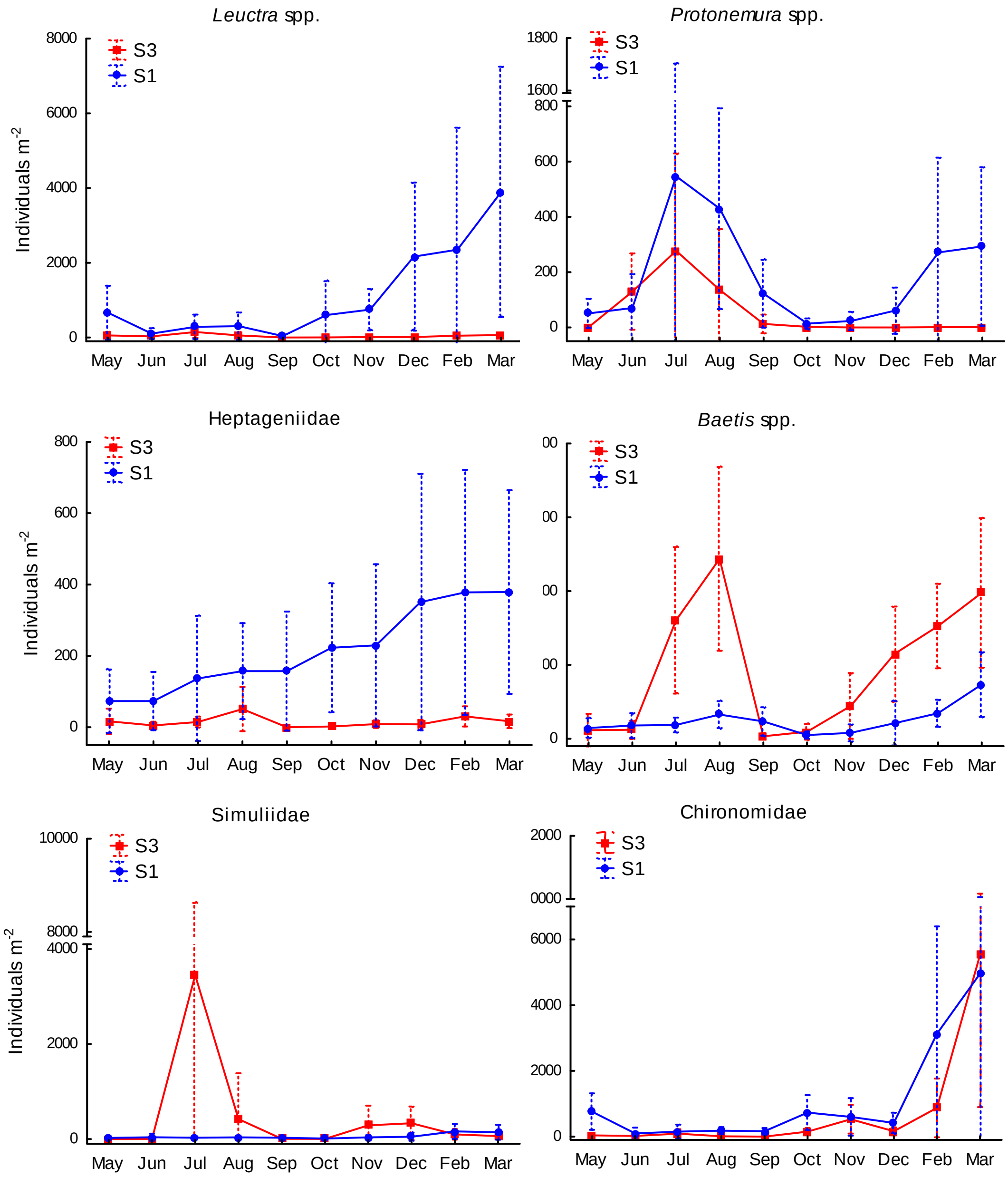


Figure 22. Mean density (±SD) of common macroinvertebrate taxa from surber samples (*n*=10) at each site collected from May 2009 to March 2010.

Figure 22 shows mean density (±SD) estimated from samples collected at site S1 and S3 during the study period for six common taxa. In September, *Leuctra* densities decreased below 40 individuals m^{-2} at the reference site, hence the reduction of *Leuctra* abundance at site S3 after sediment flushing was not significant (Two-way ANOVA, p>0.05). After September, *Leuctra* densities began to increase, becoming the dominant taxa at site S1. On the contrary, at site S3 abundances remained low (<100 individuals m^{-2}). A similar pattern was observed for Plecoptera *Protonemura*: densities naturally decreased in September at site S1 and the effect of flushing at site S3 was not significant (Two-way ANOVA, p>0.05), but while in autumn at site S1 density began to increase, at site S3 numbers remained very low (<10 individuals m^{-2}). The Heptageniidae were low in abundance at site S3 during all the sampling period. The highest density of 50 individuals m^{-2} was measured in August. After the flushing disappeared and in the following months number remained low, while at site S1 they began to increase reaching densities of 400 individuals m^{-2}. Baetidae abundance was markedly higher at site S3 respect to S1. After the flushing *Baetis* density decreases significantly (Two-way ANOVA, p<0.001), but a fast recovery to pre-flushing densities was evident. Simuliid numbers increased substantially in July at site S3, before the flushing, reaching rather high densities (3,450 individuals m^{-2}). At site S1, Simuliidae abundance was much more stable during the year and always low (<200 individuals m^{-2}). The flushing in September caused a significant decrease in simuliid numbers at site S3 (Two-way ANOVA, p<0.001). The Chironomidae increased to high densities in late winter-early spring at both sites and contributed to the re-colonization processes at site S3. Trichopterans densities (mostly Limnephilidae and Rhyacophilidae) were quite low during summer at both sites and began to increase in autumn-winter. Flushing didn't reduce significantly their numbers (Two-way ANOVA, p>0.05).

Water quality and ecological status classification

Response of macroinvertebrate community to flushing was similar among years (Figure 23). Biological quality measured according to EBI decreased of one class after every flushing and this largely depended on the significant reduction in taxonomic richness. In particular, after the 2006 flushing quality decreased from class I (not polluted or not significantly impaired) to class II (slightly impaired), while the following years from a quality class of II-I/ II (slightly impaired) to III (impaired environment). Ecological status according to STAR_ICM index was high throughout the study period at site S1, confirming the validity as reference site. At site S3 the index decreased after the 2009 flushing, peaking the lowest values in October.

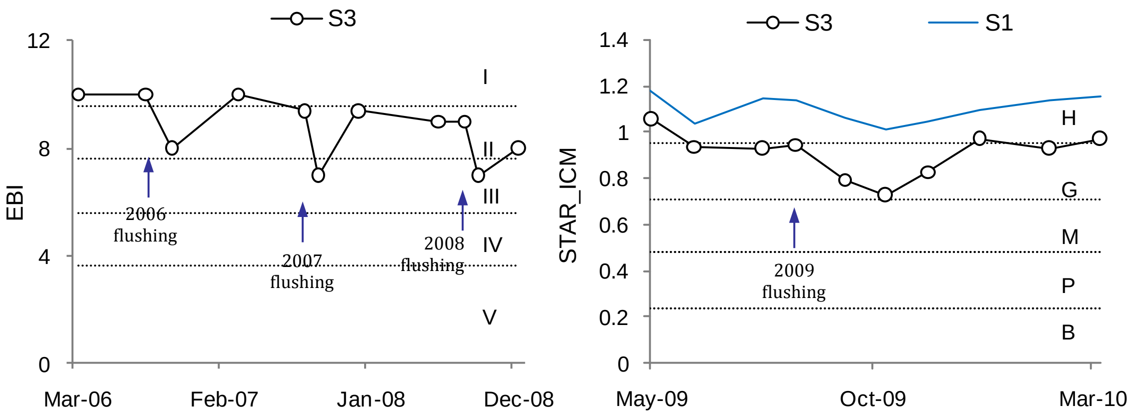


Figure 23. Trend of biological quality . EBI class: I = not polluted or not significantly impaired, II = slightly impaired; III = impaired environment; IV = poor; V = bad. STAR_ICM classes: H = high, G = good, M=moderate, P= poor, B=bad.

In December, after 3 months, values similar to preflushing condition were reached. Although the index clearly respond to sediment removal, ecological status obtained by classification of samples into quality classes didn't change. Site S3 was classified into good ecological status on all sampling date, except in May and December when the site fell into high status class. Good status was maintained after the flushing.

Multivariate analysis

A PCA was run to explore differences in taxa assemblage along time in both sites. The results of ordination analysis are presented in Figure 24. The biological differences measured between sites are supported by PCA results. Axis 1 (27.2%) and axis 2 (12.3%) discriminate the observations on the factor-plane in two main groups, corresponding to the sampling sites. Positive coordinates on axis 1 were mainly associated with samples collected at the reference site. Taxa that contribute more to this axis were *Leuctra, Perla, Crenobia* and the Diptera Psychodidae. The three samples collected before the beginning of 2006 sediment removal (blue labels) place at a certain distance from the unpaired sites, suggesting that macroinvertebrate assemblage at site S3 was already impacted before the 2006 flushing in comparison to the reference site community, as a results of the Minimum Flow release.

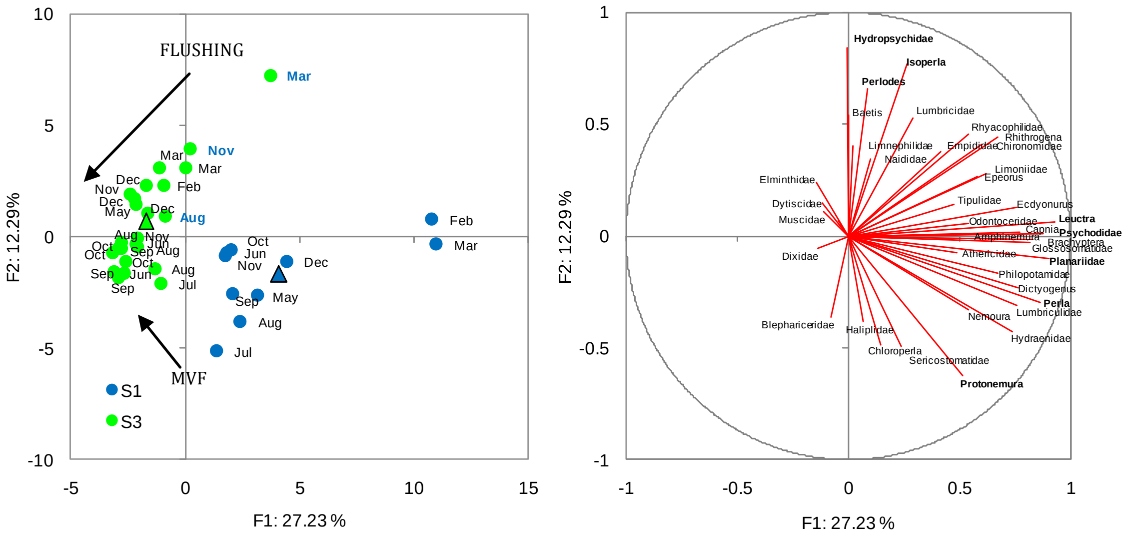


Figure 24. Ordination from a principal component analysis of invertebrate abundances at sites S1 and S3. In the factor map, triangles represent the mean for each site. In the taxa ordination length and direction of each arrow indicates the contribution of each respective taxon to axes F1 and F2. Blue labels indicate the samples collected at site S3 before the beginning of flushing in 2006.

2.5 Discussion

Sediment flushing may be effective in mitigating reservoir loss due to silting in the Alpine area, but the concentration of the sediments in the downstream sections need to be limited to meet ecological requirements. In free flow flushing events SSC can peak sharply after reservoir draw-dawn and values of several hundred grams per liter are reported (Morris and Fan, 1997); such high concentration can virtually destroy all downstream biocenosis, independently on further developing of the works. Threshold concentrations should be defined in the planning phase of operations according to the expected impacts on stream ecosystem. Unfortunately, the reliable prediction of the environmental effects of sediment flushing, and thus the planning of SSC thresholds, is hindered by the complexity of the phenomenon and by the limited scientific information so far provided. The ecological response to sediment flushing depends on a large number of parameters, often difficult to quantify because of their spatial and temporal variability, including pre-flushing condition, sediment characteristics (size and chemical), release strategy and the morphology of the effluent watercourses. In this study, a simple literature model for predicting biological effects of suspended sediment loads on fish community (Newcombe and Jensen, 1996) was used to plan a SSC limit and at the end of operations the model was tested against the collected data to verify its reliability in the planning phase of analogous operations.

Flushing period was selected balancing water availability, potential impact on fish fauna and economic loss due to Grosio powerhouse out of service. Literature (Morris and Fan, 1997; White, 2001) typically suggests to perform sediment flushing works during maximum seasonal runoff, possibly in the falling limb of a flood hydrograph. In the studied site higher seasonal runoff occurs between June and July. Neverthless, limited hydraulic capacity of the bottom outlet and connected risks losing SSC control of outflowing waters and/or compromising efficiency of mechanical dislodging, led to postpone optimal period from the end of June to the end of August. In addition, to a certain degree, water availability was not a priority as further water volumes in case of need could be provided by Premadio/Valgrosina canal. As verified in the course of this study, more severe impact of sediment releases on fish fauna was on the youngest individuals. Since brown trout reproduction is mainly between November and February, the end of August seemed a reasonable compromise between water availability and development of brown trout young of the year. Finally, end of June and the first half of July represent, at least in the last decade, the period of the year of maximum electricity request (and therefore price). Consequently, to shift sediment flushing two months after the end of June was also preferable from this point of view.

Although previous studies report that reservoir flushing can alter the water chemistry downstream of dams, mainly by reducing the oxygen concentration (Buermann et al., 1995; Hesse and Newcomb, 1982; Garric et al., 1990), in our study reservoir draining did not cause any adverse effects on DO. This positive finding can be explained by the low water residence times (τ_w= 1 d), which avoid any hypolimnetic water stagnation, with the absence of any polluting source in the watershed drained by the reservoir water collecting system, and by considering the low water temperatures (daily water temperature excursion 7 – 11°C).

The best activities for controlling the SS concentration below the dam resulted the joint use of a controlled water inflow through the empty reservoir and of an excavator dislodging the sediments from the banks of the main channel which developed on the reservoir bottom. On the whole, if compared to 2006 flushing, next years evacuations were characterized by lower SSC, both in terms of average and peak values. Difficulties in system control in the first year followed the long period the bottom outlet remained closed, a situation to be avoided with frequent opening if flushing management is planned. It should be underlined that SSC control was based, as obvious, on rough turbidimeter records (calibration was in fact performed a posteriori); nevertheless, good agreement was found between laboratory and probe data and successive corrections did not modify real-time results in a meaningful way. Daytime sediment discharge, alternated with nighttime clear water from the tributaries bypassing the reservoir, allowed successful maintainance of average SS concentrations below 5 g L^{-1} (excluding uncontrolled SSC peaks occurring during the first days of the operation). This threshold value was originally planned as a compromise solution between the need to limit the overall time required to evacuate the sediments (not exceeding 2 weeks of powerhouse stop) and by accepting a fish mortality within the range of 20-40%, resulted by applying the model of Newcombe and Jensen (1996). SEV reduction of one point could be achieved in the analyzed context either reducing the duration of the flushing or the target SSC. For both scenario, the one point SEV reduction would have lead to a remarkable decrease of evacuated sediment, which is rather ineffective considering the mean annual deposition in Valgrosina reservoir (approximately 15,000 - 20,000 m^3).

The employment of an excavator was useful in improving the scouring processes of the waters flowing into the empty reservoir. Sediment peaks due to episodic collapses of the main channel banks were not predictable and caused the highest SSC downstream of the dam, particularly during the first phase of the activities. Even if these SSC peaks were of short duration, concentrations reached high values (70-80 g L^{-1}) and possibly caused acute effects on aquatic organisms as documented in literature. For example, Garric et al. (1990) observed 100% mortality for brown trout juveniles (*Salmo trutta fario*) 5 hours from the beginning of flushing. Petz-Glechner et al. (2003) observed that gill epithelium of rainbow trout (*Oncorhynchus mykiss*) was damaged by 30 minutes of exposure to SSC of 80 g L^{-1} and Newcombe and MacDonald (1991) report 60% mortality at concentrations of 82 g L^{-1} lasting 6 hours, for juvenile salmonids. Similar acute negative effects were described for invertebrate: Doeg and Koehn (1994) measured a reduction of 19.4% in the total number of benthic macroinvertebrate taxa, and an average reduction of 63.9% in total abundance, within a day of a flushing event that increased SSC up to 4.61 g L^{-1}. Pruitt et al. (2001) reported that total suspended solids (SSC) concentrations greater than 284 mg L^{-1} resulted in biological impairment of invertebrate communities, while SSC of 58 mg L^{-1} or less during storm flow provided an adequate margin of safety and were protective of aquatic invertebrates.

Though sediment concentration directly affects aquatic organism, it must also be considered that negative biological effects of flushing operations can also arise due to the flood peak, which exerts a particularly strong influence on the aquatic invertebrate fauna (Resh et al., 1988; Lake, 2000). Experimental evidence on the biological effects of peak discharges were provided for artificial stream by Bond and Downes (2003) which showed that flow increases between 2 L s^{-1} to 12 L s^{-1} can disturb benthic fauna, and that the addition of sediment concentrations up to 600 mg L^{-1} had little effect on fauna. Effects of the flow peak on aquatic invertebrates were also described by Robinson et al. (2004) during a period of artificial floods mimicking the natural flow regime downstream of a reservoir. In this case, a peak flow of 43 m^3 s^{-1} and discharge of 25 m^3 s^{-1} for approximately 7 h (baseline flow approximately 2 m^3 s^{-1}) significantly reduced macroinvertebrate densities and changed the taxonomic structure of the community.

Unfortunately the unpredictability of the SSC peaks meant that monitoring is unlikely to be able to guide real time intervention aiming to dilute the outflow. This investigation showed that SSC limits should be referred to the average values calculated both on a daily basis and on the complete flushing period. In the present study, 10 g L^{-1} and 5 g L^{-1}, respectively. Exceptions to these limits should be considered for the first hours of the dam outlet opening. The daily average value, can be maintained by means of hydraulic operations aimed to increase the dilution processes during flushing, while the overall average provides a reference point for planning the duration of the flushing operation. Threshold value defined as instantaneous non-harmful concentrations as commonly suggested by existing regulations, are not meaningful for flushing activities because of the involved scales (sediment and available water volume, flush duration). For example the Italian National regulations (Legislative Decree n. 152) identify SSC of 60 and 80 mg L^{-1} as limits for the protection of salmonids and cyprinids, respectively.

Same considerations can be formulated by considering the SSC limits quoted in the literature. Although many toxicological data are reported for aquatic species (for an exhaustive review see Newcombe and Jensen, 1996), dose-response experiments based on individual species do not provide effective guidelines on threshold limits. In fact, the obtained results can exhibit a wide range with respect to critical sediment concentrations. For juvenile Salmon (Chinook), authors reported a mortality rate of 50% for exposure duration of 36 h with respect sediment doses ranging from 1,400 mg L^{-1} to 9,400 mg L^{-1} (Newcomb and Flagg, 1983). Variability in tolerance to SSC could be explained by sediment particle characteristics, water temperature, species differences and other stressor that might have synergistic effects (Bash et al., 2001).

Fine sediment accumulated in the channel, estimated from SSC and discharge measurements, was a small amount (<10%) of the total transported material. Geometry of the channel and discharge utilized for sediment removal, together with the fine grain size, caused almost all material in suspension to be transported downstream. Downstream deposition patterns were comparable among years and zones between bankfull and wetted channel at minimum flow were the most affected by fine sediment deposition. We observed a moderate aggradation along the pool margins, but not a substantial filling as previously reported for other channels subject to large influxes of sediment. Lisle (1982) described preferential filling of pools along channels in northern California during the December 1964 rainfall floods in that region. Wohl and Cenderelli (2000), during a reservoir sediment release (~7000 m^3 of silt-to pebble-sized sediment) on the North Fork Poudre River, observed that deposition along the 12 km of channel downstream from the reservoir occurred primarily in pools, filled up to 4 m deep. Several factors regulated the specific sedimentation patterns present at a pool, being function of distance downstream from the primary sediment source, of discharge magnitude and duration since the initial sediment release, of pool geometry and of sediment transport and storage along the riffle immediately upstream from the pool (Wohl and Cenderelli, 2000). The lateral constriction presents in pools may create a central jet of relatively high velocity flow with associated flow separation and eddy circulation along the pool margins (Thompson, 1997). During high-moderate flows this central jet scours the pool thalweg but does not prevent substantial deposition along the lateral eddies or on the pool exit slope, thus explaining sedimentation patterns in pools observed during river surveys.

Riffle units were covered with a thin layer of fine sediment causing clogging of the substrate. Although the interstitial sediment was a negligible fraction on the total mass of transported sediment during flushing, the loss and degradation of habitats arising from fines deposition might cause negative effects on macroinvertebrates and fish fauna.

As for the negative effects of flushing on aquatic organisms, chronic and acute effects were measured for the fish populations; i.e., a substantial decrease in fish weights and a severe juvenile mortality rate. As suggested by Lloyd (1987) it could be assumed that the high levels of SSC were lethal to sensitive specimens (other than to sensitive individuals), whilst prolonged lower levels of suspended solids and turbidity was the cause of chronic sublethal effects, such as reduced weight since individuals are not able to feed efficiently (Sigler et al., 1984; Herbert and Richards, 1963). Both of these biological effects are expected to modify fish population dynamics by threatening recruitment, and indicate that the natural recovery of trout populations will only occur over a period of many years.

Density of brown trout population markedly decreased after the beginning of the flushing operation in September 2006. Unnatural high densities before 2006 flushing, well above the reference of about 2,000 ind. ha^{-1} reported for analogous alpine situations (Forneris et al., 2006), the presence of sport-fishing re-stocking individuals, the high SSC peaks (more than one cumulative day with SSC over 10 g L^{-1}) could have contributed to this result. 2008 and 2009 sediment removals determined a relative low reduction of brown trout density and biomass respect to previous years. Data collected at section S4 provided evidence that fish population reached a stable condition, approaching density of about 1,700 – 2,000 ind. ha^{-1}, whilst at site S3 density continued to decrease over time. Moreover the high number of marked and recaptured fish in 2008 suggests that showed results are only slightly affected by fish movements. The decreasing short-term impact of flushing on fish populations, although mean concentrations and durations were approximately the same over the years, suggests that other factor can be relative more important in determining the entity of the impact. Juveniles are the most affected by flushing and survivorship would seem strongly correlated to initial abundance of individuals before the sediment removal. Limiting factors, such as competition for space (i.e. suitable refuges from the flood) and availability of food (extremely reduced after flushing) may create a density-dependent mechanism of survival that contribute to determine population size by reducing the carrying capacity of the environment.

Density-dependent mortality operates for comparatively short periods of Salmonid life cycle, during critical stages when regulation is achieved by competition for limited resources. Carrying capacity, as determined by habitat features (Armstrong et al., 2003), is independent of density, but creates a bottleneck, typically for space and food, that increase competition, thus leading to density–dependent effects (Milner et al., 2003). Key stages where such bottleneck have been demonstrate are the early post-emergence fry stage and at spawning when limited availability of spawning gravel can cause density dependent regulation of breeding female trout numbers (Elliott and Hurley, 1998).

Natural spates have been reported to induce downstream displacement or mortality in stream fish population (e.g. Carline and McCullough, 2003; Pires et al., 2008). Life stages of fish involved can be as critical as the spate's magnitude: young salmonids are more vulnerable to displacement by high water velocity and a rapid decline in susceptibility to washout has been observed with increase in size (Ottaway and Clarke, 1981; Heggenes and Traaen, 1988; Harvey, 1987). Nonetheless, several authors have documented the ability of stream fishes to withstand spates, especially salmonids. Habitat complexity and availability of low-velocity microhabitats have been reported as key factors related to resistance to floods by fish (Pearsons et al., 1992; Lobón-Cerviá, 1996). This is the first study, to the knowledge of the author, demonstrating that resources availability (habitat and food) may control juvenile survival to 'artificial spates' through a density-dependent mechanism.

Even if the presence of young of the year before each flushing indicates the success of natural reproduction for both investigated sites, a decreasing number of young specimens was caught along the years. This reduction was particularly important at site S3, since it was associated even to a steadily reduction of the proportion of young of the year. Several factors are possibly involved in the declining of recruitment, first the diminution in weight observed after the flushing due to decrease in food density (macroinvertebrate) as well as to increased metabolic rates resulting from stress provoked by high concentrations of suspended solids (Sigler et al., 1984; Barton and Schreck, 1987; Bergstedt and Bergersen, 1997). In fact, although fish may survive high concentrations of suspended solids often over prolonged periods they may subsequently suffer reduced fitness through damage of the gill epithelium through particle abrasion and clogging (Herbert and Merkens, 1961).

Sedimentation in river bed, and in particular in spawning habitats, has been recognized as one of the most important factors limiting the natural reproduction of salmonids in streams (Cordone and Kelly, 1961). Fine sediments may infiltrate into the interstitial spaces reducing substrate composition, porosity and permeability, thereby restricting the interchange of water between the fluvial and intragravel environments (Young et al., 1991; Haschenburger and Roest, 2009). This in turn can lead to lowered intergravel dissolved oxygen levels, accumulations of metabolic wastes and eventually prevents successful development and survival of fish eggs and larvae (McNeil, 1966; Buerman et al., 1995; Soulsby et al., 2001). Many field and laboratory studies (most of these concerning several native salmonids of USA) have noted that the survival of salmonids from egg to emergence declines as the percentage of fine sediment increases (Chapman, 1988; Reiser and White, 1988; Lisle, 1989; Bagliniere and Maisse, 2002; Armstrong et al., 2003; Jensen et al., 2009; Yamada and Nakamura, 2009; Sternecker and Geist, 2010; amongst others). According to Argent and Flebbe (1999) the survival of the emerged fry of brook trout (*Salvelinus fontinalis*) was inversely related to fine sediments and was reduced in half when redds were covered of 25% of fine sediment (<0.085mm). Soulsby et al. (2001) found in a Scottish River that when fine sediment less than 2 mm in size reached levels of 20%, egg mortalities of Atlantic Salmon (*Salmon salar*) and Sea Trout (*Salmo trutta*) were as high as 86%. Low egg-to-fry survival of brown trout was correlated with a high percentage of fine sediments in different studies carried out by Acornley and Sear (1999), Massa et al. (1998) or Rubin and Glimsater (1996). Olsson and Persson (1988) observed that when 20% (volume) sand was added to coarse gravel in simulated redds of brown trout, a decrease from 90% to 28% in ova and embryo survival was found. Furthermore, a high proportion (55%) of surviving alevins emerged at a premature stage. Kondolf (2000) reported that percentage finer than 1 mm (or 0.85 mm) corresponding to 50% emergence of different species of Salmonids was on average 14% in various field and laboratory studies, close to the standard of 12% indicated by McNeil and Ahnell (1964). Consequently the same author suggested that the fine sediment (finer than 0.83 mm or 1 mm) threshold for a successful incubation (at least 50% emergence success) can be estimated at 12–14%.

In our study, particle size analysis of core samples collected in riffle habitat, despite substantial within-site variability (due to potentially different sampling environments, i.e. proximity to thalweg or streambank) provided evidence of a significant increase in fine sediment. Most of the core sediment samples had a content of finer than 0.85 mm below the standard proposed by Kondolf (2000) and only three samples showed values greater than 12%. Moreover, exclusions of grains larger than 25 mm artificially increases the percentage of remaining fine sediment, consequently the values found in our samples were overestimated. Tappel and Bjornn (1983) proposed that the quality of spawning gravel be assessed based on the percentages finer than 0.85 mm and finer than 9.5 mm out of the portion of the size distribution finer than 25.4 mm. They provided equations to predict embryo survival of steelhead (*Salmo gairdneri*) and chinook salmon (*Oncorhynchus tshawytscha*) as a function of percentages finer than 0.85 mm and finer than 9.5 mm. Even if developed for other species of Salmonids, their equations can be useful in our study to delineate the magnitude of the effect, since they excluded particles greater than 25 mm. According their model, % embryo survival passed from 72% to 58% for steelhead and from 55% to 42% for chinook salmon. As suggested by the same authors, even if survival in the stream was not the exact value calculated through the formula for a given substrate before increases in fine sediment occurred, the estimated reduction (~15%) in embryo survival predicted by the equations should be close to the decrease in embryo survival in the stream. Consequently, although quality of bed material seems still acceptable to guarantee successful incubation, we can't exclude that fine sediment deposition negatively affected eggs-to-fry survival, even if moderately.

Change of substrate composition following deposition of fine sediment may affect juvenile and adult salmonid too. In particular the firsts can show reduced growth and survival in heavily silted bed since the sediment infilling reduced the availability of refugia offering protection from strong currents, predator and high temperature (Suttle et al., 2004; Milan et al., 2000). Moreover quantity (other than quality) of suitable spawning habitat can be reduced (Furniss et al., 1991; Crowe and Hay, 2004), with possibly problem of competition among adults for suitable spawning sites (Bjornn and Reiser, 1991).

The synergy of direct effects, resulting from SSC and Q peaks during flushing, and indirect effects, resulting from modifications of riverine habitats (i.e. clogging), can seriously threat the persistence of brown trout populations in the Roasco stream, especially at site S3. The trout population is blocked into an area whose extension is comprised between the dam and the gauging station downstream (beginning of site S4), which avoid fish to move upstream. The lack of connectivity with downstream river sector, prevents natural migrations movements during reproductive season and makes this population particularly endangered. Under current conditions, resilience to environmental perturbations is probably limited. On the contrary, at site S4, natural fish movements contribute to maintain a viable population and to counterbalance the factors that negatively affect the reproductive success of resident trout population. In this respect, it's very complex to assign a weight to the factors that negatively affect recruitment. The observed difference between the two stations suggests that, probably, the reduced fitness is the main cause. In fact, the migration of individuals during reproductive season can overcome this problem, but certainly not the alteration of spawning habitats quality. It is also true that the juvenile dispersal from neighboring sections throughout the year, may lead to a re-colonization of the study reach regardless of the reproductive success of resident trout population or spawning habitat quality.

Taking into account the complexity of the biological effects under study, the model used for predicting the expected fish mortality (the SEV model), can be considered adequate for delineating the order of magnitude of the short term effects of the flushing. Noticeable disagreement at site S3 for 2006 event is probably due to unusual pre-flushing density and abundance of not wild specimens artificially introduced for sport-fishing purpose. Furthermore, the high SSC peaks of 2006 event could have increased fish mortality. Moreover, as discuss above, other factors, such as density of individuals and competition for limited resources, may determine quite different population response to the same SSC.

Invertebrate assemblages exhibited a severe reduction in overall abundance and biomass. This negative effect was expected, as detectable impacts of SSC on macroinvertebrate communities may occur at concentrations between 100 and 150 mg L^{-1} (Doeg and Milledge, 1991; Doeg and Koehn, 1994), values well below those measured during the present study. However, this negative consequence of the desilting work was considered acceptable since communities appeared to recover to near pre-release conditions after 3 months (similar findings in Ciutti et al., 2000). It is difficult to determine whether the declines in benthic fauna abundance and diversity measured during the flushing events can be attributed to the hydraulic force (Resh et al., 1988; Lake, 2000) or to the high SSC since these factors may interact (O'Hop and Wallace, 1983). The findings of controlled experiments on this subject (Bond and Downes, 2003) are limited to maximum SSC lower than those measured in this study (i.e., < 600 mg/l) and suggest that flow increase alone can disturb benthic fauna. In our investigation, considering the magnitude of flow (Q increase from <1 $m^3 s^{-1}$ to values >6 $m^3 s^{-1}$) and SSC (up to maximum values of 70-80 g L^{-1}) during peaks, it is reasonable to assume that both the physical disturbance of the flowing waters (i.e. shear stress, river bed overturn) and the suspended solids (i.e. abrasion by transported suspended particle) contribute to damaging benthic invertebrate communities. Recolonization trajectories after sediment removals primarily reflected recolonization by large numbers of *Baetis* and Chironomidae, plausibily by rapid reproduction (Matthaei et al., 1996). This finding is in agreement with previous studies on the argument (Zuellig et al., 2002; Carvalho and Uieda, 2006). Macroinvertebrate stream communities most likely to recover quickly after disturbance are often dominated by baetid mayflies, taxa with multivoltine life histories and great dispersal ability (Mackay, 1992; Zuellig et al., 2002). For example, Robinson et al. (2004) found that taxa that showed a fast recovery to multiple experimental floods in the river Spöl (Switzerland) included Chironomidae, Baetidae and Simulidae. *Baetis*, in particular, appeared highly resilient to multiple floods. Although densities and biomass approached pre-flushing values in a few months, macroinvertebrate community after September 2006 was characterized by lower biodiversity, confirming that certain affected taxa had not regained their pre-release densities such as Heptageniidae (e.g. *Ecdyonurus* and *Rhithrogena*), while other were more abundant than before (e.g. *Baetis*). This results clearly indicate a response of the river biota to the cumulative impact of yearly sediment removals (change of disturbance regime). The composition and relative abundance of the species that are present in a stream often reflect the frequency and intensity of high flows (Meffe and Minckley, 1987; Scarsbrook, 2002). Consequently, a yearly occurrence of sediment flushing may rejuvenate the biological community, allowing species with fast life cycle and good colonizing ability to reestablish, but may prevent successful colonization of species with longer life cycles. Moreover, over the longer term, the continuing presence of fine sediment along the channel may inhibit re-colonization by aquatic insects. The composition of the bed sediment plays a key role in the colonization mechanism, composition and abundance of benthic community and increase of sedimentation strongly affects the benthic stream invertebrate community (Waters, 1995, Gayraud and Philippe, 2003; Rabení et al., 2005; Larsen et al., 2010). Experimental evidence on the biological effects of clogging were provided, for example, by Bo et al. (2007) which found significant differences in taxa number and abundance, both decreased with increasing clogging, among four trap types placed in the riverbed of the Lemme stream (NW Italy) and filled with increasing content of sand (from 100% gravel to 100% sand). Rheostenic and litophilous elements, such as

Ephemeroptera Heptageniidae, were almost exclusively found in gravel-filled traps, avoiding more clogged substrata. On the contrary, the number of Oligochaeta tends to increase in the traps filled with conspicuous amounts of fine elements.

Sediment deposited in the wetted channel after the flushing was gradually washed away during the following months. However, the release of a minimum vital flow (see chapter 3) and the consequent absence of peak flows, that in natural condition remove accumulated fine sediment and loosen the gravel bed, may delay channel recovery from the increased sediment influx and contribute to extend during time riverbed impairment.

CHAPTER 3

Effects of Minimum Flow releases on benthic macroinvertebrates

3.1 Introduction

Numerous studies have shown the importance and the role of a river's flow regime for sustaining physical and biological processes of lotic ecosystem (Bunn and Arthington, 2002; Lloyd et al., 2003; Doyle et al., 2005; Poff and Zimmerman, 2010). Discharge, and its variability, has been suggested to be a ''master variable'' that limits the distribution and abundance of species (Power et al., 1995) and regulates the ecological integrity of flowing water systems (Poff et al., 1997).

Natural flows span a wide range, including periods of low flow resulting from precipitation deficits. Low flows are often seasonal and occur at a similar time each year, but human activities, such as water diversion or dam closure, can artificially create or extend low flows that deviate from the natural flow regime. Given the pervasive effects of flow on the structure and function of stream ecosystems, the potential effects are numerous and depend upon the extent to which the natural pattern of flow is altered (Poff et al., 1997; Richter et al., 1997). Assessing how much water a river needs to prevents adverse effects on the environment, while still enabling water diversion, is a task many water resources managers face. However, there is shortage of robust scientific data describing the quantity and timing of water required to protect stream ecosystems and, currently, there are no generally applicable techniques that can be used to predict how low the flow can become without a stream community becoming temporarily or permanently impaired.

Among aquatic organism, macroinvertebrate play a crucial role in the energy dynamics of lotic system, acting as shredders, collectors and filterers of organic matter (Cummins, 1993) and serving as a primary food source to the fish community and other aquatic organisms. Freshwater macroinvertebrate provide an excellent indicator of river ecosystem health and have historically been used to develop system descriptive indices, e.g. number of Ephemeroptera–Plecoptera–Trichoptera (EPT; Lenat, 1983) and species richness (Hering et al., 2004). Although accounting for variability in hydraulic conditions has been recognized as important in ecological quality assessment (e.g. Monk et al., 2006; Buffagni et al., 2009), none of the actual assessment systems include flow-related metrics. Flow exerts a direct physical force on stream organisms and affects them indirectly by influencing substrate composition, water chemistry, resource acquisition, habitat availability and suitability (Statzner et al., 1998; Hart and Finelli, 1999) (Figure 25).

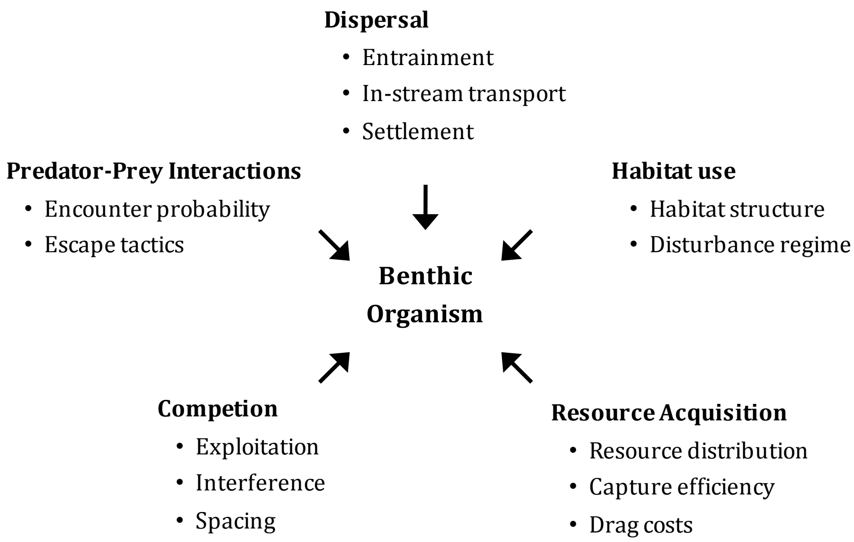


Figure 25. Alternative causal pathways by which flow can affect benthic organisms (Modified from Hart and Finelli, 1999).

Invertebrate community changes with reduce discharge are probably a result of changes to the instream environment. With decreasing discharge there is commonly a loss of wetted area (e.g. Gore, 1977; Dewson et al. 2007a; Dewson et al. 2007c) and reduced water velocity and depth (e.g. Minshall and Winger, 1968; McIntosh et al., 2002). Changes to nutrient concentrations (e.g. Ladle and Bass, 1981; Rader and Belish, 1999), increased water temperatures (e.g. Everard, 1996; Rader and Belish, 1999) and lowered dissolved oxygen levels (e.g. Everard, 1996; Dewson et al., 2007a) have also been reported as response to discharge reduction in permanent streams. Reduced discharge can both reduce (e.g. Englund and Malmqvist, 1996; Wood and Petts, 1999; McIntosh et al., 2002) and increase (e.g. Gore, 1977; Wright and Berrie, 1987) invertebrate abundance in different situations. However taxonomic richness usually declines with discharge (e.g. Wood and Armitage, 1999; Wood et al., 2000) and several previous studies have implicated reduced discharge in increasing active drift (Minshall and Winger, 1968; Gore, 1977; Death et al., 2009). The habitat changes generated by water abstraction vary widely, depending on characteristic such as channel morphology, substrate stability, nutrient enrichment, stream size and temperature regime. Thus, the highly variable results from studies of invertebrate responses to reduced discharge are not surprising (Castella et al., 1995; Dewson et al., 2003).

Most of our current knowledge of invertebrate community response to reduce flows comes from investigations of droughts and seasonal low flow, or from studies that investigated the effects of altered flow regimes from dams (mainly large rivers). Investigations of the effects of reduced flow on macroinvertebrate communities in small to medium mountain streams are relative few, moreover studies provides contradictory results that vary from no effects (Roy and Messier, 1989; Castella et al., 1995; Rader and Belish, 1999) to reduction in invertebrate diversity and abundance (Dudgeon, 1992; Petts and Bickerton, 1994; Englund and Malmqvist, 1996; Mckay and King, 2006).

The release of a minimum flow (Minimum Vital Flow, MVF) as mitigation measure of the negative effects of dams and diversions on aquatic biota, has been established by Italian law since the early ‘90s. Although the recent development of the regulatory framework allow the seasonal scheduling of water releases to accomplish environmental flow requirements, these management options are seldom applied and the MVF is simply reduced to a low flow limit maintained during the whole year. Accordingly, concerns arise about the effectiveness of MVF as a protection measure for aquatic biota.

The main objective of this chapter was to evaluate the use of macroinvertebrate community for detecting the effects of flow reduction (MVF) in Alpine stream ecosystems. Specifically, the response of single and multimetric indices, as well multivariate approaches, to water diversion was tested. We used a reference condition approach, analyzing the composition of macroinvertebrate assemblages at sites that were affected by a different degree of water diversion and reference site spread across the same geographic region. The information presented in this chapter are the results of the first year of the EU Interregional project ECOIDRO, *Water Use and Safeguard of Environment and Biodiversity in the River Basins of Adda, Mera, Poschiavino and Inn*.

3.2 Study area

The study was conducted in the Province of Sondrio, a territory located in Northern Italy and corresponding to the northern part of the Adda river basin. The overall surface of this alpine sector amounts to 3,212 km^2, and it consists of two main sub-basin sectors, the Valtellina and Valchiavenna valleys. The main river stems that drain these valleys are respectively Adda and Mera river, both tributary of Lake Como. Adda is the major river of the province of Sondrio, it flows from north-east to south-west in the first part and from east to west in the second, for an overall length of 119 km and an area of 2,598 km^2. At the bottom of the Valtellina (198m a.s.l.), it flows into lake Como, with an annual average discharge of 88.0 m^3 s^{-1}. The area is characterized by a continental pluviometric regime with a summer maximum and a winter minimum. Average annual rainfall ranges from 800 mm to 2,000 mm, being the highest values measured in the southwestern part of the province, near lake Como, and the lowest in the northeastern portion. Annual runoff cycle is greatly influenced by snow storage and melt dynamics and is regulated by means of hydroelectric reservoirs that retain at their maximum capacity, an overall volume of 419.1 millions of cubic meters (Mm^3).

The province of Sondrio is characterized by intensive exploitation of water for hydroelectric energy production (Figure 26), with about 80% of rivers and streams within the watershed diverted for hydropower generation (Songini, 2004). Table 11 illustrates the characteristics of the hydroelectric power plants and the related facilities, divided by the four main companies that hold concessions for water abstraction and production of electricity: Edipower, Enel, Edison and A2A. This complex system of channels, diversions, dams and plants together form the main area of hydropower generation in Lombardy region, and one of the most nationally relevant in Italy. The production within this territory, in facts, covers alone almost half of hydropower capacity of the region Lombardy and about 12% of the total hydroelectric production in the country (Autorità di Bacino del Fiume Po, 2001).

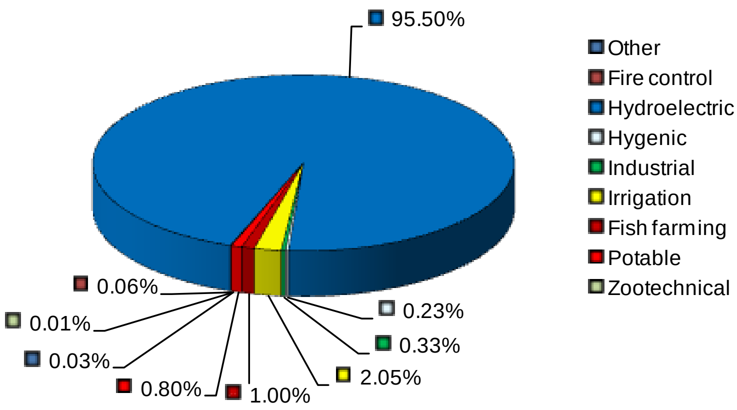


Figure 26. Mean annual flow licensed for abstraction distinguished by water use typology (Provincia di Sondrio, 2008).

Table 11. Number and characteristics of the hydroelectric facilities in the province of Sondrio (Modified from Songini, 2004).

	Water intakes	Water channels and pipes		Turbines	Average annual production (1997-2001)	Reservoirs	
		Free (km)	Pressurized (km)			Amount (n)	Overall capacity ($m^3 \times 10^6$)
	(n)			(n)	($kWh \times 10^6$)		
EDIPOWER	53	27.4	34.1	21	1108.5	14	59.33
ENEL	70	31.9	76.4	31	1777.6	17	93.54
EDISON	73	18.8	42.4	18	855.3	12	76.82
A2A	61	11.5	126.5	19	1991.9	8	189.32
OTHER	46	-	17.0	33	90.0	5	0.04
OVERALL	**303**	**89.6**	**296.3**	**122**	**5823.3**	**56**	**419.06**

Currently, the release of a Minimum Vital Flow is the main measure for mitigation of the environmental impacts of dams and weirs in the area (Greco et al., 2004). The MVF is calculated either through a regional formula or through an experimental approach (i.e. site-specific case studies). The regional formula follows the River Po Watershed Authority criteria and involves the release of at least 10% of mean annual discharge (hydrological component).

3.3 Methods

3.3.1 Hydrological alteration gradient

The stream sampling sites (n=30) were selected to represent different degree of hydrological alteration, according to their location with respect to main flow diversions in the study area (Figure 27). The sites were classified into classes based on the severity of flow reduction using the ratio between the mean annual flow under current (Qc) and natural condition (Qn) as an index of hydrological stress (H_s). Five H_s classes, corresponding to increasing deviation from natural mean flow, were considered: 0 – reference sites, 1 (mean annual flow between 35 and 100% of Qn), 2 (mean annual flow between 20 and 35% of Qn), 3 (mean annual flow between 10 and 20% of Qn) and 4 (mean annual flow ≤10% of Qn). Discharge per unit area estimated for each sub catchment by the Basin Authority of the Po River (Autorità di Bacino del Fiume Po, 2001) was used to calculate Qn knowing the catchment area at the measurement section. Monthly discharge measurements using a cross-sectional method (Gordon et al., 1992) were carried out at impacted sites. The average of measured values was used to calculate Qc and H_s class for each stream segment. Both type of sites (reference and impacted) were exposed to anthropogenic influences other than water diversion, which was appropriate for our purpose because we aimed to assess only the impact of abstraction, not the impact of all human activity.

3.3.2 Macroinvertebrate sampling

Macroinvertebrate samples were collected according to the AQEM sampling strategy (Hering et al., 2004; Buffagni and Erba, 2007), a quantitative multihabitat sampling approach developed in accomplishment of Water Framework Directive (2000/60/EC). Ten quantitative sample units were proportionately allocated in relation to the occurrence of microhabitats in the studied reaches. The invertebrate samples were collected during 2009 (Table 12) using a standard Surber sampler (500 µm mesh). Collected macroinvertebrates were preserved in formalin (4%), identified to genus (Plecoptera, Ephemeroptera, Tricladida) or family level and counted. One year of work was not enough to conclude a seasonal sampling (four seasons) for all 30 sites, hence for some streams only one sampling season data (late summer-early autumn) were available.

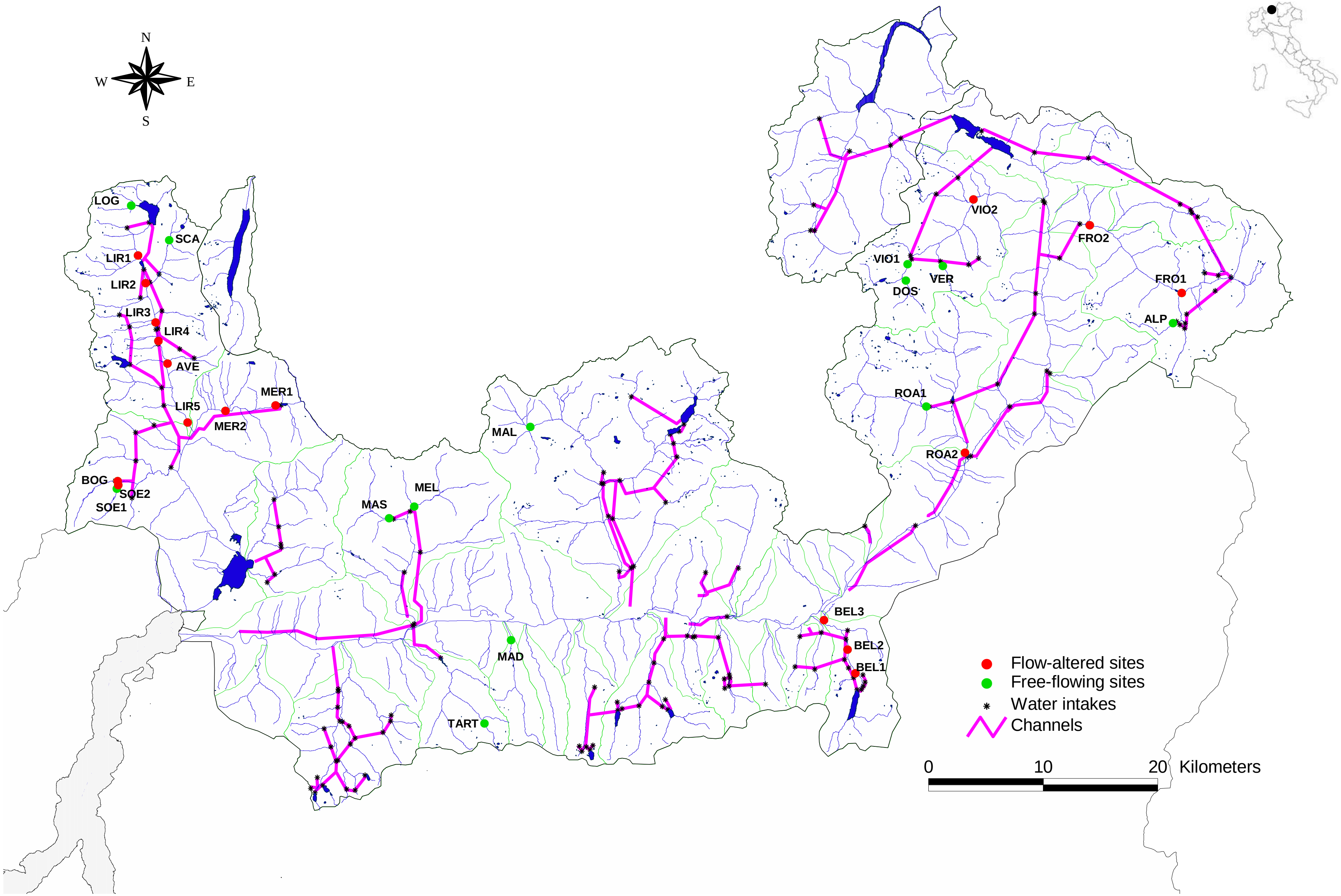


Figure 27. Map of the study area with sampling sites and main diversions.

3.3.3 Environmental data

Geographical and typological information were collected for each site: geographical coordinates, distance to source, upstream drainage areas, percentage of glaciated area in the catchment (glaciers and permanent snow), altitude and slope. Slopes were calculated from the distance along the thalweg between the contour lines crossing the stream upstream and downstream each sampling site. On-site environmental observations and measurements were made at the times of macroinvertebrate sampling. The occurrence (10% steps) in the studied reaches of different microhabitat, mineral and biotic (Buffagni et al., 2007a), and the shading by the riparian canopy (scored on a 4-point scale corresponding to increasing cover) were visually evaluated. The main physico-chemical parameters were measured during the samplings using field multi-probes or following standard methods (APAT-IRSA, 2004): water temperature (°C), conductivity (μs cm^{-1}), dissolved oxygen (mg L^{-1}), pH, $N\text{-}NO_2$ (mg L^{-1}), $N\text{-}NO_3$ (mg L^{-1}), $N\text{-}NH_4$ (mg L^{-1}), total phosphorus (TP, mgL^{-1}), P-orthophosphate ($P\text{-}PO_4$, mg L^{-1}), BOD_5 (mg L^{-1}) and COD (mg L^{-1}). In order to evaluate the chemical alteration of stream sites, concentration values of ammonium, nitrate, total phosphorus, COD and BOD_5 were classified into the five levels (corresponding to increasing concentration) defined for each macro-descriptor in the LIM (*Livello di Inquinamento da Macrodescrittori,* 152/99, 1999) assessment method. $N\text{-}NH_4$, $N\text{-}NO_3$, TP, COD and BOD_5 respectively less than 0.03, 0.3, 0.07, 5 and 2.5 mg l^{-1} were classified as first class, less than 0.1, 1.5, 0.15, 10 and 4 mg l^{-1} as second class, less than 0.5, 5.0, 0.30, 15 and 8 mg l^{-1} as third class and less than 1.5, 10, 0.60, 25 and 15 mg l^{-1} as fourth class. Finally, values greater than these thresholds were classified as fifth class. Functional description of the stream segments was performed with the application of the Fluvial Functional Index (FFI) index (AA.VV., 2007). The FFI is a development of the Riparian and Environmental Inventory (RCE) created by Peterson from the Institute of Limnology at Lund University (Petersen, 1992). This index provide a synthetic assessment of the quality of fluvial environment and of its functionality on the basis of specific biotic and abiotic properties (e.g. riparian area, land use, physical and morphological structure of the banks, structure of the river bed, flow, biological characteristics). A specific form divided in 14 question is filled in the field. There are 4 possible responses to each question and for each answer there is a fixed score. The final score, called FFI value, is the sum of the answer scores, and classifies river stretches surveyed into in five levels of functionality, where level I is the best and level V the worst situation.

3.3.4 Data analysis

A Principal Component Analysis was applied to the normalized environmental variables for ordering the stream sampling sites. Average invertebrate density (individuals m^{-2}), number of Ephemeroptera, Plecoptera Tricopthera taxa (EPT m^{-2}), richness (taxa m^{-2}), Shannon diversity index, Simpson's Dominance index (Simpson, 1949) and the Lotic-invertebrate Index for Flow Evaluation (LIFE) (Extence et al., 1999) were calculated from samples collected with the quantitative multihabitat approach. The LIFE score is a method for linking benthic macroinvertebrate data to prevailing flow regimes. The index was calculated at the family level by assigning each taxa to one of 6 flow-preference groups. Each family was then placed in a second category relating to its abundance and these two values were then used to look up the flow score in an appropriate table. The final index was calculated as the ratio between the sum of the individual taxon flow scores for the whole sample, and the number of taxa used to calculate the sum. Higher flows should result in higher LIFE scores. The Average Score per Taxon (ASPT) (Armitage et al., 1983), the Extended Biotic Index (EBI) (Woodiwiss, 1978; Ghetti, 1997) and multimetric STAR_ICM index were also calculate although these indices were not designed to detect the effects of discharge reduction. The STAR_ICM index, i.e. the new official Italian method for classification based on macroinvertebrate communities, is a multi-metric index based on six different metrics (ASPT index, Log_{10}(sel_EPTD+1), 1-GOLD, number of families of EPT, total number of families and Shannon-Weiner diversity index). The identification level required for calculation is family. After normalization by the median value of reference sites' samples, these metrics are combined into the STAR_ICM index. Metric and index calculations were run with ICMeasy software, version 1.2 (Buffagni and Belfiore, 2007). Faunal similarities among the stations sampled were investigated using the Bray-Curtis similarity index. BC similarity was calculated between the macroinvertebrate assemblage of each flow-altered or reference site and comparison derived from all the reference sites (in the case of comparison for exposure sites) and all other reference sites (in the case of comparisons for reference sites). The comparison data was obtained as the average density of each taxon at the matched reference sites. Biological data were analyzed by multivariate analyses after average densities by taxon and station had been transformed to the square root to reduce the importance of abundant taxa. First, overall comparisons of macroinvertebrate assemblages at exposure and reference sites were made with two-dimensional non-metric multidimensional scaling (NMS) based on the Bray-Curtis similarity index of taxa composition between stations (Clarke and Green, 1988) in the Primer statistical package (Clarke and Gorley, 2006). The ANOSIM procedure (Analysis of Similarities, Clarke, 1993) was used to test for significant overall differences in macroinvertebrate assemblages between the free-flowing and flow-altered sites. Species contributing mostly to the dissimilarity among station groups were investigated using the SIMPER percentages procedure (Clarke, 1993). Lastly, a PCA was run to summarize the information contained in the biological data set and to identify main axis of biological variation. In order to reduce the total number of biological variables, only taxa present in the dataset with a frequency ≥10 (i.e. present in more than 10 station) were included (49 taxa out of a total of 80), as rare species were not considered necessary in defining the main ecological gradients (Cao et al., 2001). Spearman's rho was used to identify PCA axes related to hydrological alteration class. Ordination analysis was run by means of the XLSTAT version 7.5.3 Copyright Addinsoft 1995-2005 software. A Mann-Whitney U test was used to determine differences in median community invertebrate metrics calculated for reference and flow-altered sites. The response of macroinvertebrate community to the degree of flow alteration was explored using linear regression (General Linear Model) and Spearman's rho correlation.

3.4 Results

3.4.1 Overall environmental characteristics

All of the selected sites were situated on siliceous dominated geology, with altitude ranging from 300 m to 2,300 m a.s.l. (Table 12). Most of the streams had very small (<25 km^2) or small (25-150 km^2) catchment area and low stream order. The average length of the streams (distance from the source) was 10 km. The degree of glaciations varies depending on location from 0% to 10%, but streams without glacial influence were more numerous. Conductivity ranged from 19.6 $\mu s\ cm^{-1}$ (Masino and Mello) to 565.8 $\mu s\ cm^{-1}$ (Liro), while pH from 6.5 (Loga) to 8.4 (Madrasco) (Table 12).

Table 12. Site list with sampling seasons and main environmental parameters.

Stream	Code	Sampling season: Spring	summer	Autumn	Winter	Distance (km from source)	Altitude (m)	Slope (%)	Catchment area (km^2)	Glaciaton (%)	Mean flow ($m^3 s^{-1}$)	H_s	Conductivity ($\mu S\ cm^{-1}$)	pH
Avero	AVE	x	x	x	X	5.1	800	15.9	9.2	0.0	0.05	3	97.3	7.5
Belviso	BEL_1	x		x		9.5	1210	10.0	33.5	0.0	0.08	3	64.3	7.5
Belviso	BEL_2	x		x		7.2	1000	7.1	44.3	0.0	0.16	3	70.0	7.4
Belviso	BEL_3		x	x		13.3	390	12.3	58.8	0.0	0.19	4	78.3	7.4
Boggia	BOG	x	x	x		6.2	1050	8.4	15.6	0.0	0.15	3	25.1	7.3
Frodolfo	FRO_1	x	x	x		19.6	1750	2.7	69.5	9.5	0.81	2	217.9	7.9
Frodolfo	FRO_2	x	x	x	X	9.1	1350	2.6	212.0	9.4	1.90	2	242.8	8.3
Liro	LIR_1	x	x	x	X	8.5	1300	7.7	40.0	2.3	0.89	1	79.7	7.3
Liro	LIR_2	x	x	x	X	11.2	1150	7.4	59.4	1.5	0.37	3	565.8	7.7
Liro	LIR_3	x	x	x	X	14.8	1080	3.5	130.0	0.7	0.89	3	401.9	7.5
Liro	LIR_4	x	x	x	X	16.5	940	10.0	135.4	0.7	0.58	4	400.4	7.8
Liro	LIR_5	x	x	x	X	24.5	300	5.1	193.4	0.5	1.73	3	217.6	7.5
Loga	LOG	x	x			3.7	1910	6.0	9.5	0.7	0.46	0	41.3	6.5
Madrasco	MAD				X	10.8	350	18.5	26.2	0.0	0.84	0	56.2	8.4
Mallero	MAL			x		5.2	1640	7.6	14.2	2.0	0.59	0	61.2	7.7

Table 12. (Continued).

Stream	Code	Sampling season: Spring	Summer	Autumn	Winter	Distance (km from source)	Altitude (m)	Slope (%)	Catchment area (km²)	Glaciaton (%)	Mean flow ($m^3 s^{-1}$)	H_s	Conductivity ($\mu S\ cm^{-1}$)	pH
Masino	MAS			x		7.3	1030	11.8	31.8	0.5	1.28	1	19.6	7.4
Mello	MEL			x		7.2	1040	3.4	37.5	2.2	1.51	0	20.1	6.9
Mera	MER_1	x	x	x	x	24.0	580	2.8	191	-	0.69	4	98.2	7.7
Mera	MER_2	x	x	x	x	28.6	380	2.9	237	-	2.12	2	90.1	7.4
Roasco	ROA_2	x	x	x	x	15.0	600	4.0	146.5	0.0	0.58	3	136.7	8.0
Roasco Occ.	ROA_1	x	x	x	x	11.2	1300	6.9	60.6	0.0	2.16	0	95.4	7.8
Scalcoggia	SCA		x			2.8	1670	8.1	6.2	0.0	0.32	0	79.6	7.1
Soè	SOE_1	x	x			4.0	1050	11.2	6.3	0.0	0.33	0	26.3	7.3
Soè	SOE_2	x	x	x		4.4	1050	6.4	6.3	0.0	0.07	3	25.9	7.1
T. dell'Alpe	ALP			x		5.4	2300	8.5	8.0	1.3	0.22	0	218.0	8.0
Tartano	TAR			x		3.0	1500	10.2	5.8	0.0	0.19	0	41.4	7.6
Verva	VER		x			4.5	2100	7.1	10.8	1.5	0.37	0	36.0	6.8
Viola	VIO_2	x	x	x	x	15.0	1400	10.9	75.0	0.74	1.22	1	109.4	7.8
Viola Bormina	VIO_1		x			5.2	1980	9.0	32.2	4.5	1.07	0	72.2	6.6
Viola Dosdè	DOS		x			3.5	2120	3.8	17.2	8.5	0.55	0	75.5	7.0

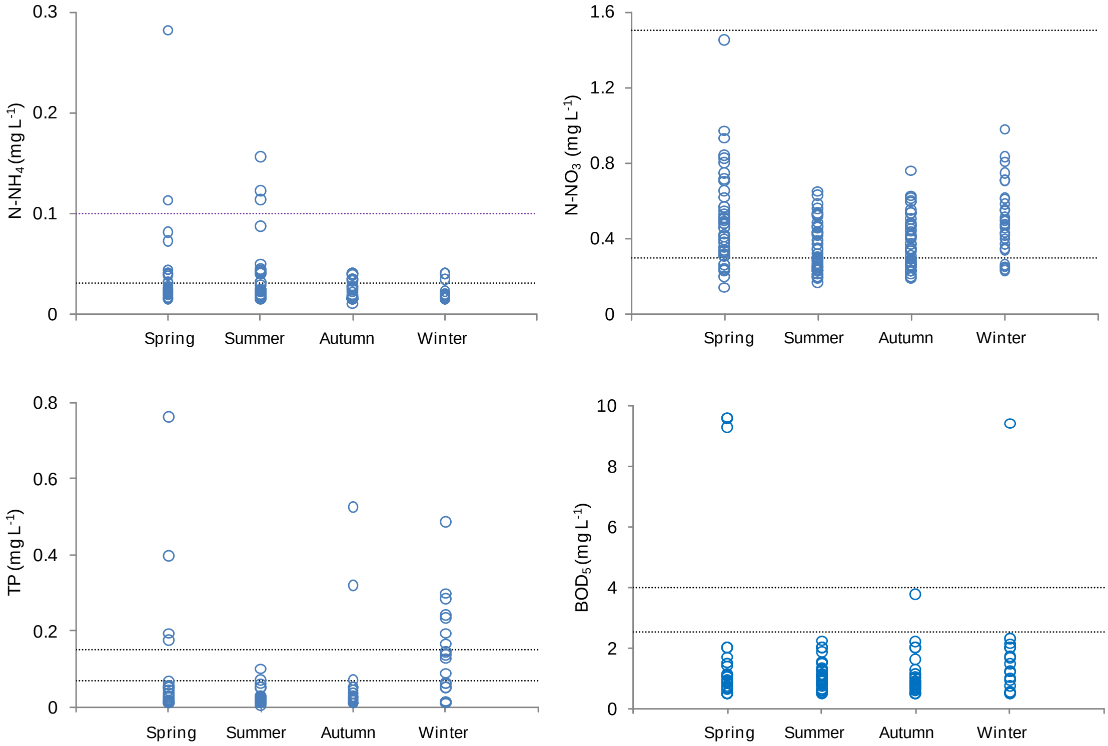


Figure 28. Distribution of main chemical parameters values (n=171) per sampling season. Dashed lines represent threshold values of the first and second LIM concentration level.

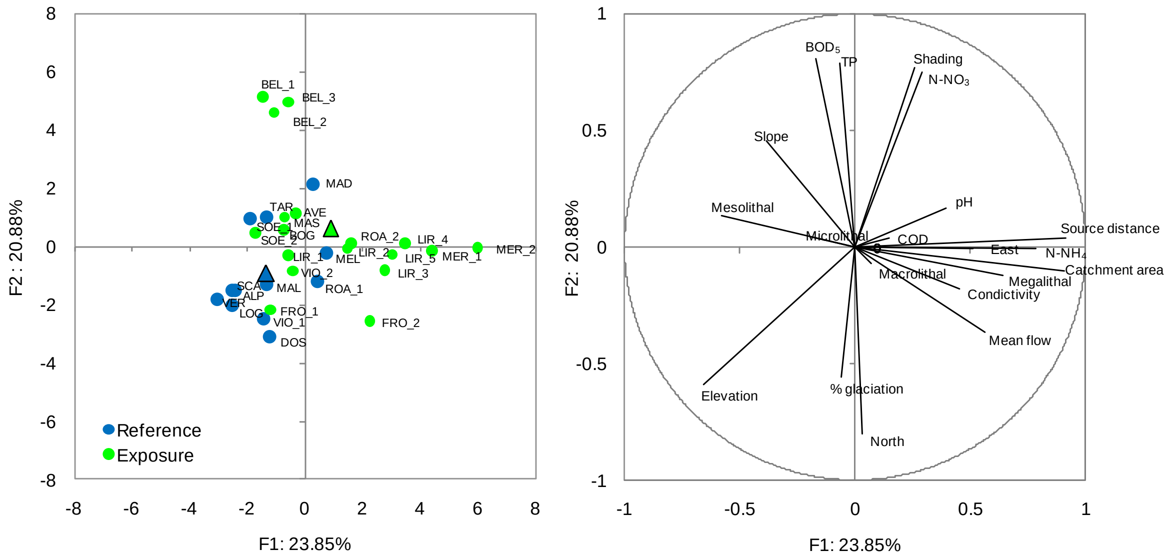


Figure 29. Ordination from a principal component analysis (PCA) of selected environmental variables and sites. In the factor map triangles represent mean for each group. The variance explained by the first and second axis is 23.85% and 20.88% respectively. In the variable ordination length and direction of each line indicates the contribution of each respective variable to axes F1 and F2.

Due to the absence of organic pollution, most of the chemical parameters had low concentration values, below the threshold of the first LIM level (81,5%). $N\text{-}NO_3$ concentrations were higher respect to the other parameters, being 65,5% of measured values between the threshold of the first and second concentration level. On the whole, the chemical quality of study streams was good or very good (Figure 28).

Twelve of 30 sites were characterized by a natural flow regime (i.e. located upstream any water diversion) and were classified as reference sites (Hs=0). Three sites were classified in the first class (mean annual flow between 35% and 100% of Qn), three sites in the second class (mean annual flow between 20 and 35% of Qn), nine sites in the third (mean annual flow between 10 and 20% of Qn) and three sites in the fourth class (mean annual flow ≤10% of Qn). Figure 30 presents the flow values measured during 2009 at the altered-flow sites (with the exception of MAS, BEL_1, BEL_2 and BEL_3 stations). The natural annual hydrograph is dominated by a peak discharge that generally occurs in late June. Distance of measurement sections from upstream diversions influenced the degree of flow alteration in the study reaches, and measured discharges were often higher than MVF released from upstream intakes. Sites classified into first or second class of hydrological stress, for instance, maintained some aspects of all components of the natural flow regime (i.e. peak discharge), thanks to the contribution of residual watershed discharge. On the contrary, sites closer to water diversions were characterized by a rather constant flow and low seasonal or monthly variability. Reference and non-reference sites were geographically interspersed, however the reference sites tended to have smaller drainage areas and higher altitude.

In the PCA ordination of study streams based on normalized environmental variables (Figure 29), reference and non-reference sites partially overlap. However reference sites had on average lower scores on both the first (mainly related to catchment area, source distance and $N\text{-}NH_4$) and the second axis (mainly related to TP, BOD_5, $N\text{-}NO_3$ and shading).

According to Fluvial Functional Index (FFI), most of the sampling sites were classified as good (II class) or fair (III) (Table 13). Although the natural condition of the catchment and the absence of morphological alterations of the fluvial habitats, no reference sites was classified as excellent by FFI. The overall judgment was penalized by the lack of a well-developed perifluvial vegetation, especially for those sites localized in the alpine vegetation zone. Four non-reference sites were classified in poor or fair-poor status (at least one bank). The presence of a dam upstream (alteration of water flow regime), morphological alterations (i.e. artificial river banks) or/and a narrow perifluvial vegetation zone were the main factors that contributed to lower the final judgment.

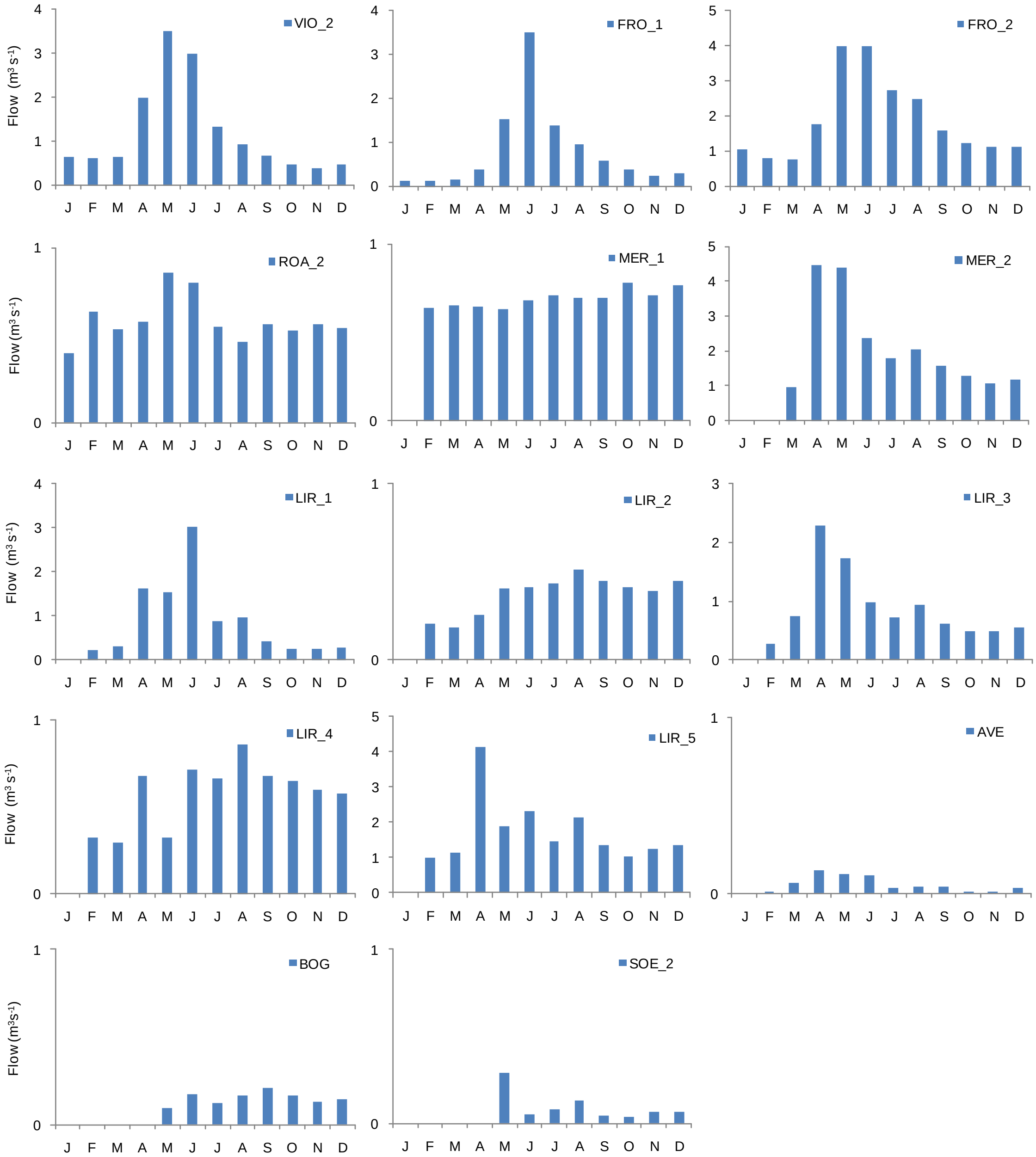


Figure 30. Measured discharged at altered-flow sites during 2009.

Table 13. Fluvial Functional Index (FFI) of study sites.

	IFF			IFF	
Stream Code	Dx	Sx	Stream Code	Dx	Sx
ALP	III	III	VIO_2	-	-
DOS	II-III	II-III	ROA_2	-	-
VER	II	II	LIR_1	II	II-III
VIO_1	II	II	LIR_2	III	II
LOG	II-III	II-III	LIR_3	III	III-IV
SCA	II	II	LIR_4	III	II
MAL	II	II	LIR_5	IV	III
TAR	II	II	MER_1	IV	IV
ROA_1	II	II	MER_2	III	II-III
SOE_1	II	II	AVE	II	II-III
MEL	I-II	I-II	SOE_2	II	II
MAS	I	I	BOG	III	III-IV
MAD	II-III	II	BEL_1	I	I
FRO_1	-	-	BEL_2	I	I
FRO_2	-	-	BEL_3	II	II

3.4.2 Invertebrate comparisons

Eighty taxa of macroinvertebrates were collected from the 30 study sites of which 35 were EPT and 16 were Diptera. The most common taxa were ephemeropterans *Baetis* spp. *Ecdyonurus* spp. and *Rhithrogena* spp., and diptera Chironomidae and Limoniidae. Density estimates of the recorded taxa are presented in Appendix C. Values of Shannon–Wiener diversity (H), Margalef's richness (D_{Mg}), eveness (J), Simpson's Dominance (C), taxa richness (T), number of EPT taxa (EPT), ASPT score, density, STAR-ICM, EBI and LIFE indexes per site are summarized in Appendix E.

Mean abundances per site (individuals m^{-2}) differed considerably among sites, ranging from one hundred to 4,000 ind. m^{-2}. The highest abundance of just over 12,000 individuals m^{-2} occurred at ROA_1 site on the Roasco di Sacco stream during March. Lowest values were recorded in upland streams, particularly VIO_1 and DOS sites. The distribution of abundance per site is shown in Figure 31.

Information on taxa richness, number of EPT taxa, ASPT score, EBI and STAR-ICM per site are presented in Figure 32 and Figure 33. All sites, including both reference and impacted sites (sites which experience water diversion), were on average in good or high ecological status according to the STAR_ICM index.

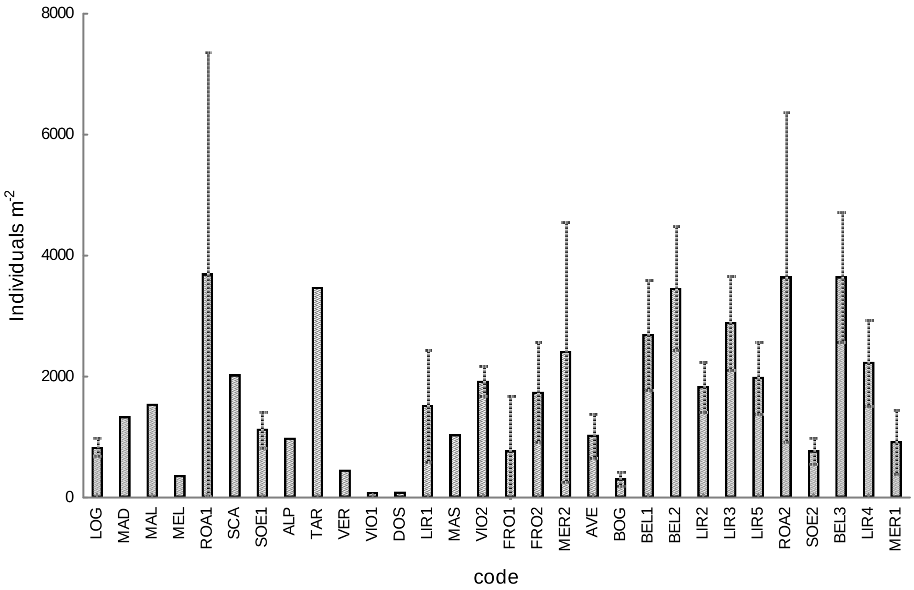


Figure 31. The range of abundance values in the study sites ordered by increasing Hs class. For sites where more than one sample was available, bars represent the mean macroinvertebrate density (± 1SD).

Scores and numbers of taxa recorded vary widely among sites primarily with stream size and location rather than alteration of flow regime. For example, considering the reference sites, T and EPT ranged respectively from 9 to 33 and from 5 to 21. Moreover sites draining alpine zone were characterized by lower STAR-ICM values (mean = 0.82, good ecological status) respect to those located below the tree line (mean = 1.05, high ecological status). Since all impacted sites were below the tree line, the comparisons between mean values of metrics for reference and impacted sites were performed considering two different subsets, the first including all the reference sites, the second excluding those sites at higher altitude (ALP, LOG, DOS, VIO_1, VER).

The Bray-Curtis (BC) similarity for pairs of reference sites was moderately low, averaging 44.63 (SD = 12.88), confirming that the collected macroinvertebrate assemblages differed substantially among reference sites. When the fauna of each site was contrasted with comparison data derived from all reference sites, excluding self-comparison for reference sites, case similarity between a site's macroinvertebrate and its comparison data was on average not significantly different between comparisons for exposure sites and comparison for reference sites, with either high altitude sites included or not in the analysis (Mann-Whitney U test, $p>0.05$) (Figure 34).

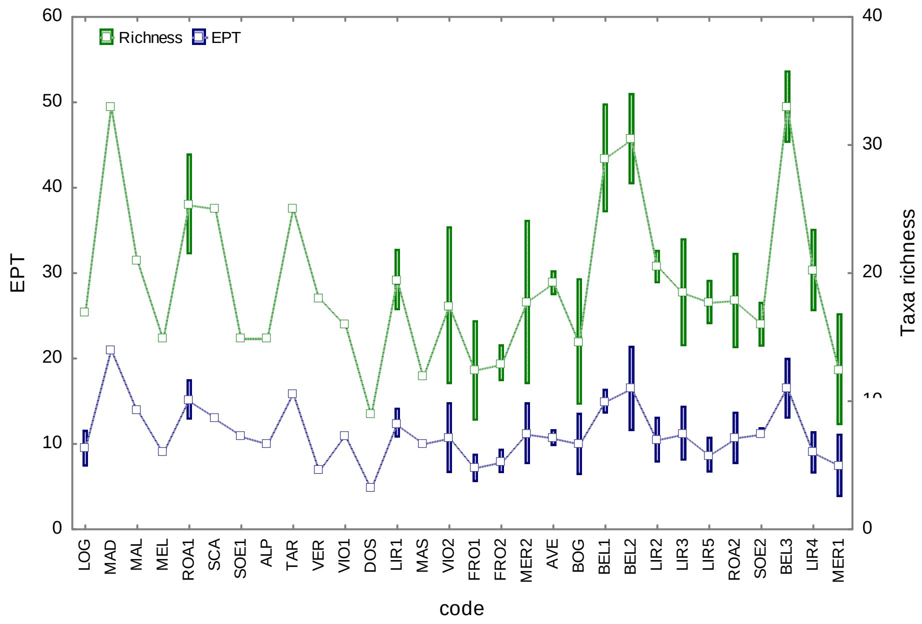


Figure 32. The range of taxa richness and EPT number in the study sites ordered by increasing Hs class. For sites where more than one sample was available, bars represent the mean value (± 1SD).

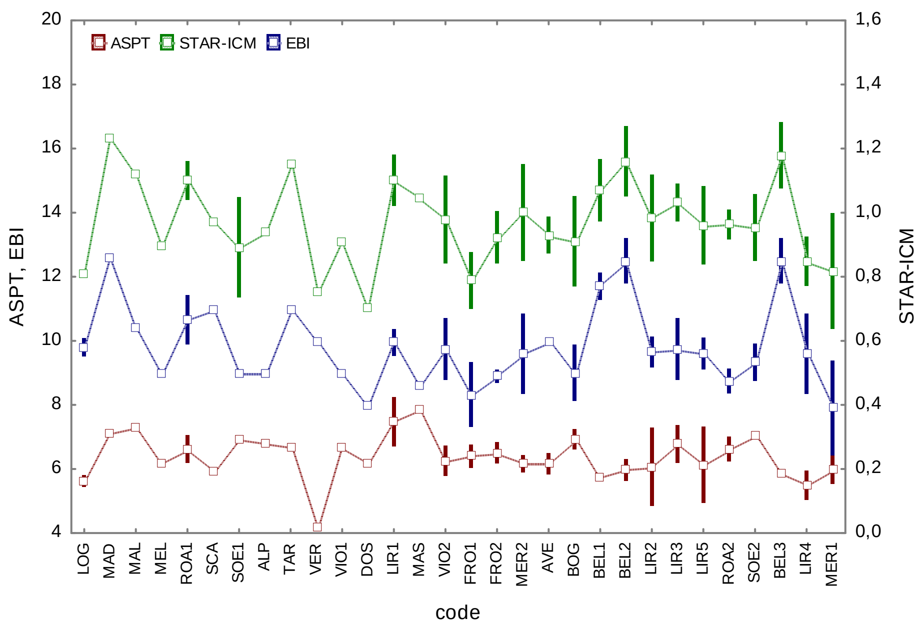


Figure 33. The range of STAR-ICM, ASPT and EBI values in the study sites ordered by increasing Hs class. For sites where more than one sample was available, bars represent the mean value (± 1SD).

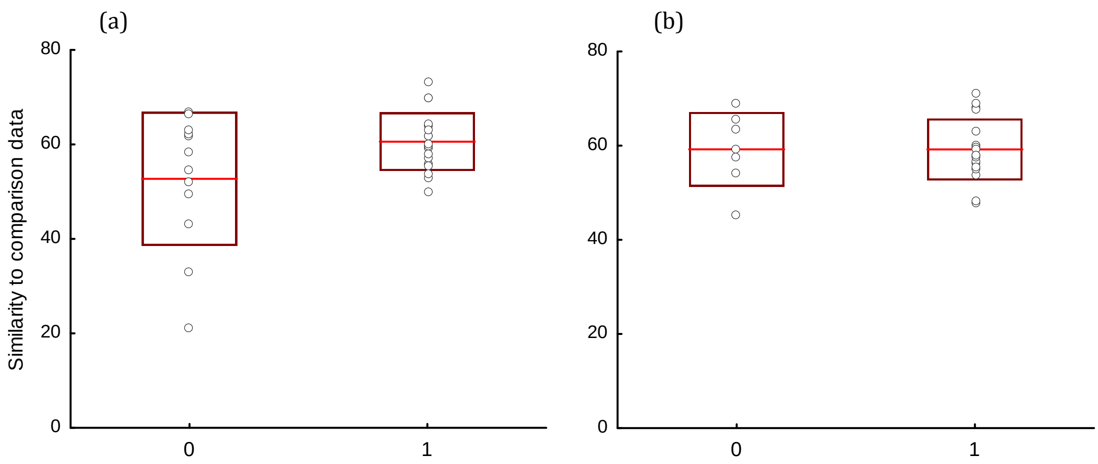


Figure 34. Mean (±SD) and distribution of Bray-Curtis similarity of macroinvertebrate data from individual reference sites (0) and impacted sites (1) to comparison data from reference sites: (a) whole data set; (b) exclusion of high-altitude reference sites.

Average (±SD) invertebrate density, T, EPT, H, C, LIFE, ASPT, EBI and STAR_ICM indices calculated for reference and impacted sites are presented in Table 14. Figure 35 shows distribution of values within Hs classes. Coefficient of determination (r^2) of GLM regression and Spearman's rho correlation between macroinvertebrate metrics and Hs class are shown in Table 15.

Table 14. Mean (±SD) density, number of Ephemeroptera Plecoptera Tricopthera (EPT) taxa, richness, Shannon diversity, Simpson's Dominance, LIFE, ASPT, EBI and STAR_ICM index at free-flowing and impacted sites.

Metric	Reference (n=12)	Reference (n=7)	Exposure (n=18)
Density	1339.43 (±1204.02)	1944.74 (± 1232.01)	1890.43 (± 966.72)
T	19.53 (± 6.55)	22.77 (± 6.40)	18.89 (± 6.16)
EPT	11.81 (± 4.34)	14.17 (± 3.86)	10.91 (± 2.74)
H	2.09 (± 0.41)	2.12 (± 0.39)	1.76 (± 0.30)
C	0.20 (± 0.11)	0.21 (± 0.12)	0.28 (± 0.08)
EBI	9.96 (± 1.26)	10.52 (± 1.25)	9.74 (± 1.29)
LIFE	8.30 (± 0.52)	8.24 (± 0.17)	8.32 (± 0.27)
ASPT	6.36 (± 0.85)	6.69 (± 0.49)	6.42 (± 0.62)
STAR-ICM	0.96 (± 0.16)	1.05 (± 0.13)	0.98 (± 0.11)

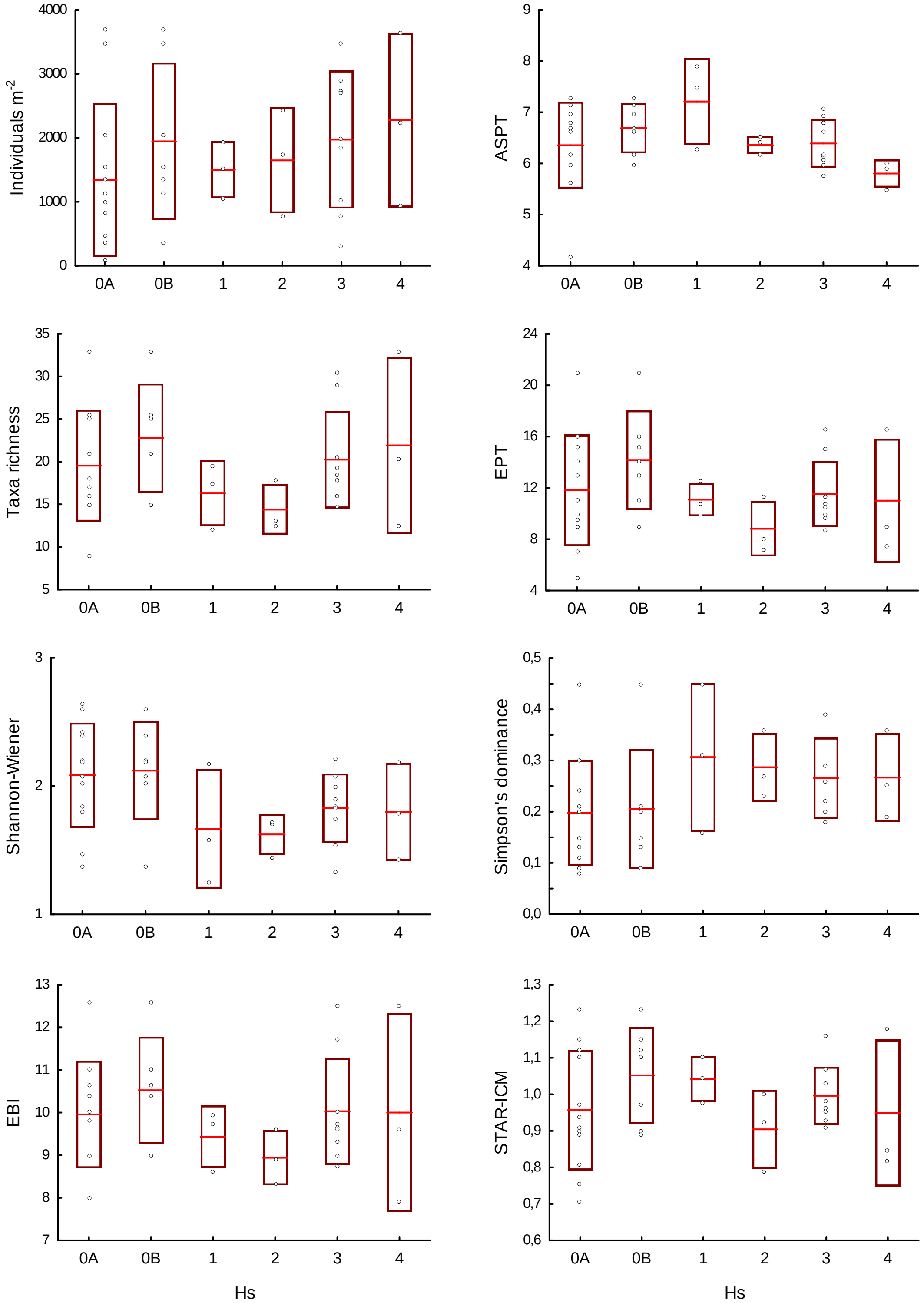


Figure 35. Mean (±SD) and distribution of density, ASPT, T, EPT, H, C, EBI and STAR-ICM index within Hs classes. 0A = all reference sites; 0B= exclusion of high altitude (>1800m a.s.l) reference sites.

Table 15. Spearman's rho and R^2 values calculated using GLM method for (A) the whole data set and (B) with the exclusion of free-flowing sites located at an altitude greater than 1800 m a.s.l.

Metric	A (n=30)		B (n=25)	
	Spearman's rho	GLM r^2 value	Spearman's rho	GLM r^2 value
Single metrics				
Density	0.30	0.10	0.08	0.05
T	0.07	0.11	-0.06	0.21
EPT	-0.06	0.07	-0.29	0.26
H	-0.29	0.22	-0.25	0.26
C	0.34	0.17	0.25	0.13
EBI	-0.08	0.08	-0.17	0.15
LIFE	-0.04	0.05	-0.03	0.16
ASPT	-0.29	0.21	**-0.52***	**0.41***
BC similarity	0.17	0.15	-0.10	0.03
Multimetrics				
STAR-ICM Her03	0.08	0.08	-0.24	0.18
Multivariate				
PCA	**-0.63***	**0.44***	**-0.74***	**0.58***

*p < 0.05

Considering the whole set of data collected at the reference sites only Shannon diversity and Simpson's Dominance indices differed significantly (Mann-Whitney U test, p<0.05) between reference and exposure sites (Table 14). If high-altitude reference sites were excluded, significant difference were found between the number of EPT taxa and the Shannon diversity index (Mann-Whitney U test, p<0.05). ASPT score was the only metric significantly related (GLM, p<0.05) to the degree of hydrological alteration (Table 15), but only with the exclusion of high-altitude reference sites.

One-way ANOSIM found statistically significant overall differences in assemblages composition between free-flowing and flow-altered sites (global R=0.273, p=0.002) and, although a large within-group variability, segregation of the two types was quite apparent in two-dimensional NMS ordinations (Figure 36). Average dissimilarity between groups based on Bray-Curtis index was 54.28 (SIMPER analysis). Individual taxa that contributed more to the observed dissimilarity were *Baetis* (9.9%), Chironomidae (7.69%), *Leuctra* (5.73%), Simuliidae (4.54%) Limnephilidae (4.43%), being these more abundant at exposure sites.

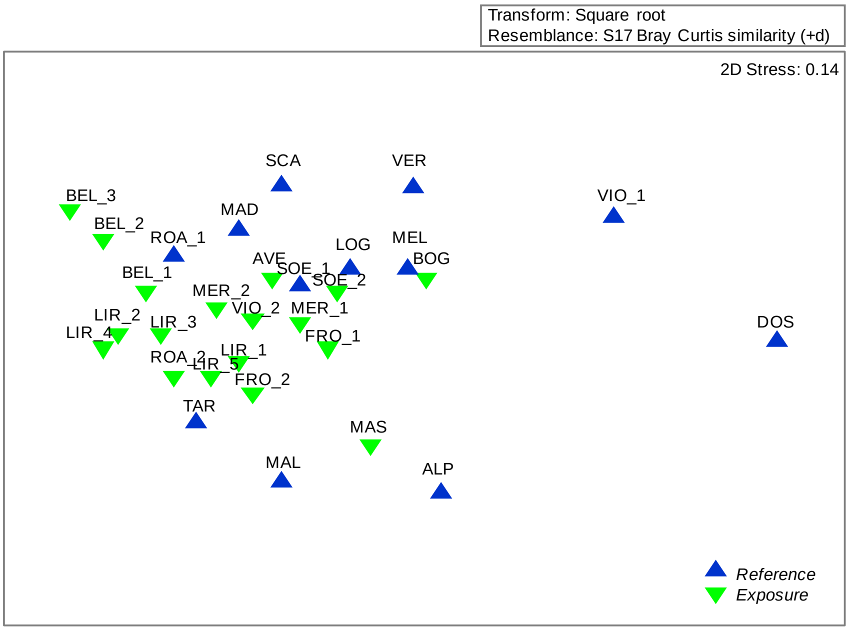


Figure 36. NMS of sampling sites based on Bray-Curtis similarity.

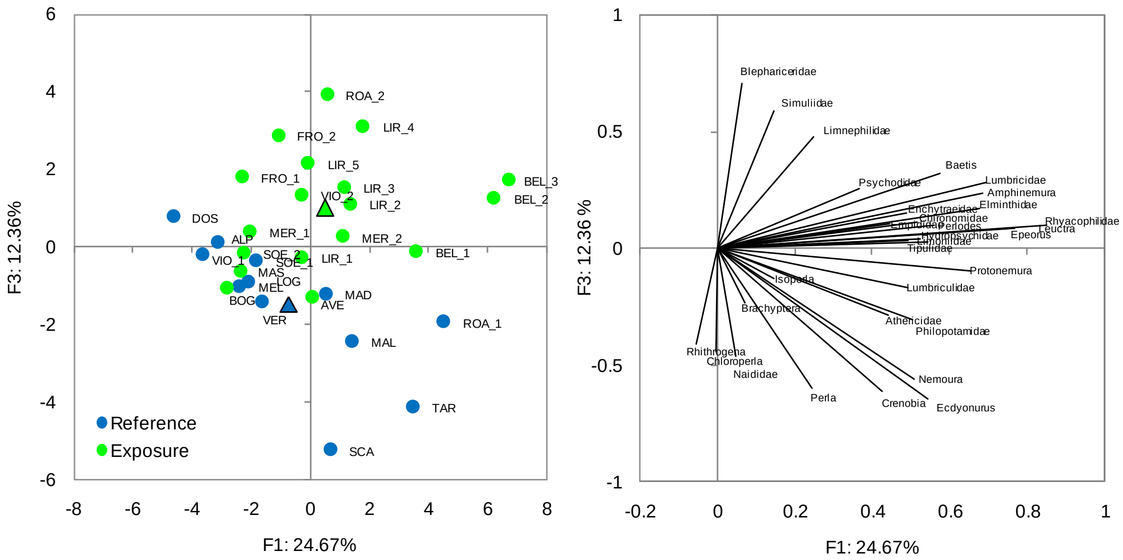


Figure 37. Ordination from a Principal Component Analysis (PCA) of invertebrate abundances at study sites. In the factor map, triangles represent the mean for each group. In the taxa ordination length and direction of each arrow indicates the contribution of each respective taxon to axes F1 and F2.

Principal Component Analysis (PCA) based on invertebrate abundance showed that, plotting the observations on the first (24.7% of total variance) and third axis (12.4% of total variance) of variation, the reference and impacted sites clustered in separate groups, implying that the invertebrate faunas of the reference and flow-altered sites differed (Figure 37). Spearman's rho correlation confirmed that the variation in the community composition along PCA axis 3 (12.4% of total variance) could be explained by hydrological alteration class (ρ= -0.63, p<0.05). The main taxa determing axis F3 were Blephariceridae (eigenvector = 0,36), *Ecdyonurus* (eigenvector = -0,33), *Crenobia* (eigenvector = -0,32), *Perla* (eigenvector = -0,31) and Simuliidae (eigenvector = 0,30).

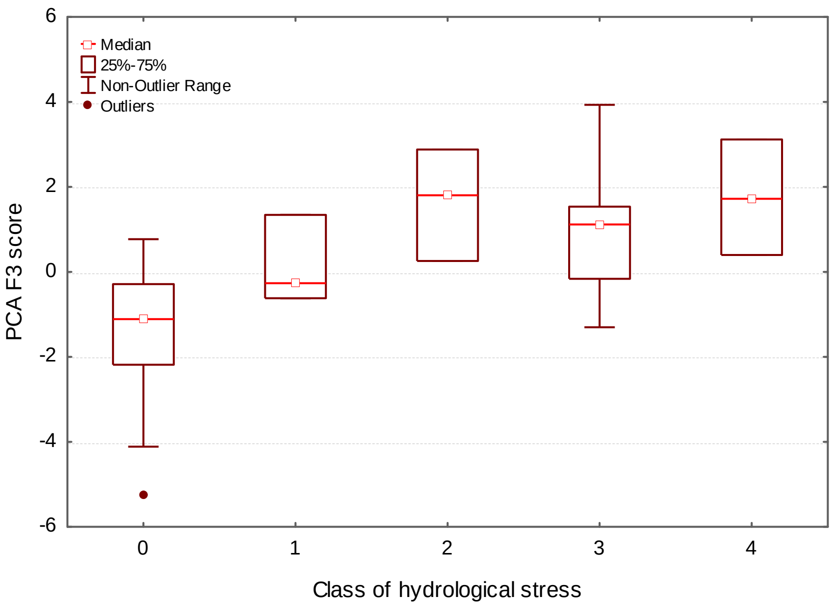


Figure 38. Distribution of F3 scores within class of hydrological alteration

The average sample score on F3 between reference and exposure sites was significantly different (Mann-Whitney U test, p<0.001), moreover axis 3 score was significantly related to the hydrological alteration gradient (GLM, r^2 = 0.44, p<0.05) (Figure 38).

3.2 Discussion

The influence of flow reduction on macroinvertebrate communities was investigated comparing sites that were affected by a different degree of water diversion and free-flowing sites spread cross the same geographical area. This approach differs from previous studies of the impacts of abstraction on stream macroinvertebrate assemblages, most of which compared sites upstream and downstream of diversions on the same watercourse (Armitage and Petts, 1992; Kinzie et al., 2006; McIntosh et al., 2008) or experimentally diverted a portion of stream flow from some reaches while leaving others with unaltered flow (McKay and King, 2006; James et al., 2008; James and Suren, 2009). All of these approaches have certain limitations. In simple upstream-downstream comparisons the effects of abstraction can be confounded by other environmental differences between the upstream and the downstream sites, whereas manipulative experiments can apply before-after-control-impact design that permit stronger causal inference. However, experimental studies may have limited realism because the spatial scales, volumes and timing of water removal may not match those associated with actual water use (Chessman et al., 2009). Moreover the level of replication and observed variability may strongly limit the inference capability of the experimental study (McKay and King, 2006). The major limitation of the approach used in this study was probably the ability to find sufficient sites similar for environmental characteristics to impacted sites, but without substantial upstream abstraction. Differences in environmental factors may be more important in determining macroinvertebrate assemblages at this scale of analysis, hiding the potential negative effects of water diversion. In our study, the location of reference sites was constrained by the extensive use of water resources for hydroelectric production in the study area. Accordingly, it was not possible to find more streams to be used as reference site at lower elevations or with greater drainages areas.

Nevertheless, this study provided evidence that water diversion was impacting on aquatic macroinvertebrate assemblages causing a moderate but significant reduction of Shannon-Wiener diversity (23%) and, if reference sites draining alpine zone (>1,800m a.s.l.) were excluded from the analysis, of EPT number (17%). Diversity of macroinvertebrate species decreases with increasing altitude because of the shortest timespan since deglaciation, the limited food supply and the extreme physical conditions (Landolt and Sartori, 2001; Füreder, 2007). The lower macroinvertebrate diversity found in high altitude sites was expected for these reasons.

Low or reduce flows in permanent streams often cause decreases in taxonomic richness. Changes to instream habitat environments, such as loss of habitat types, increased water temperature, increased sedimentation and altered periphyton assemblages, might cause changes in taxonomic richness as flow decreases (Dewson et al., 2007b). McIntosh et al. (2002) compared macroinvertebrate communities above and below a diversion in a Hawaiian stream. Taxa preferring fast-flowing cascade habitats were lost downstream of the diversion because their habitats disappeared when flow downstream of the diversion was reduced by 92 to 97% during summer. The number of taxa in benthic invertebrate communities in a large river with greatly reduced residual flow was lower than in similar unregulated rivers because braided channel habitats were lost in the regulated river (Cazaubon and Giudicelli, 1999). Wood and Armitage (1999) suggested that richness in a small gravel-bed stream was lower than in other English lowland streams because sediments gradually covered gravel surfaces as flow declined. Species richness in the same gravel-bed stream generally increased with increased flow during a 6-y period (Wood et al., 2000). Englund and Malmqvist (1996) concluded that species richness was lower in sites with reduced flow than in sites with normal flow because food supply for grazing invertebrates was lower at the sites with reduced flow.

Biotic interactions may provide valuable information on the potential mechanisms underlying patterns of community level responses to low flow. Streams diversions eliminate or reduce the typical frequency and magnitude of spates, and as environmental stability increases, the importance of biotic interactions may also change (Menge and Sutherland, 1987; Peckarsky et al., 1990; Death and Winterbourn, 1995). In addition, a reduce flow can alter inter- and intra-specific interactions within the ecosystem by decreasing water velocity and habitat size (Dewson et al. 2007b). Consequently, changes in community structure due to alteration of direction and strength of predator-prey and competitive interactions, can occur over time (Extence, 1981; Peckarsky et al., 1990). For example Zhang et al. (1998) found that species richness and abundance of simuliid larvae were greater than predicted at low-flow sites and attributed this result to lower densities of predators and less competition at low-flow sites than at high-flow sites.

The lack of a more consistent impact of diversion on study streams appear to contrast with many previous study that have inferred substantial negative effects of abstraction on assemblages of stream macroinvertebrate. However most such studies have investigated abstractions that removed the entire flow of a stream or a very large proportion of it (Dudgeon, 1992; Petts and Bickerton, 1994; Kinzie et al., 2006). Channel morphology and habitat diversity can strongly influence invertebrate community response to decreases in flow. For example water abstraction generally had less effect on the fauna of upland streams than on the fauna of small lowland streams in the UK (Armitage and Petts, 1992; Castella et al., 1995). In upland streams within- and between-season flow variability is naturally high as is substratum heterogeneity. The resident fauna is therefore adapted, within limits, to extreme conditions which may involve severe loss of wetted area. The wide range of water velocity, which is related in part to the substrate diversity, may offers a variety of microhabitats capable of supporting the benthic fauna albeit in a reduce area of the channel. In addition abstraction effects can be ameliorated by inputs from small tributaries which abound in upland drainage basin and may facilitated rapid, and repeated, re-colonization of affected reaches (Armitage and Petts, 1992). These environmental factors may have contributed to the mitigation of the negative impacts of water diversion on macroinvertebrate communities in the present study too. Even under low-flow conditions, moderate flow velocities may be maintained within steep slopes mountain streams, ensuring the availability of a different range of microhabitats to sustain fauna. Moreover the high drainage density may allow a rapid recovery of some aspects of the natural flow regime, such as seasonal variability of flow rates.

The severity of the flow reduction may influence invertebrate response because it affects the amount of habitat lost and the magnitude of change in instream condition. With smaller proportion of flow diverted impacts on macroinvertebrate should differ from those observed under more extreme abstraction. For example Chessman et al. (2009) attributed the lack of evident impact of water abstraction on macroinvertebrate assemblages in 85 sites, 54 of which designated as reference sites, spread in the Australian state of New South Wales, mainly to the low proportion of stream flow diverted (1-20%). Rader and Belish (1999) found that diversions can have variable effects on benthic communities in high elevation headwater streams of the Rocky Montains, USA, depending on the degree of alteration to the natural flow regime. Invertebrates appeared resilient to mild diversion where some aspect of all components of the natural flow regime were transported downstream, conversely only in the more heavily diverted watercourses (almost 100% of flow diverted) invertebrate abundance and diversity declined of 50 and 16% respectively. Wills et al. (2006) compared macroinvertebrate fauna between and upstream control reach and a treatment reach downstream of a side channel that experimentally diverted between 0 and 90% of flow from a Michigan stream over several years. At 90% reduction, the density of macroinvertebrate was significantly lower in the treatment reach than in the control reach but at 50% reduction the total density and densities of some components of the fauna were significantly higher in the treatment reach. Conversely, Dewson et al. (2003) recorded higher macroinvertebrate densities downstream than upstream of consented abstractions of 43-98% of mean annual low flow from three New Zealand streams, but similar upstream and downstream densities in a stream where abstraction was only 28%. In a series of studies of 22 rivers in the United Kingdom with various levels of abstraction, impacts were varied and often undetectable. Armitage and Petts (1992) found significant upstream–downstream differences in macroinvertebrates abundances (catch per unit effort) for only six of this rivers, in biotic index values for only three and in taxonomic richness for only one. The most obvious impact of abstraction was at site where groundwater withdrawal had resulted in drastic dewatering of the channel. In further upstream-downstream comparisons of macroinvertebrate assemblages on all 22 rivers using ordination methods, Castella et al. (1995) failed to find any coherent patterns in relation to the type or entity of abstraction. These authors attributed clearly defined differences to abstraction only in four cases, characterized by site-specific and threshold hydraulic conditions (i.e. mean velocity less than 0.05 m s^{-1}, mean depth less than 10 cm, reduction in macrophyte cover, increase proportion of fine sediments). On the whole, these studies demonstrated that minor abstractions generally had less effects on biota than major abstractions. However they failed to find relationship between degree of alteration and impacts to stream biota.

In this study, ASPT score was the only biological metric significantly related to the amount of water diverted. Other author have previously demonstrated that common metrics used to detect organic pollution can be responsive to changes in flow conditions, since most organic-pollution sensitive species generally show a preference for cooler, more turbulent water (Extence and Ferguson, 1989; Monk et al., 2006; Bona et al., 2008; Buffagni et al., 2009). LIFE scores, specifically designed to reflect faunal responses to altered flow regime in the U.K., didn't respond to the gradient of flow reduction. This result, possibly cause by the different geographical setting, is in disagreement with previous studies that reported LIFE scores as a good indicators for flow condition in comparison with other widely used indices (Extence et al., 1999; Monk et al., 2006, 2008; Dunbar et al., 2010). These findings support the idea that biological metrics based on invertebrate community and traditionally used in bioassessement programs, may be poor descriptors of the magnitude of flow reduction in Alpine streams, unless extreme low-flow conditions are reached.

Finally, a consistent pattern in the dataset was demonstrated through multivariate analysis by groups of taxa that were respectively reduced (*Perla, Ecdyonurus, Crenobia*) or increased (*Baetis*, Chironomidae, Blephariceridae, Simuliidae, Limnephilidae) in abundance at the impaired sites. These data suggest that diversions are potentially responsible for a shift in macroinvertebrate composition.

CHAPTER 4

Conclusions

In this dissertation, I have examined the negative effects induce on stream ecosystem by hydropower schemes in an Alpine area. The research included the study of the ecological consequences of yearly sediment flushing and of reduction of discharge downstream of diversions. In this final chapter I draw conclusions from these evaluations and the research as a whole. Moreover implications for future management and directions for future studies in this area are presented.

Reservoir flushing

The flushing of large volumes of sediment from reservoirs cannot be considered to have a negligible impact on the stream ecosystem due to the occurrence of unpredictable short-lasting SSC peaks and the presence of high SSC throughout the operation period. A periodic flushing of the reservoir is therefore recommended both to improve the SSC control downstream of the dam and to maintain storage capacity in the long term. Within these limits, out-flowing sediment loads can be effectively regulated by hydraulic operations and mechanical digging. Raw turbidimeter data were appropriate to support real-time monitoring of flushing operations thanks to the particle size of flushed material (mainly silt) and the resulting good agreement between probe and laboratory data.

The flushing of large volumes of accumulated sediment had clear effects on the stream ecosystem. Density of brown trout population decreased drastically after the beginning of flushing operations in September 2006. Juveniles, in particular, were the most vulnerable individuals. Nonetheless, the fish population appear to reach a state of equilibrium with the new environmental conditions. Limiting factors, such as availability of space and food

during and immediately after the flushing, creates a density-dependent mechanism of survival that contribute to determine population size by reducing environmental carrying capacity (bottleneck). It should be underline that current density values are comparable to those reported for analogous Alpine situation under natural condition. However, habitat fragmentation strongly limits resilience of the population and consequently sustainability of yearly flushing operations. The lack of connectivity with downstream sector prevents natural migrations movements that seems necessary to counterbalance the negative effects of flushing on the resident trout populations. Possible actions to mitigate this biological consequence at vulnerable sites include: to re-establish connectivity (i.e. fishways), to ban fishing activities and to undertake restocking program, with the use of hatchery-reared fish, for maintaining good recruitment levels between flushing plans. Finally, it's important to emphasize the length of the study period, that was essential to understand the mechanisms of persistence of fish populations and the recovery processes.
The examined model for predicting fish responses to the SSC increase, was successfully validated with the sampled data, although further experimentation appears to be necessary for its reliable use in the planning phase of analogous sediment removals.
Invertebrate assemblages exhibited a sever reduction in overall abundance and biomass after each flushing. However, the resilience of the benthic communities guarantee a good recovery of density, biomass and stream quality, as measured by the normative quality indices. Quick and repeated re-colonization was possible thanks to the benthic drift from the undisturbed catchment area of the stream, where the minimum instream flow is diverted. Nevertheless, the composition of macroinvertebrate community was clearly altered by sediment removals. The yearly occurrence of sediment flushing rejuvenates the biological community, allowing species with fast life cycle and good colonizing ability to become dominant (e.g. *Baetis*).
After the flushing flows operated with clear waters, observation of the studied river sector revealed no relevant sediment deposits in the wetted channel. However, particle size analysis of core samples collected in the riffle habitat provide evidence of a significant increase in interstitial fine sediment, that could adversely affect the recovery processes. The relative importance of clogging in determining negative effects on fish and macroinvertebrate communities, in the analyzed context, is still unclear and specific studies should be done in the future to delve into this important issue.

Indications for planning future flushing activities are related to the avoidance of SSC peaks, to the maintenance of maximum SSC below 10 g L^{-1} (daily average) and 5 g L^{-1} (overall average), and to the maintenance of clear water flowing during the night. Exceptions to these indications are expected during the first hours of dam outlet opening. When monitoring SSC, particular attention should be paid to the average daily concentrations, rather than instantaneous concentrations. The overall average concentration could be used as a reference value for the planning of the duration of the flushing operation that guarantees its economic sustainability. The decreasing acute impact of flushing on fish populations, although average SSC and durations were approximately the same over the years, provide evidence that the SSC range experimented in this study was tolerated by fish population and supports these SSC thresholds. The possibility of reducing the frequency of operations (on alternate years for example) should be considered in the future plans, in accordance with the need of preservation of the storage volumes. This would lighten the stress on the downstream biota and facilitate a better recovery of biological communities.

As for the transferability of the research findings to the controlled flushing of other reservoirs, two aspects should be considered. The first, related to the indications concerning the aspects to be considered for planning the flushing and the monitoring activities, can be readily applied for small Alpine reservoir presenting similar technical infrastructures (i.e. inflow water that may by-pass the reservoir during the flushing activities, high volumes for dilution). The second aspect, i.e. the applicability of these SSC threshold to the controlled flushing of other reservoirs, is mainly challenged by differences in stream hydraulic and morphology of the receiving water course. This study has provided threshold values which minimize the ecological consequences of sediment release and which can be applied in other similar situations (small to medium high-gradient streams) in the future, but that may be not suitable in contexts different from the one where such thresholds were developed.

Minimum Flow releases

The main objective of this research was to evaluate the use of the biological component (macroinvertebrate community) for detecting the effects of hydrological alterations in Alpine stream ecosystems. This was accomplished comparing sites that were affected by a different degree of water diversion and free-flowing sites spread cross the same geographical area. Single and multimetric indices, as well multivariate approach were used for detecting differences between macroinvertebrate assemblages.

The macrobenthic community of the sites located at high altitudes, above the tree line (roughly >1,800m a.s.l.), appears constrained by extreme environmental conditions which is subject to: oligotrophic water, limited flood supply, cold temperature, high flow rates during deglaciation. For these reasons, in this zone, benthic fauna may not be a suitable indicator to detect the impacts of water diversion, since the possible relationship between minimum flow and community impairment is hidden by other abiotic factors controlling the biological processes in high elevations. The lack of an appropriate biological indicator in high altitudes streams, where environmental conditions do not allow the stable presence of fish populations and the development of a highly-diverse invertebrate community, call for the use of alternative approaches based on other factors, such as, for example, the degree of water filling of the riverbed, landscape value determined by the visual perception of surface runoff, the flow required to prevent freezing of the entire water mass or prevent partial and/or total drought of the sections downstream diversions.

On the contrary, invertebrate communities at lower elevations, being not controlled by extreme environmental conditions, may be used for the evaluation of the impacts of flow reductions. However, based on what we have measured, invertebrate-based metrics may be poor indicators of the degree of hydrological alteration. In fact, although the results suggests that reduction in stream flow by diversion had significant negative effects on macroinvertebrate diversity, only one metric was significantly related to the amount of water diverted. Resource managers should consider the potential consequences of water diversion to all components of stream communities, including benthic macroinvertebrates. However, caution should be applied when using biological metrics in Alpine streams, because most relationships between invertebrate-based metrics and magnitude of flow reduction were insignificant. In this respect, multivariate analysis seems a promising approach, since, not only allowed the quantification of the variation of community composition related to the degree of flow alteration, but also the identification of taxa particularly sensitive to alteration of flow regime. Clearly, these findings are based on short-term temporal analysis, leading to an incomplete description of invertebrate communities. Further research in this direction are required to better describe natural communities, in particular with respect to seasonal life cycles, and to validate these findings.

Overall conclusions

The information gained from this study have allowed a better understanding of the impacts on stream ecosystems of hydropower schemes in an Alpine region. Long-term research carried out at Valgrosina reservoir was essential to understand the mechanisms of persistence and recovery of biological communities and to validate sustainability of the proposed operative modalities. Similar studies should be carried out in systems with different characteristics in order to represent a solid basis to guide the planning of sediment management practices aimed at the preservation of reservoir storage.

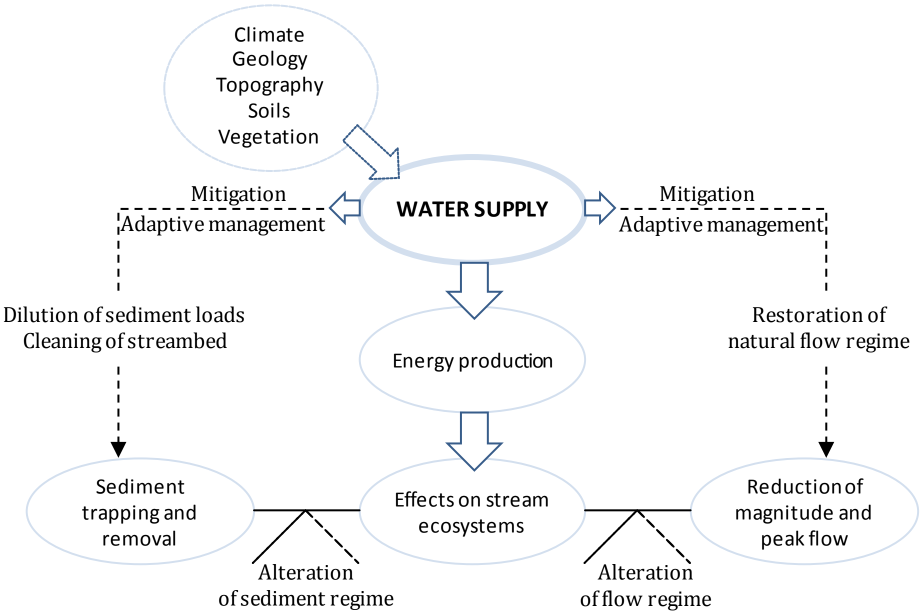


Figure 39. Conceptual framework of the study.

As for the effects of flow reduction by diversions, methods are needed to relate changing river flow to ecological response and to support environmental flow assessment and restoring actions. In particular, there is need for information applicable at regional scales, and to move away from management actions that emphasize the uniqueness of individuals rivers, and hence require extensive site specific data. Difficulties in finding a biological indicator sensitive enough to discriminate different levels of water diversion, are enhanced by the fact that biological community at altered-flow sites often did not appear excessively impaired. Relationships between flow alteration and macroinvertebrate response found in this study will have to be further developed and tested with suitable data, but certainly represent a good starting point for the development of more ecologically guidelines for the maintenance of river flows.

In summary, research findings showed how both activities have the potential to alter biological communities, but even that could be managed together to minimize risks. Mitigation actions, required to compensate for these impacts, use the same resource, i.e. the available water supply (Figure 39). Most of this volume is stored for energy production, while the remaining part is intended to mitigate the adverse effects caused by the same hydropower facilities: dilution of suspended sediment loads during flushing operations, cleaning of the riverbed, restoration of natural flow regime in its components (magnitude, frequency, duration, timing and flashiness). Currently, the release of a MVF, defined as a low flow limit maintained during the whole year, does not appear enough as mitigation measure for both sediment release operations and water withdrawal. Establishing specific criteria for flow restoration is challenging because our understanding of the interactions of individual flow components with geomorphic and ecological processes is incomplete. However, quantitative standard can in principle be developed based on the reconstruction of the natural flow regime. Restoration actions based on such guidelines should be viewed as experiments to be monitored and evaluated, that is adaptive management, to provide critical new knowledge and guide further actions. In this respect, seasonal modulation of water releases and restoration of peak flows for channel maintenance and habitats restoration (including removal of accumulated fine sediment) should be considered in the future when allocating available water supply between production and mitigation actions.

Considering obtained results this project will help to support current and future research of Alpine stream ecosystems and will provide information for a management of freshwater resource balancing, economical, technical and environmental issues.

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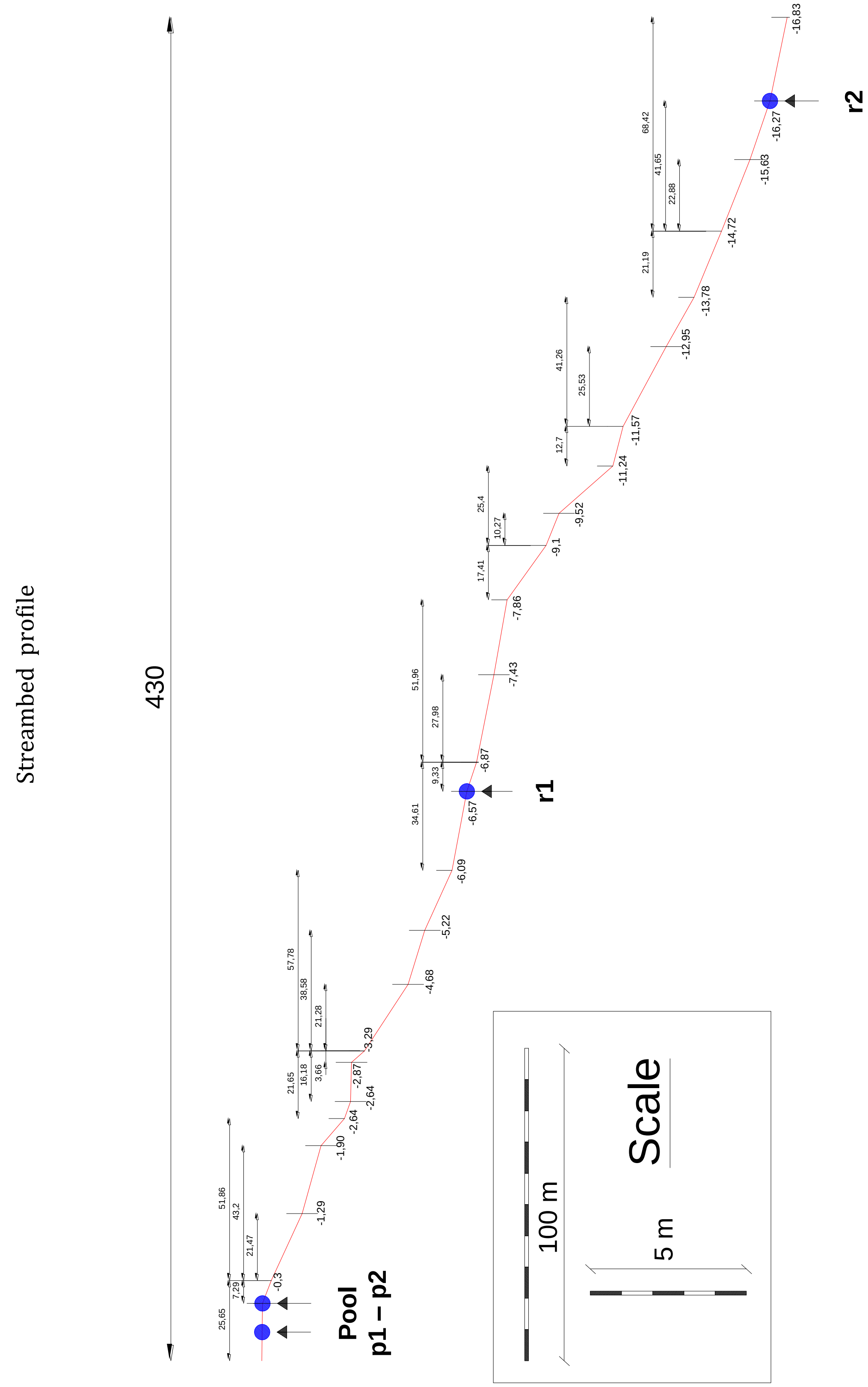
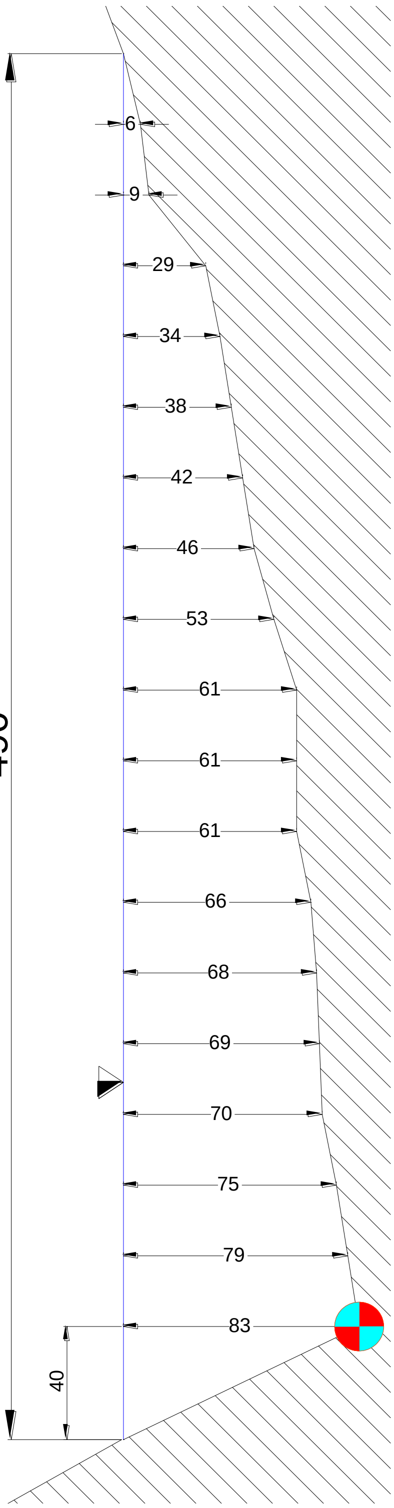
Appendix A



Croos section p1 – Pre-flushing



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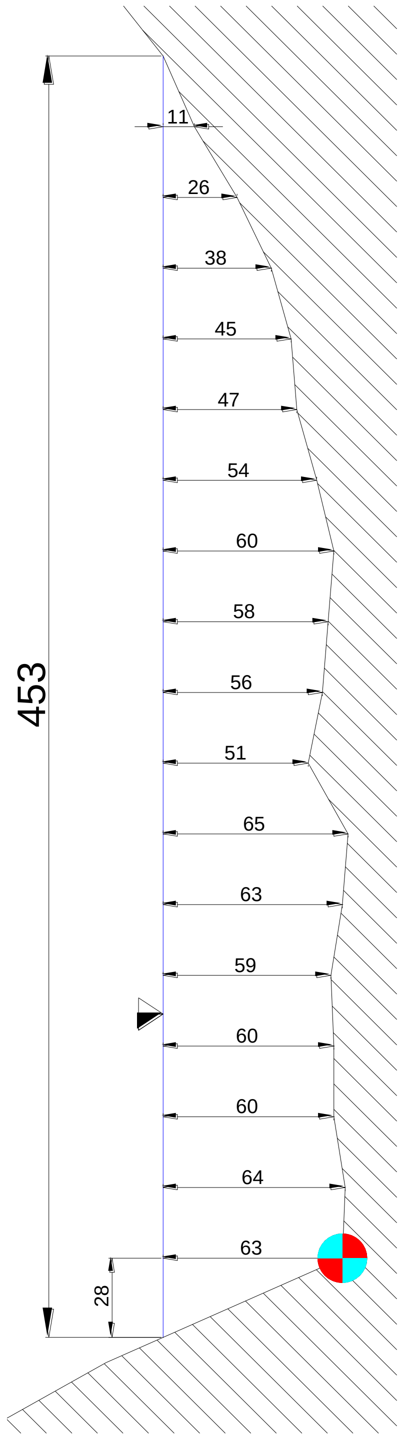
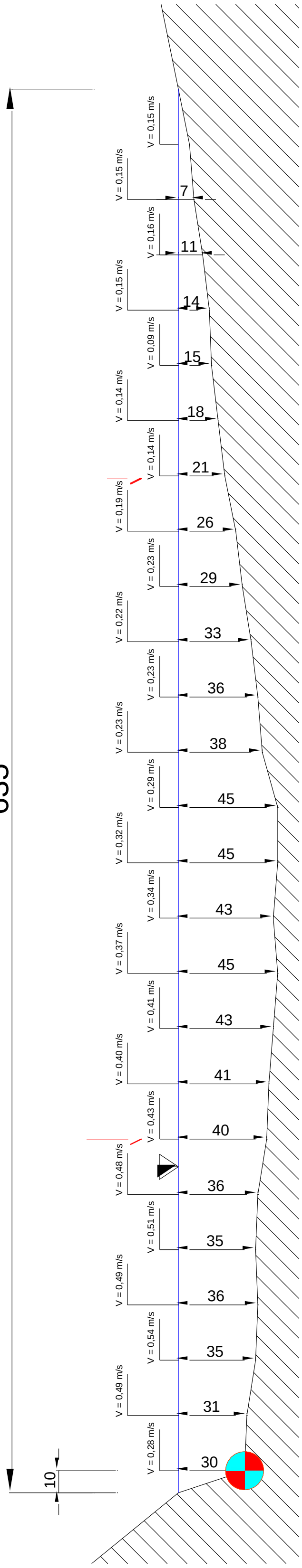


Cross section p2 – Pre-flushing



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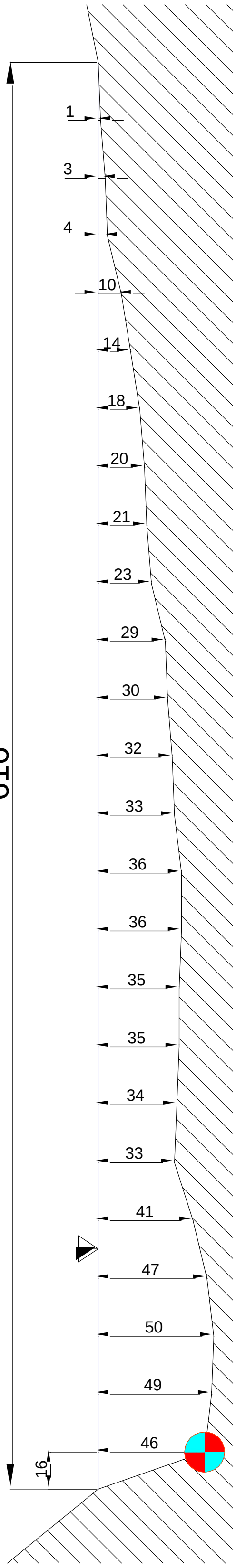

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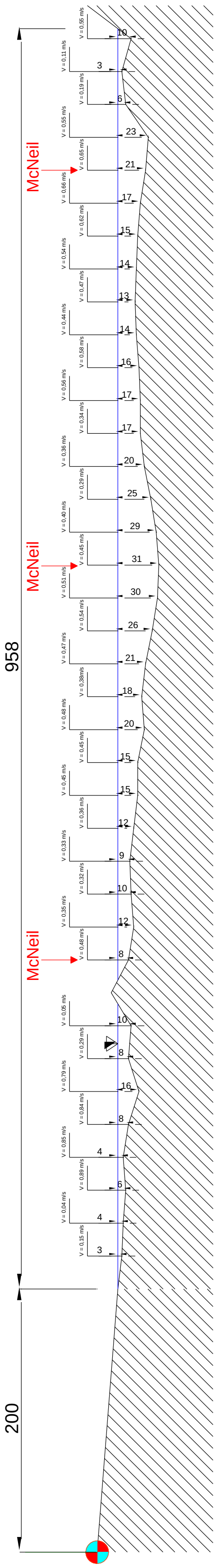

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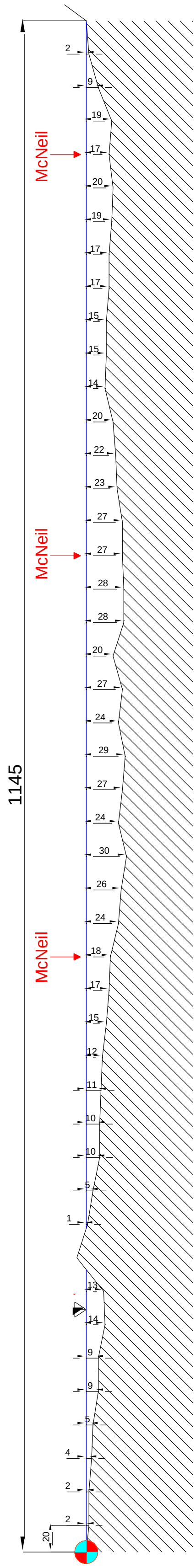

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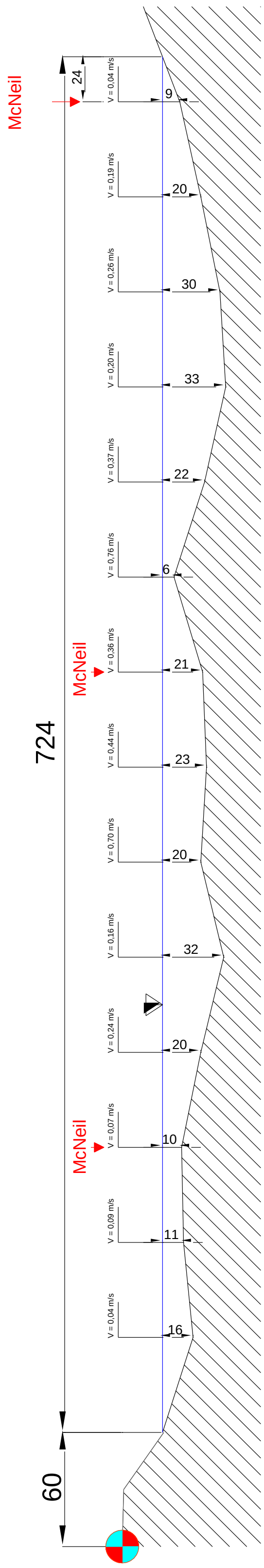

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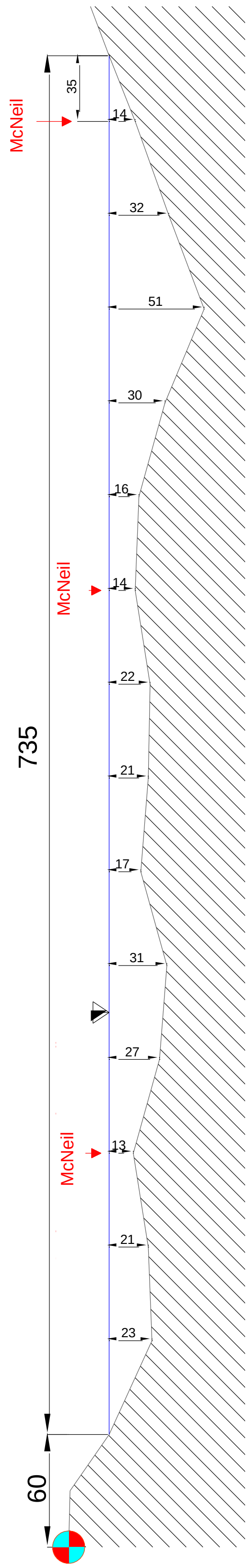

Appendix B

Density of macroinvertebrate taxa recorded at site S3.

Sample date	17-Nov-05	29-Mar-06	8-Aug-06	5-Oct-06	30-Mar-07	7-Aug-07	19-Sep-07	27-Dec-07	27-Jun-08
PLECOPTERA									
Leuctra	301	1364	202	24	40	170	8	8	20
Amphinemura	-	91	-	-	8	-	-	-	-
Nemoura	-	-	-	-	-	-	-	-	-
Protonemura	-	-	155	4	-	-	24	-	56
Perla	4	4	12	-	4	-	-	-	-
Isoperla	170	428	-	-	52	4	-	52	4
Perlodes	12	24	4	-	-	-	-	-	-
EPHEMEROPTERA									
Baetis	603	2675	698	36	2557	1110	48	3226	297
Ephemerella	-	-	4	-	-	-	-	-	-
Ecdyonurus	274	321	95	4	-	44	-	20	-
Epeorus	48	115	-	-	4	-	-	-	-
Rhithrogena	341	844	20	-	-	-	-	20	16
TRICHOPTERA									
Glossosomatidae	-	-	-	-	-	-	-	-	-
Hydropsychidae	194	230	20	16	32	-	16	44	4
Limnephilidae	32	44	28	-	147	67	-	71	159
Philopotamidae	-	-	4	-	4	-	-	-	-
Rhyacophilidae	63	151	12	16	123	8	4	107	12
COLEOPTERA									
Dytiscidae	-	-	-	-	-	-	-	-	-
Elminthidae	-	-	-	4	12	-	4	8	-
Hydraenidae	-	-	-	-	-	-	-	-	-
DIPTERA									
Athericidae	4	20	-	-	-	8	8	-	4
Blephariceridae	-	-	-	-	-	-	-	-	-
Chironomidae	91	2699	674	-	79	16	12	484	1181
Dixidae	-	-	-	-	-	4	-	-	-
Empididae	-	16	4	-	28	-	-	-	-
Limoniidae	28	91	40	4	71	12	4	67	32
Muscidae	-	-	-	-	-	-	-	-	-
Psychodidae	-	4	-	-	-	-	-	-	-
Simuliidae	87	151	91	-	28	40	-	60	63
Tipulidae	4	-	-	-	-	8	-	4	-
OLIGOCHAETA									
Enchytraeidae	-	-	-	-	4	-	-	-	-
Lumbricidae	-	20	-	-	4	-	4	-	-
Naididae	16	-	36	-	8	-	-	4	-
TRICLADIDA									
Crenobia	-	178	4	-	-	-	-	4	-
NEMATODA									
Mermithidae	-	-	-	-	-	-	-	-	-

(Continued)

Sample date	9-Aug-08	23-Sep-08	8-Oct-08	23-Oct-08	7-Nov-08	22-Nov-08	16-Dec-08	5-May-09	4-Jun-09
PLECOPTERA									
Leuctra	4	-	8	-	16	12	28	56	31
Amphinemura	-	-	-	-	-	-	-	-	-
Nemoura	-	-	-	-	-	-	-	-	-
Protonemura	63	4	-	-	-	-	-	1	130
Perla	-	-	4	-	-	-	-	-	-
Isoperla	4	-	-	-	-	24	36	87	4
Perlodes	4	-	-	-	-	8	8	-	-
EPHEMEROPTERA									
Baetis	2315	40	163	36	289	404	773	116	121
Ephemerella	-	-	-	-	-	-	-	-	-
Ecdyonurus	-	-	-	-	4	4	-	8	-
Epeorus	-	-	-	-	-	-	-	-	-
Rhithrogena	-	-	4	-	-	-	-	9	5
TRICHOPTERA									
Glossosomatidae	-	-	-	-	-	-	-	-	-
Hydropsychidae	-	-	-	-	-	28	16	3	2
Limnephilidae	52	-	-	16	420	131	242	1752	2
Philopotamidae	-	-	-	-	-	-	-	1	-
Rhyacophilidae	20	-	103	20	143	289	139	20	13
COLEOPTERA									
Dytiscidae	-	-	-	-	-	4	-	-	-
Elminthidae	-	-	-	-	-	-	-	2	-
Hydraenidae	-	-	-	-	-	-	-	1	1
DIPTERA									
Athericidae	4	4	-	-	-	-	-	-	-
Blephariceridae	-	-	-	-	-	-	-	-	32
Chironomidae	20	12	824	828	174	499	2006	37	22
Dixidae	4	-	-	-	-	-	-	-	-
Empididae	-	-	-	-	16	-	4	3	-
Limoniidae	12	4	12	-	8	-	4	39	15
Muscidae	-	-	-	-	-	-	-	1	-
Psychodidae	-	-	-	-	-	4	-	1	-
Simuliidae	56	-	-	44	36	71	63	-	3
Tipulidae	-	-	-	-	-	-	-	-	-
OLIGOCHAETA									
Enchytraeidae	-	-	-	4	-	-	-	-	-
Lumbricidae	-	-	4	-	-	-	-	1	-
Naididae	-	-	-	-	-	-	-	-	-
TRICLADIDA									
Crenobia	-	-	-	-	-	4	-	3	-
NEMATODA									
Mermithidae	-	-	-	4	-	4	-	-	-

(Continued)

Sample date	21-Jul-09	13-Aug-09	16-Sep-09	14-Oct-09	12-Nov-09	18-Dec-09	4-Feb-10	11-Mar-10
PLECOPTERA								
Leuctra	149	57	2	3	13	13	49	66
Amphinemura	-	-	-	-	-	-	-	2
Nemoura	-	-	-	-	1	-	1	1
Protonemura	277	136	13	2	-	-	1	1
Perla	2	3	1	1	-	1	-	1
Isoperla	-	-	-	-	-	106	310	508
Perlodes	-	-	-	-	1	-	1	1
EPHEMEROPTERA								
Baetis	1605	2438	34	94	443	1148	1526	1977
Ephemerella	-	-	-	-	-	-	-	-
Ecdyonurus	11	51	-	2	9	1	7	8
Epeorus	1	-	-	-	-	2	3	1
Rhithrogena	3	-	-	-	-	5	21	8
TRICHOPTERA								
Glossosomatidae	-	-	-	-	-	-	-	3
Hydropsychidae	1	-	-	-	1	10	20	23
Limnephilidae	-	-	-	-	15	78	153	104
Philopotamidae	14	9	-	2	3	3	2	2
Rhyacophilidae	39	36	17	81	142	130	161	78
COLEOPTERA								
Dytiscidae	-	-	-	-	-	-	-	-
Elminthidae	-	-	-	-	1	-	-	-
Hydraenidae	2	-	1	-	1	-	1	1
DIPTERA								
Athericidae	3	8	3	-	-	1	-	-
Blephariceridae	264	52	4	-	-	-	-	-
Chironomidae	91	9	-	152	528	162	874	5534
Dixidae	-	-	-	-	-	-	-	-
Empididae	-	-	-	-	-	5	3	10
Limoniidae	32	43	-	3	11	19	68	119
Muscidae	-	-	-	-	-	-	-	-
Psychodidae	1	1	-	-	-	1	-	2
Simuliidae	3450	421	7	5	290	333	99	64
Tipulidae	-	-	-	-	-	-	-	-
OLIGOCHAETA								
Enchytraeidae	-	-	-	-	-	-	1	-
Lumbricidae	-	-	-	-	-	-	-	-
Naididae	-	-	-	-	-	-	-	-
TRICLADIDA								
Crenobia	-	1	-	-	2	-	4	2
NEMATODA								
Mermithidae	-	-	-	-	1	1	-	5

Density of macroinvertebrate taxa recorded at site S1

Sample date	May-09	Jun-09	Jul-09	Aug-09	Sep-09	Oct-09	Nov-09	Dec-09	Feb-10	Mar-10
PLECOPTERA										
Capnia	-	-	-	-	-	-	-	-	1	2
Chloroperla	6	-	39	2	-	-	-	-	-	-
Leuctra	663	106	287	305	38	603	747	2166	2352	3898
Amphinemura	547	288	1	-	-	-	-	3	68	224
Nemoura	-	-	2	65	27	36	45	18	9	12
Protonemura	51	70	547	430	122	14	23	61	272	293
Perla	14	5	41	50	23	28	4	49	89	54
Dictyogenus	-	2	1	-	1	-	-	1	2	3
Isoperla	6	10	-	4	8	8	8	17	39	85
Perlodes	-	-	-	-	2	2	-	-	1	-
Brachyptera	1	-	-	-	-	-	12	80	317	174
EPHEMEROPTERA										
Baetis	147	180	187	328	234	50	80	211	346	735
Ecdyonurus	59	59	80	88	101	149	151	226	153	203
Epeorus	7	1	28	20	22	14	4	5	20	6
Rhithrogena	8	14	28	50	35	60	74	120	205	170
TRICHOPTERA										
Glossosomatidae	5	3	-	-	-	-	-	3	13	40
Hydropsychidae	-	-	-	-	-	-	-	-	2	2
Limnephilidae	18	32	6	3	14	46	55	76	64	86
Odontoceridae	-	-	-	-	-	-	-	-	2	-
Philopotamidae	1	-	-	11	9	5	8	14	4-	12
Rhyacophilidae	35	23	23	36	46	133	77	101	281	293
Sericostomatidae	4	-	1	1	1	-	-	-	-	-
COLEOPTERA										
Elminthidae	-	-	-	-	-	-	-	-	-	1
Haliplidae	-	-	1	-	-	-	-	-	-	-
Hydraenidae	7	16	22	32	40	39	1	6	18	67
Hydrochidae	1	-	-	-	-	-	-	-	-	-
DIPTERA										
Athericidae	9	13	3	1	3	2	4	4	4	12
Blephariceridae	3	1	16	2	-	2	-	-	-	-
Ceratopogonidae	-	-	-	-	-	-	-	-	-	-
Chironomidae	762	96	152	180	163	731	601	417	3129	4960
Dixidae	-	1	-	-	-	-	-	-	-	-
Empididae	21	24	1	-	-	-	-	-	5	13
Limoniidae	88	70	20	15	28	55	155	72	106	68
Psychodidae	24	8	-	-	-	2	8	33	173	279
Simuliidae	24	38	29	36	28	10	37	48	161	143
Tipulidae	-	-	-	-	-	2	-	4	6	3
OLIGOCHAETA										
Enchytraeidae	-	2	-	-	-	-	-	-	2	-
Lumbricidae	-	2	-	-	-	1	-	-	4	-
Lumbriculidae	1	5	-	10	9	-	4	8	4	16
Naididae	-	-	-	-	-	-	-	-	6	-
TRICLADIDA										
Crenobia	243	11	47	192	116	151	82	49	544	294
NEMATODA										
Mermithidae	-	1	-	-	-	-	-	1	-	1

Appendix C

Density of macroinvertebrate taxa recorded at altered and free-flow sites.

	Site	ALP	DOS	LOG	LOG	MAD	MAL	MEL	SCA	SOE_1
	Date	Oct-09	Sep-09	Jun-09	Sep-09	Feb-10	Oct-09	Oct-09	Sep-09	May-09
PLECOPTERA	*Capnia*	-	-	-	-	12	-	-	-	-
	Chloroperla	-	-	2	-	-	76	-	2	60
	Leuctra	40	-	132	10	178	90	18	50	54
	Amphinemura	-	-	-	-	14	-	-	-	-
	Nemoura	-	-	6	14	22	46	4	46	2
	Protonemura	2	-	34	4	14	-	16	90	282
	Perla	-	-	-	-	10	-	-	118	-
	Dictyogenus	10	-	-	4	-	84	-	-	-
	Isoperla	2	2	2	6	4	166	42	42	44
	Perlodes	2	-	-	-	-	6	-	-	-
	Brachyptera	226	10	-	-	-	106	-	-	-
	Rhabdiopteryx	-	-	-	-	4	-	-	-	-
EPHEMEROPTERA	*Baetis*	442	8	138	12	52	130	76	76	306
	Ecdyonurus	4	2	54	70	80	274	86	148	20
	Epeorus	-	-	-	2	6	38	-	-	-
	Rhithrogena	212	26	86	112	188	192	6	28	102
	Habroleptoides	-	-	-	-	2	-	-	-	-
TRICHOPTERA	Glossosomatidae	-	-	-	-	2	-	-	-	-
	Hydropsychidae	-	-	-	-	122	-	-	-	-
	Limnephilidae	-	-	-	4	10	88	4	4	56
	Odontoceridae	-	-	-	-	6	-	-	-	-
	Philopotamidae	-	-	-	-	2	-	-	-	-
	Polycentropodidae	-	-	-	-	-	-	-	4	-
	Psychomyidae	-	-	-	-	2	-	-	-	-
	Rhyacophilidae	2	-	-	4	12	46	10	12	2
	Sericostomatidae	-	-	-	-	2	2	-	2	2
COLEOPTERA	Dytiscidae	-	-	-	-	2	-	-	-	-
	Elminthidae	-	-	-	-	32	-	-	2	-
DIPTERA	Athericidae	-	2	6	-	6	-	6	2	-
	Blephariceridae	2	16	-	-	-	-	-	-	-
	Chironomidae	28	24	232	374	346	30	60	780	386
	Empididae	-	-	2	-	4	72	-	4	-
	Limoniidae	6	-	2	4	160	84	16	14	20
	Muscidae	6	-	-	-	2	-	-	-	-
	Psychodidae	-	-	-	-	-	2	-	-	-
	Simuliidae	-	-	228	32	38	2	6	6	6
	Tipulidae	-	-	-	-	2	10	-	-	-
OLIGOCHAETA	Enchytraeidae	-	-	2	-	-	-	-	2	-
	Haplotaxidae	-	2	-	10	-	-	-	-	-
	Lumbricidae	2	-	-	-	-	-	-	-	-
	Lumbriculidae	-	-	2	-	2	-	2	18	4
	Naididae	-	-	2	52	-	-	-	444	-
	Propappidae	-	-	-	-	-	-	-	2	-
TRICLADIDA	*Crenobia*	-	-	10	14	2	4	18	134	-
NEMATODA	Mermithidae	-	-	-	-	-	-	-	2	-

(Continued)

	Site	SOE_1	TAR	VER	VIO_1	MAS	FRO_1	FRO_1	FRO_1	FRO_1
	Date	Sep-09	Oct-09	Sep-09	Sep-09	Oct-09	Apr-09	Jun-09	Aug-09	Oct-09
PLECOPTERA	Chloroperla	8	24	-	-	2	-	6	4	-
	Leuctra	12	216	8	-	6	386	20	-	-
	Nemoura	6	32	62	8	-	-	2	-	-
	Protonemura	2	278	28	4	44	14	130	30	-
	Perla	-	6	-	-	-	-	-	-	-
	Dictyogenus	-	-	-	2	-	8	-	-	2
	Isoperla	10	98	-	10	34	42	-	-	18
	Brachyptera	-	110	-	4	-	-	-	-	-
	Rhabdiopteryx	-	-	-	-	-	72	-	-	-
EPHEMEROPTERA	Baetis	36	1490	78	10	682	990	102	54	78
	Ecdyonurus	-	280	42	8	112	4	-	2	4
	Epeorus	2	14	-	-	18	-	4	-	-
	Rhithrogena	8	260	34	8	112	8	-	4	4
TRICHOPTERA	Hydropsychidae	24	72	-	4	-	-	-	-	-
	Limnephilidae	10	36	76	6	-	68	4	-	-
	Philopotamidae	-	178	-	-	2	-	-	-	-
	Psychomyidae	-	2	-	-	-	-	-	-	-
	Rhyacophilidae	10	92	-	2	28	2	-	-	8
	Sericostomatidae	-	-	-	-	-	-	-	2	-
COLEOPTERA	Elminthidae	6	2	-	-	-	-	2	-	-
	Helodidae	-	-	-	-	-	-	2	-	-
	Hydraenidae	-	8	-	-	-	-	-	-	-
DIPTERA	Athericidae	-	28	4	8	-	-	-	-	-
	Blephariceridae	-	-	-	-	-	-	122	-	-
	Chironomidae	782	112	50	2	-	688	152	106	20
	Dixidae	-	-	-	-	-	-	10	-	-
	Empididae	2	-	4	4	-	-	-	-	-
	Limoniidae	12	80	8	2	2	56	14	26	16
	Muscidae	-	-	-	-	-	2	-	-	-
	Psychodidae	-	-	-	-	-	-	2	-	-
	Simuliidae	-	22	-	8	2	30	32	2	-
	Tipulidae	-	2	-	-	-	2	-	-	-
OLIGOCHAETA	Enchytraeidae	-	-	4	-	-	-	2	-	-
	Haplotaxidae	-	-	2	-	-	-	-	-	-
	Lumbriculidae	-	-	20	-	-	-	-	2	-
	Naididae	-	-	22	-	-	4	-	-	-
	Tubificidae	-	2	6	-	-	-	-	-	-
TRICLADIDA	Crenobia	-	38	8	-	-	2	-	-	-
NEMATODA	Mermithidae	-	-	2	-	-	-	-	-	-

(Continued)

	Site	FRO_1	FRO_2	FRO_2	FRO_2	FRO_2	VIO_2	VIO_2	VIO_2	VIO_2
	Date	10-Mar	9-Feb	9-Apr	9-Aug	9-Nov	9-Feb	9-Apr	9-Aug	9-Nov
PLECOPTERA	*Chloroperla*	-	-	-	-	-	-	-	-	6
	Leuctra	-	270	488	4	240	-	-	-	-
	Amphinemura	-	-	-	-	-	192	390	2	6
	Nemoura	-	-	-	-	-	-	2	12	8
	Protonemura	1	32	8	68	-	6	-	314	2
	Dictyogenus	1	-	-	-	-	-	-	2	-
	Isoperla	-	46	8	2	-	12	26	200	124
	Perlodes	-	-	-	2	-	-	-	4	4
	Brachyptera	-	82	-	-	36	-	-	-	-
	Rhabdiopteryx	39	-	-	-	-	-	-	-	-
EPHEMEROPTERA	*Baetis*	434	388	602	370	142	90	50	572	102
	Ephemerella	-	-	-	-	-	-	-	-	2
	Ecdyonurus	-	2	-	-	6	48	4	56	22
	Epeorus	-	2	-	-	-	-	-	-	4
	Rhithrogena	2	26	10	2	8	8	12	108	646
TRICHOPTERA	Glossosomatidae	-	-	-	-	-	-	-	-	10
	Hydropsychidae	-	-	-	-	-	-	-	2	4
	Lepidostomatidae	-	-	-	-	-	-	-	-	2
	Limnephilidae	2	602	284	6	292	490	56	282	1104
	Rhyacophilidae	5	50	10	6	42	14	-	40	10
	Hydrophilidae	-	-	-	4	-	-	-	-	2
DIPTERA	Athericidae	-	-	-	-	-	4	-	2	4
	Blephariceridae	-	-	-	24	-	-	-	10	-
	Chironomidae	8	408	602	50	26	446	1250	12	12
	Empididae	2	-	2	-	-	-	-	-	8
	Limoniidae	19	88	38	42	172	26	14	68	54
	Psychodidae	-	-	2	-	2	-	2	-	2
	Simuliidae	6	582	270	522	-	262	82	370	14
	Tipulidae	-	2	-	-	-	-	-	4	2
GASTROPODA	Hydrobioidaea	-	-	-	2	-	-	-	-	-
OLIGOCHAETA	Haplotaxidae	-	-	-	-	-	-	-	2	-
	Lumbriculidae	-	-	-	-	2	-	-	-	-
	Tubificidae	-	-	6	-	-	-	-	-	-
TRICLADIDA	*Crenobia*	-	-	-	-	-	2	-	2	2
	Dugesia	-	-	-	-	-	-	4	-	-

	Site	BEL_1	BEL_1	BEL_2	BEL_2	BEL_3	BEL_3	LIR_1	LIR_1	LIR_1
	Date	9-Jun	9-Nov	9-Jun	9-Nov	9-Jul	9-Nov	9-Feb	9-May	9-Sep
PLECOPTERA	*Chloroperla*	-	-	-	-	-	-	24	-	-
	Leuctra	68	152	278	388	1202	52	144	12	34
	Amphinemura	8	576	-	648	-	442	-	4	-
	Nemoura	-	78	-	24	-	124	12	-	2
	Nemurella	22	-	-	-	-	-	-	-	-
	Protonemura	308	70	294	2	292	108	26	24	88
	Perla	2	-	-	-	-	-	-	2	2
	Isoperla	42	116	-	4	-	-	62	14	22
	Perlodes	-	8	-	4	-	16	-	-	-
	Brachyptera	-	-	2	-	-	-	134	-	-

(Continued)

	Site	BEL_1	BEL_1	BEL_2	BEL_2	BEL_3	BEL_3	LIR_1	LIR_1	LIR_1
	Date	Jun-09	Nov-09	Jun-09	Nov-09	Jul-09	Nov-09	Feb-09	May-09	Sep-09
TRICHOPTERA	Beraeidae	-	-	-	2	-	2	-	-	-
	Brachycentridae	-	-	-	86	-	10	-	-	-
	Glossosomatidae	-	-	-	-	-	-	6	-	-
	Goeridae	4	-	-	6	4	22	-	-	-
	Hydropsychidae	-	2	-	8	70	1202	-	-	6
	Limnephilidae	2	52	32	90	4	46	36	16	28
	Odontoceridae	-	4	2	44	-	-	-	-	-
	Philopotamidae	-	10	8	8	4	18	-	-	6
	Polycentropodidae	6	-	6	-	6	186	-	-	-
	Rhyacophilidae	44	42	60	128	74	70	32	8	20
	Sericostomatidae	-	6	-	16	4	26	-	-	-
	Thremmatidae	-	-	-	-	-	-	-	-	4
COLEOPTERA	Dytiscidae	-	-	56	-	-	8	-	-	8
	Elminthidae	4	16	98	486	70	278	-	-	-
	Eubriidae	-	-	-	-	-	-	602	76	-
	Helodidae	-	2	-	8	-	-	-	-	-
	Hydraenidae	-	2	-	4	-	-	-	-	-
	Hydrochidae	-	-	-	-	-	-	10	-	-
DIPTERA	Athericidae	4	12	12	18	4	22	-	-	-
	Blephariceridae	-	-	2	-	36	-	6	4	-
	Ceratopogonidae	-	-	-	-	2	-	4	-	-
	Chironomidae	292	590	98	740	218	358	-	-	94
	Cordyluridae	-	-	-	-	-	-	52	58	-
	Empididae	4	-	10	4	18	4	-	-	2
	Ephydridae	-	-	-	-	-	-	-	4	-
	Limoniidae	50	74	8	32	12	8	-	-	116
	Psychodidae	-	16	-	-	-	-	4	52	-
	Simuliidae	2	2	40	-	26	4	-	-	178
	Stratiomyidae	-	-	-	-	-	-	148	2	-
	Tabanidae	-	-	-	-	6	16	-	-	-
	Tipulidae	-	-	2	-	6	-	-	-	2
GASTROPODA	Lymnaeidae	-	-	-	-	-	4	-	-	-
ODONATA	Cordulegaster	-	-	-	-	-	6	-	-	-
OLIGOCHAETA	Enchytraeidae	2	-	10	2	2	4	-	-	-
	Haplotaxidae	2	-	-	2	-	2	-	-	-
	Lumbricidae	-	36	18	18	58	12	-	-	-
	Lumbriculidae	4	16	56	8	-	4	-	-	-
	Naididae	-	4	-	-	4	2	-	-	4
	Propappidae	8	144	14	58	2	-	-	-	-
HIRUDINEA	*Dina*	-	-	-	-	2	12	-	-	-
TRICLADIDA	*Crenobia*	6	6	10	18	4	-	-	-	-
	Dugesia	2	6	32	-	-	-	-	-	-
	Dendrocoelum	-	-	-	-	-	-	4	-	-
	Polycelis	-	-	-	-	8	-	-	-	-
NEMATODA	Mermithidae	-	2	-	-	-	-	-	-	-

(Continued)

	Site	LIR_1	LIR_2	LIR_2	LIR_2	LIR_2	LIR_3	LIR_3	LIR_3	LIR_3
	Date	Nov-09	Feb-09	May-09	Sep-09	Nov-09	Feb-09	May-09	Sep-09	Dec-09
PLECOPTERA	*Chloroperla*	12	-	-	-	-	6	-	-	12
	Leuctra	156	216	174	110	34	602	602	70	1032
	Amphinemura	56	344	380	-	1000	14	154	-	70
	Nemoura	-	2	-	-	-	10	-	2	16
	Protonemura	34	6	26	30	-	164	140	352	2
	Perla	18	-	-	-	-	-	-	-	2
	Isoperla	30	-	16	-	2-	52	36	4	18
	Brachyptera	142	2	-	-	-	26	-	-	76
EPHEMEROPTERA	*Baetis*	298	422	92	1200	178	602	602	974	636
	Ecdyonurus	118	6	2	70	222	28	116	66	84
	Epeorus	-	6	4	2	2	2	-	12	20
	Rhithrogena	256	2	-	-	-	30	12	-	6
TRICHOPTERA	Beraeidae	-	-	-	-	2	-	-	-	-
	Glossosomatidae	12	-	-	-	-	-	-	-	-
	Hydropsychidae	-	-	6	-	4	-	-	-	-
	Limnephilidae	44	2	2	10	8	10	602	-	368
	Philopotamidae	10	2	-	-	-	2	-	-	-
	Polycentropodidae	-	-	-	-	2	-	-	-	-
	Psychomyidae	-	16	-	-	8	-	-	-	-
	Rhyacophilidae	90	68	56	18	72	16	2	8	56
COLEOPTERA	Elminthidae	102	-	-	4	24	-	-	-	6
	Eubriidae	-	38	92	-	-	18	16	-	-
	Hydraenidae	-	-	-	2	-	-	-	-	-
	Hydrochidae	-	14	-	-	-	-	-	-	-
DIPTERA	Athericidae	6	-	-	6	6	-	-	-	2
	Blephariceridae	-	-	-	6	-	4	2	14	-
	Ceratopogonidae	-	-	-	2	-	-	-	-	-
	Chironomidae	728	-	-	72	634	-	-	118	1400
	Cordyluridae	-	314	488	-	-	376	226	-	-
	Empididae	10	-	-	20	28	-	-	6	82
	Ephydridae	-	10	54	-	-	46	12	-	-
	Limoniidae	136	-	-	84	114	-	-	52	36
	Psychodidae	-	16	62	-	-	22	42	-	-
	Simuliidae	12	-	-	4	14	-	-	624	122
	Stratiomyidae	-	4	-	-	-	602	-	-	-
	Tipulidae	-	-	-	2	28	-	-	-	2
OLIGOCHAETA	Lumbricidae	-	-	4	2	-	-	-	-	-
	Lumbriculidae	-	-	-	4	18	-	2	-	-
	Naididae	-	-	6	22	-	6	-	-	8
	Propappidae	-	-	280	-	-	-	14	-	-
	Tubificidae	-	-	-	2	-	-	-	-	6
TRICLADIDA	*Crenobia*	-	-	-	-	6	-	-	2	-
	Dendrocoelum	-	4	12	-	-	-	-	-	-
	Planaria	-	-	6	-	-	-	-	-	-
OTHER	Gordiidae	-	-	-	2	-	-	-	-	-

(Continued)

	Site	LIR_4	LIR_4	LIR_4	LIR_4	LIR_5	LIR_5	LIR_5	LIR_5	MER_1
	Date	Feb-09	May-09	Sep-09	Nov-09	Feb-09	Apr-09	Sep-09	Nov-09	Feb-09
PLECOPTERA	*Leuctra*	100	304	28	194	214	94	64	86	38
	Amphinemura	-	136	-	42	-	-	-	2	-
	Nemoura	2	2	-	-	4	-	-	-	-
	Protonemura	2	60	52	2	12	38	128	-	-
	Isoperla	-	10	-	4	40	72	2	78	-
EPHEMEROPTERA	*Baetis*	62	602	1000	142	602	602	826	1380	490
	Ecdyonurus	12	18	12	6	6	100	22	42	2
	Epeorus	2	-	-	4	2	-	-	14	-
	Rhithrogena	-	-	-	2	256	88	4	194	2
TRICHOPTERA	Glossosomatidae	2	-	-	-	-	-	-	-	-
	Hydropsychidae	-	-	-	-	2	-	-	16	-
	Limnephilidae	30	14	2	276	602	50	-	98	-
	Philopotamidae	-	-	-	12	-	-	-	-	-
	Rhyacophilidae	8	46	20	114	10	-	22	40	8
	Thremmatidae	-	-	-	2	-	-	-	-	-
COLEOPTERA	Dytiscidae	-	-	4	-	-	-	-	-	-
	Elminthidae	-	-	6	24	-	-	4	20	-
	Eubriidae	204	22	-	-	34	54	-	-	-
DIPTERA	Athericidae	2	-	6	4	-	-	-	-	-
	Blephariceridae	12	4	4	-	2	8	36	-	-
	Ceratopogonidae	-	-	-	6	-	30	-	-	-
	Chironomidae	-	-	456	1000	-	-	106	542	602
	Cordyluridae	602	602	-	-	64	110	-	-	-
	Empididae	-	-	10	102	-	-	34	34	2
	Ephydridae	90	62	-	-	90	18	-	-	-
	Limoniidae	-	-	96	334	-	-	6	48	4
	Psychodidae	62	178	-	-	34	6	-	-	-
	Ptychopteridae	-	2	-	-	-	-	-	-	-
	Simuliidae	-	-	1000	38	-	-	362	50	8
	Stratiomyidae	4	-	-	-	324	56	-	-	-
	Tipulidae	-	-	2	12	-	-	-	2	-
OLIGOCHAETA	Enchytraeidae	-	-	2	26	-	-	-	6	-
	Haplotaxidae	-	-	-	10	-	-	-	4	-
	Lumbricidae	-	-	2	8	-	-	6	4	-
	Lumbriculidae	-	4	4	-	-	-	6	-	-
	Naididae	2	10	-	-	-	-	4	10	48
	Propappidae	6	602	-	-	-	12	-	-	-
	Tubificidae	-	-	-	6	-	-	-	-	-
TRICLADIDA	*Crenobia*	-	-	2	-	-	-	-	-	-
	Dugesia	-	-	-	-	-	-	2	-	-
	Dendrocoelum	-	4	-	-	-	-	-	-	-
	Planaria	-	-	-	-	-	30	-	-	-
OTHER	Gordiidae	-	-	-	-	-	-	4	-	-

(Continued)

	Site	MER_1	MER_1	MER_1	MER_2	MER_2	MER_2	MER_2	AVE	AVE
	Date	Apr-09	Sep-09	Dec-09	Feb-09	Apr-09	Aug-09	Dec-09	Feb-09	May-09
PLECOPTERA	*Chloroperla*	-	-	-	-	-	-	4	-	-
	Leuctra	76	50	18	86	216	768	148	34	104
	Amphinemura	6	-	-	2	-	-	-	4	28
	Nemoura	-	-	-	2	-	-	2	6	2
	Protonemura	6	38	-	24	14	50	2	-	156
	Perla	-	-	-	-	-	6	-	-	-
	Dictyogenus	-	-	-	-	-	-	-	2	-
	Isoperla	20	-	2	42	-	6	92	-	8
	Perlodes	-	-	-	-	-	2	-	2	-
	Brachyptera	-	-	-	24	-	-	-	-	-
EPHEMEROPTERA	*Baetis*	602	640	64	602	602	2400	522	404	276
	Ephemerella	-	-	-	-	-	8	-	-	-
	Ecdyonurus	122	84	80	-	46	166	214	14	114
	Epeorus	8	2	-	24	-	-	90	-	-
	Rhithrogena	184	-	-	116	2	46	26	-	76
	Habroleptoides	2	-	-	-	-	-	-	-	-
	Paraleptophlebia	-	-	-	-	-	-	2	-	-
TRICHOPTERA	Hydropsychidae	4	-	-	12	-	6	82	-	4
	Hydroptilidae	2	-	-	-	-	-	-	-	-
	Limnephilidae	112	-	18	4	-	2	72	26	52
	Odontoceridae	-	-	-	-	-	4	-	-	-
	Philopotamidae	-	-	-	-	-	-	-	4	4
	Rhyacophilidae	34	24	20	20	6	66	48	8	6
	Thremmatidae	-	-	-	-	-	4	-	-	-
COLEOPTERA	Dytiscidae	-	-	-	-	-	-	-	2	-
	Elminthidae	-	-	-	-	-	2	4	12	14
	Hydraenidae	-	-	-	-	-	-	-	2	-
DIPTERA	Athericidae	-	-	-	-	-	-	-	20	30
	Blephariceridae	-	2	-	-	-	-	-	-	-
	Chironomidae	166	4	4	184	332	1800	74	602	144
	Empididae	10	-	-	4	-	42	-	6	26
	Limoniidae	32	-	2	12	6	76	104	30	52
	Simuliidae	2	34	4	36	-	156	22	6	4
OLIGOCHAETA	Lumbricidae	2	2	-	-	-	16	12	-	-
	Lumbriculidae	-	-	-	-	-	12	36	-	-
	Naididae	28	-	-	8	44	4	-	-	326
TRICLADIDA	*Crenobia*	-	-	4	-	-	-	2	2	6
	Dugesia	-	-	-	-	-	2	-	-	-
NEMATODA	Mermithidae	-	2	-	-	-	-	2	-	-

(Continued)

	Site	AVE	AVE	BOG	BOG	BOG	SOE_2
	Date	Sep-09	Nov-09	May-09	Sep-09	Nov-09	May-09
PLECOPTERA	*Chloroperla*	-	2	54	-	6	52
	Leuctra	68	6	8	6	28	48
	Amphinemura	-	44	10	-	18	2
	Nemoura	-	-	10	-	36	8
	Protonemura	70	-	70	2	2	394
	Perla	-	-	-	4	2	-
	Dictyogenus	-	4	-	-	-	-
	Isoperla	6	4	2	-	16	2
EPHEMEROPTERA	*Baetis*	270	48	100	14	30	102
	Ecdyonurus	164	90	30	20	28	18
	Epeorus	4	-	-	-	-	-
	Rhithrogena	14	-	96	30	24	54
TRICHOPTERA	Beraeidae	-	2	-	-	-	-
	Hydropsychidae	10	24	-	-	-	-
	Limnephilidae	-	88	-	-	4	6
	Philopotamidae	26	-	2	-	2	-
	Polycentropodidae	-	-	-	-	-	-
	Rhyacophilidae	12	6	2	-	2	6
COLEOPTERA	Dytiscidae	8	-	-	-	-	-
	Elminthidae	-	2	2	-	10	-
DIPTERA	Athericidae	66	72	8	-	-	2
	Blephariceridae	-	-	2	-	-	4
	Chironomidae	54	148	36	98	60	28
	Empididae	2	2	-	-	-	-
	Limoniidae	8	8	10	12	8	4
	Simuliidae	-	-	12	-	22	2
	Tipulidae	4	-	-	-	-	-
GASTROPODA	Planorbidae	-	-	-	6	-	-
OLIGOCHAETA	Lumbricidae	-	2	-	-	-	-
	Lumbriculidae	4	-	-	-	-	-
	Naididae	162	2	-	-	-	-
TRICLADIDA	*Crenobia*	2	-	-	-	2	-
OTHER	Niphargidae	-	-	-	-	-	2

Appendix D

Mean (±SD) H, D_{Mg}, J, C, T, EPT, density and biomass recorded at site S3.

Date	H	D_{Mg}	J	C	T	EPT	Density	Biomass
17-Nov-05	2.11 ± 0.07	1.63 ± 0.38	0.82 ± 0.05	0.16 ± 0.01	13.3 ± 2.9	10.0 ± 1.0	2271.1 ± 451.8	-
29-Mar-06	1.92 ± 0.10	1.57 ± 0.19	0.70 ± 0.09	0.21 ± 0.04	16.0 ± 3.0	11.0 ± 1.0	9496.6 ± 5533.4	3.65
08-Aug-06	1.71 ± 0.18	1.43 ± 0.19	0.70 ± 0.07	0.25 ± 0.05	11.7 ± 2.1	7.7 ± 1.5	2100.7 ± 731.7	1.99
05-Oct-06	0.87 ± 0.84	0.74 ± 0.69	0.91 ± 0.08	0.56 ± 0.40	3.7 ± 3.1	3.0 ± 2.6	107.0 ± 125.8	0.12
30-Mar-07	0.87 ± 0.08	1.04 ± 0.37	0.41 ± 0.10	0.62 ± 0.08	9.3 ± 4.0	5.3 ± 2.5	3203.0 ± 2257.7	3.16
07-Aug-07	0.91 ± 0.11	0.92 ± 0.27	0.47 ± 0.05	0.57 ± 0.03	7.3 ± 2.1	4.3 ± 0.6	1490.0 ± 660.2	0.80
19-Sep-07	0.81 ± 1.04	0.70 ± 0.89	0.85 ± 0.18	0.63 ± 0.43	4.0 ± 4.4	2.3 ± 2.3	131.0 ± 157.3	0.12
27-Dec-07	0.89 ± 0.25	1.07 ± 0.17	0.39 ± 0.14	0.60 ± 0.18	10.0 ± 2.0	6.0 ± 1.0	4177.6 ± 1579.5	0.93
27-Jun-08	1.08 ± 0.47	1.10 ± 0.22	0.49 ± 0.18	0.51 ± 0.23	9.0 ± 1.7	5.7 ± 1.5	1847.0 ± 346.5	1.28
09-Aug-08	0.45 ± 0.23	0.86 ± 0.29	0.22 ± 0.08	0.83 ± 0.09	7.7 ± 2.5	4.3 ± 2.1	2556.5 ± 1032.7	0.71
23-Sep-08	0.67 ± 0.59	0.62 ± 0.63	0.94 ± 0.11	0.60 ± 0.35	2.3 ± 1.2	1.3 ± 0.6	63.4 ± 48.1	0.03
23-Oct-08	0.49 ± 0.26	0.53 ± 0.41	0.35 ± 0.09	0.78 ± 0.11	4.3 ± 2.5	2.0 ± 1.0	951.3 ± 290.5	0.25
22-Nov-08	1.52 ± 0.13	1.11 ± 0.13	0.71 ± 0.03	0.27 ± 0.03	8.7 ± 1.5	5.7 ± 2.3	1486.3 ± 571.1	0.42
16-Dec-08	1.17 ± 0.10	0.91 ± 0.07	0.55 ± 0.06	0.42 ± 0.04	8.3 ± 0.6	5.7 ± 0.6	3317.5 ± 754.0	1.84
05-May-09	0.83 ± 0.72	0.84 ± 0.50	0.40 ± 0.32	0.64 ± 0.32	6.9 ± 3.5	4.8 ± 2.3	2139.7 ± 1506.3	-
04-Jun-09	1.39 ± 0.24	1.05 ± 0.30	0.79 ± 0.10	0.31 ± 0.07	6.2 ± 2.2	4.1 ± 1.7	379.9 ± 300.1	-
21-Jul-09	1.05 ± 0.39	0.96 ± 0.27	0.49 ± 0.18	0.51 ± 0.19	9.3 ± 2.4	5.1 ± 2.1	5944.3 ± 5473.8	-
13-Aug-09	0.66 ± 0.25	0.87 ± 0.26	0.32 ± 0.10	0.73 ± 0.11	8.2 ± 2.3	4.6 ± 1.6	3262.7 ± 1331.8	-
16-Sep-09	0.80 ± 0.68	0.70 ± 0.62	0.89 ± 0.14	0.57 ± 0.34	3.2 ± 2.0	2.1 ± 1.4	81.1 ± 85.1	-
14-Oct-09	0.94 ± 0.46	0.59 ± 0.35	0.79 ± 0.12	0.48 ± 0.23	4.0 ± 1.9	2.4 ± 1.4	344.7 ± 282.1	-
12-Nov-09	1.16 ± 0.22	0.84 ± 0.27	0.64 ± 0.13	0.41 ± 0.11	6.6 ± 2.2	3.8 ± 1.5	1460.9 ± 1163.1	-
18-Dec-09	1.28 ± 0.23	1.01 ± 0.25	0.61 ± 0.09	0.40 ± 0.10	8.6 ± 2.4	5.5 ± 1.8	2017.6 ± 1223.1	-
04-Feb-10	1.39 ± 0.29	1.16 ± 0.26	0.59 ± 0.09	0.35 ± 0.09	10.7 ± 2.2	7.3 ± 1.6	3303.7 ± 996.1	-
11-Mar-10	1.09 ± 0.35	1.04 ± 0.44	0.48 ± 0.12	0.46 ± 0.14	10.8 ± 3.7	6.8 ± 2.5	8519.5 ± 5368.3	-

Appendix E

Values of H, D_{Mg}, J, C, T, EPT, ASPT score, density, STAR-ICM, EBI and LIFE indeces recorded at altered and free flow sites. For sites where more than one sample was available, values represent the mean.

Site	H	D_{Mg}	J	C	T	EPT	ASPT	Density	STAR-ICM	EBI	LIFE
ALP	1.47	1.41	0.54	0.30	15.00	10.0	6.78	986.0	0.94	9.00	9.29
AVE	1.99	1.84	0.67	0.20	19.25	10.8	6.16	1032.0	0.93	10.00	8.07
B1	2.08	1.64	0.78	0.18	14.67	10.0	6.93	315.3	0.91	9.00	8.39
BEL_1	1.90	2.46	0.56	0.26	29.00	15.0	5.76	2700.0	1.07	11.70	8.12
BEL_2	2.21	2.51	0.65	0.20	30.50	16.5	5.97	3462.0	1.16	12.50	8.10
BEL_3	2.19	2.71	0.62	0.19	33.00	16.5	5.90	3653.0	1.18	12.50	8.12
DOS	1.80	1.23	0.82	0.20	9.00	5.0	6.17	92.0	0.71	8.00	9.00
FR0_2	1.71	1.13	0.67	0.23	13.00	8.0	6.51	1745.5	0.92	8.90	8.46
FRO_1	1.44	1.25	0.58	0.36	12.40	7.2	6.40	777.0	0.79	8.32	8.32
LIR_1	2.17	1.79	0.73	0.16	19.50	12.5	7.48	1526.0	1.10	9.95	8.63
LIR_2	1.74	1.80	0.58	0.29	20.50	10.5	6.08	1838.5	0.98	9.65	7.92
LIR_3	1.84	1.52	0.64	0.22	18.50	11.25	6.79	2896.0	1.03	9.75	8.49
LIR_4	1.78	1.74	0.59	0.25	20.25	9.0	5.50	2246.0	0.85	9.60	8.15
LIR_5	1.83	1.53	0.64	0.26	17.75	8.8	6.14	1993.5	0.96	9.60	8.36
LOG	1.84	1.65	0.65	0.24	17.00	9.5	5.63	834.0	0.81	9.80	8.37
MAD	2.40	3.08	0.69	0.13	33.00	21.0	7.13	1342.0	1.23	12.60	8.27
MAL	2.60	1.89	0.86	0.09	21.00	14.0	7.29	1548.0	1.12	10.40	8.31
MAS	1.25	1.10	0.50	0.45	12.00	10.0	7.89	1044.0	1.05	8.60	8.89
MEL	2.20	1.64	0.81	0.15	15.00	9.0	6.18	370.0	0.90	9.00	8.00
MER_1	1.43	1.20	0.57	0.36	12.50	7.5	6.02	930.0	0.82	7.90	8.54
MER_2	1.72	1.52	0.61	0.27	17.75	11.3	6.17	2419.5	1.00	9.60	8.68
ROA_1	2.08	2.14	0.65	0.20	25.40	15.2	6.62	3701.2	1.10	10.66	8.17
ROA_2*	1.33	1.41	0.50	0.39	16.00	9.6	6.62	2745.4	0.96	8.74	8.40
SCA	2.02	2.18	0.63	0.21	25.00	13.0	5.95	2032.0	0.97	11.00	8.07
SOE_1	1.37	1.38	0.51	0.45	15.00	11.0	6.96	1138.0	0.89	9.00	8.45
SOE_2	1.54	1.57	0.55	0.39	16.00	11.3	7.08	776.0	0.95	9.33	8.25
TAR	2.18	2.04	0.68	0.21	25.00	16.0	6.70	3482.0	1.15	11.00	8.44
VER	2.42	1.92	0.84	0.11	18.00	7.0	4.17	458.0	0.76	10.00	7.17
VIO_1	2.64	2.31	0.95	0.08	16.00	11.0	6.70	90.0	0.91	9.00	8.11
VIO_2	1.58	1.51	0.56	0.31	17.50	10.8	6.27	1928.0	0.98	9.75	7.84

*2009-2010 data