Title: Ecological assessment of a sediment by-pass tunnel on a receiving stream in Switzerland

Short title: Ecological assessment of a sediment by-pass tunnel

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Abstract

Reservoir siltation is a major problem worldwide, decreasing reservoir storage capacity, trapping entrained sediment, and altering the natural sediment regime. Sediment By-pass Tunnels (SBT) are used to connect reservoirs with downstream receiving waters during high flows to reduce sediment accumulation in the reservoir. When operating, large volumes of sediment-laden waters are released into the receiving river for short periods of time (hours). The aim of this study was to assess the impact of SBT events on the downstream riverine ecosystem. We measured physico-chemical properties, sediment respiration, periphyton biomass and chlorophyll-a, and macroinvertebrate assemblages along a 5 km stretch of river during the first two years of SBT operation. During the study, 5 major SBT events occurred. Few changes were found in physico-chemical properties, mainly due to the input of tributaries entering the system. Results showed a clear reduction in sediment respiration, an indicator of ecosystem metabolism, especially after large SBT events. Periphyton levels and macroinvertebrate density/richness also decreased after SBT events. A non-metric dimensional scaling (NMDS) distinguished both temporal and spatial shifts in macroinvertebrate assemblages after SBT events, being related to downstream distance and SBT event magnitude. In summary, SBT events acted as a pulse disturbance, similarly to natural floods, followed by recovery of measured ecosystem indicators. SBT events can enhance sediment and flow connectivity, although the magnitude and frequency of operations should be controlled to prevent catastrophic disturbances.

Introduction

The number of large dams is increasing worldwide due to growing demands for water supplies and energy, thus being one of the greatest man-made modifications of natural flow/sediment regimes in rivers (Nilsson *et al.*, 2005). Besides the 50,000 large dams (>15 meters in height) existing in the world, 3700 new dams are planned for the near future (Zarfl *et al.*, 2014; Nilsson *et al.*, 2005). These large dams retain about 99% of upstream sediment delivery in reservoirs (Williams & Wolman, 1984), decreasing reservoir storage capacity by ca. 570 km³ annually (Sumi, Okano & Takata, 2004; Williams & Wolman, 1984). Besides the well-documented ecological and morphological effects on downstream waters (Brandt, 2000; Graf, 2006; Ward & Stanford, 1979), sediment accumulation in reservoirs also causes technical problems in reservoir operations (Wang & Kondolf, 2014; Kondolf *et al.*, 2014). To overcome technical problems caused by siltation, dam operators have used different

techniques, such as flushing or sluicing of reservoirs, to maintain reservoir capacity while reducing sediment deficits in rivers below dams (Kondolf *et al.*, 2014; Morris & Fan, 1998). However, these techniques have been shown to impact the ecological properties of receiving rivers.

Numerous studies have documented sediment flushing impacts on river biota and morphology (Brandt, 2000); for example, the negative consequences of sediment deposition on riverine fish and invertebrates (Rabení, Doisy & Zweig, 2005; Crosa *et al.*, 2010). Nevertheless, some studies have shown that when proper sediment management is adopted, a balance between technical and ecological demands can be reached while ecosystem degradation can be minimized (Espa *et al.*, 2015). An alternative technology in hydrologically and topographically-suitable impounded rivers is represented by Sediment By-pass Tunnels (SBTs), which are used to connect reservoirs with downstream receiving waters (Sumi, Okano & Takata, 2004). SBTs typically are operated during high flow events, when water and sediment input to reservoirs is maximum (Auel, Berchtold & Boes, 2010). In combination with prior water surface elevation lowering in the reservoir, high flow events can mobilize and transport large volumes of accumulated sediments through the SBT. In some countries, such as Switzerland and Japan, SBTs are fairly common with 12 currently operating in Switzerland and more planned for construction (Kondolf *et al.*, 2014). Unfortunately, SBTs are being increasingly used despite a lack of knowledge on the ecological consequences to downstream river/floodplain ecosystems. We are aware of only one study that assessed the ecological impact of SBT operations (Sumi, Kantouch & Suzuki, 2012), but they did not examine the longitudinal or temporal effects of SBT inputs. Further, no studies to date have investigated SBT events on ecosystem functioning.

Flow regulation from dams has strongly modified the natural flow/sediment regimes of rivers and associated floodplains, thereby influencing water quality, physical habitat and biotic interactions among others (Poff *et al.*, 1997; Wohl *et al.*, 2015). In turn, these changes have lead to ecosystem degradation via river-bed erosion/aggradation, habitat loss and homogenization, and organic matter accumulation (Williams & Wolman, 1984; Brandt, 2000). During the last decades, management programs focusing on flow regimes have been successfully implemented (Olden *et al.*, 2014; Robinson, 2012), whereas other similar programs have shown deficiencies by neglecting sediment dynamics (Yang *et al.*, 2011; Bhowmik & Demissie, 1989). SBTs appear to be a potential management option that could integrate more natural flow and sediment regimes in rivers below dams while still reducing

siltation in reservoirs. Being a rather new management strategy, there is essentially nothing known on the ecological effects of SBT operations on receiving waters and how the characteristic of the operations, such as magnitude, frequency and duration of SBT events, regulate the effects on downstream stretches.

The primary goal of this study was to investigate the ecological impacts of SBT events in a Swiss river during the first years of operation to gain a better understanding of SBT use in flow/sediment regime management. We hypothesized a reduction in ecosystem properties such as organic matter processing, primary production and macroinvertebrate community composition due to the high scouring power of SBT events, with flow magnitude/duration and sediment amount as key factors influencing ecosystem response variables. Further, we expected that tributaries will have an important role in recolonization by macroinvertebrates and organic matter inputs in the system following such events. We tested our hypothesis by sampling different ecosystem properties along a longitudinal gradient below the SBT. Specifically, hyporheic sediment respiration was used as an indicator of organic matter processing, periphyton biomass as an indicator of primary production, and macroinvertebrate assemblages as an indicator of biodiversity response to SBT events.

Study site description

The study was conducted in a lower section of the Albula River, Canton Grisons, in southeast Switzerland (Fig. 1). The Albula is ca. 40-km long and drains a 950 km² catchment with an average elevation of 2300 m a.s.l. Headwaters flow from the Bergün mountains before converging as the Albula river, which then flows into the Vorderrhein river, a tributary of the Rhein river. The Albula is a 6th order river with a mean discharge of 15 m³/s with natural flow peaks occurring in spring/summer due to snowmelt and precipitation; the HQ100 (100-year flood) is $\approx$ 130 m³/s. The Albula, together with the Julia River, is impounded at the Solis reservoir, built in 1986 and located just downstream of the town of Tiefencastel. The Albula river supplies 95% of the natural inflow while Julia only 5%. Below the dam, the river flows through a narrow canyon with an average basal-width of about 25 m. Residual flow in the river below the dam is around 1 m³/s, and can be considered as a minimum flow for the canyon. Three main tributaries (Rain Digl Lai, Grossbach and Prodavosbach) join the Albula in the canyon stretch, providing additional flow and are potential sources of sediment (Fig. 1).

A Sediment By-pass Tunnel (SBT) was built at Solis in 2012 to reduce siltation of the reservoir. The tunnel is 973-m long, 4.5 x 4.7-m in size, has a 1.9% slope and flow capacity of

170 m³/s, and connects the middle part of the reservoir with the downstream receiving river at ca. 300 m below the dam. The tunnel is operated during high flow events when large volumes of sediment-laden water enter the upper reservoir (Auel, Berchtold & Boes, 2010). The 5-km study section was located downstream of Solis dam at an average elevation of 740 m a.s.l. and slope of 1.6%. Six sampling sites were situated along the stretch to account for longitudinal distance effects and potential flow/sediment inputs from two major tributaries. Site 1 was ca. 50 m upstream of the SBT outlet and used as a control site unaffected by SBT events. Site 2 was ca. 200 m downstream of the SBT outlet. Sites 3 and 4 (400 and 500 m downstream) were situated above and below the first tributary (Rain Digl Lai), whereas Sites 5 and 6 (4.9 and 5.0 km downstream) were above and below the lower tributary (Prodavosbach) (Fig.1). Two sampling points also were situated at the first and lower tributaries, 10 m upstream of the confluences with the main river. The third major tributary (Grossbach, ca. 1.5 km downstream) was inaccessible and thus not included in this study. As a normal dam operation, spillway and bottom outlet releases occur throughout the year, affecting all study sites.

Methods

Sediment by-pass tunnel operation and sampling campaigns

The study took place during 2014 and 2015 and included 5 separate SBT events that differed in their physical characteristics (Table I). For instance, SBT events showed differences in maximum discharge through the tunnel, reaching a maximum discharge of 179 m³/s during August 2014 operations. The maximum total discharge also fluctuated among events, from more than 270 m³/s in August 2014 to only 62 m³/s in June 2015. The difference between maximum SBT discharge and the maximum total discharge was caused by simultaneous dam overflow during SBT operations (Fig. 2). The total sediment released through the tunnel was estimated by geophone sensors located in the tunnel outlet and correspond to suspended sediment between 0 and 22 mm (Rickenmann & McArdell, 2007; Hagmann, Albayrak & Boes, 2015). Flushing releases also were used to flush fine sediments deposited on the river bed after each SBT event (Fig. 2).

Sampling campaigns were conducted five times each year in late spring, summer and early autumn, when the possibility of having a high flow event was greatest. Sampling dates were adjusted each year to account for sampling after SBT events, although a sampling was performed in May of each year as a “control” sampling un-affected by SBT operations (Fig. 2). In 2014, three major SBT events occurred. After the first SBT event in late May, sampling

was conducted just after flow returned to residual levels. Two major SBT events then occurred the following two months, but sampling was postponed until after the last event in August due to flushing releases maintaining high flows in the canyon. Two further sampling campaigns were conducted in September and November 2014 to examine recovery following these major SBT events. The first SBT event in 2015 also occurred in late May with a post event sample collected soon after (Fig. 2). Another SBT event occurred in June 2015 followed by flushing releases over a number of days. Sampling campaigns took place once flows returned to residual levels in July, September and October 2015.

Flow regime and physico-chemical parameters of study sites

Inflow and discharge data were obtained by the company (EWZ) operating the dam: inflow discharge represents the total discharge of the two rivers entering Solis reservoir, Albula and Julia; SBT discharge is measured at the tunnel; and spillwater and bottom outlet are measured at the dam (Fig. 2). All data were recorded every 15 minutes. On each sampling date, a 0.5-litre water sample was collected at each site for chemical analysis in the laboratory. Water samples were filtered (cellulose nitrate filter 0.45 μm , Sartorius Stedim Biotech) and analyzed for dissolved organic carbon (DOC), total organic carbon (TOC), nitrate-N ($\text{NO}_3\text{-N}$), phosphorus ($\text{PO}_4\text{-P}$) and total phosphorus (TP) using methods described in detail in Tockner *et al.* (1997). Electrical conductivity (E.C. at 20°C) and temperature were measured in the field using a portable conductivity meter (WTW, Germany).

Sediment respiration, benthic organic matter and sediment size distribution

Hyporheic sediment respiration (SR, $n = 3$ per site and date) was measured as change over time in dissolved oxygen concentration. Plexiglas® tubes (5.2 cm diameter, 32 cm long) were used to incubate hyporheic sediments (Uehlinger, Naegeli & Fisher, 2002). The top 10-15 cm of the streambed were removed first to avoid respiratory effects of epilithic periphyton. Hyporheic sediment was sieved to eliminate particles >8 mm and standardize samples among sites (Doering *et al.*, 2011). Each tube was half-filled with sediments and the remaining portion filled with stream water, closed using rubber stoppers, and then buried into the streambed for 3 to 4 hours for incubation. Oxygen concentration and temperature were measured before and after incubation using a portable oxygen meter (Hach HQ40d connected to a LD10101 oxygen probe). After incubation, the contents of each tube were stored at -20°C

for later analysis of organic matter. Based on the consumption of O₂ in the tube (r ; mg O₂/ m³ h⁻¹), respiration per kg of sediment was calculated (R , mg O₂/ kg sed h⁻¹) as follows:

$$R = r V_w / G_w$$

where V_w was the water volume in the tube (m³) and G_w the weight of dry sediment (kg sed). R was then normalized using the Arrhenius equation at a reference temperature of 20°C to account for seasonal variation in water temperature:

$$R_{20^{\circ}\text{C}} = R_T / 1.072^{T-20^{\circ}\text{C}}$$

where $R_{20^{\circ}\text{C}}$ is respiration rate at 20°C and T is water temperature in the tube at the end of the incubation as described in (Naegeli & Uehlinger, 1997). In the laboratory, coarse organic matter (CPOM, >2 mm) was separated from sediments using a 2-mm sieve. CPOM samples were dried at 60°C for 48 h, weighed, combusted at 450° for 4 h and reweighed. Sediments, together with organic particles <2 mm, were dried at 60°C for 48 h, combusted at 450°C for 4 h and reweighed to determine fine particulate organic matter (FPOM). CPOM and FPOM were expressed as g of ash-free dry mass (AFMD)/kg sediment. Combusted sediments were sieved to separate grain size fractions 8-4 mm, 4-2 mm, 2-1 mm, 1-63 µm, and < 63 µm, and the median grain size (d_{50}) was calculated using Gradistat v8 (Blott & Pye, 2001).

Periphyton and macroinvertebrates

Five cobbles were randomly collected from each site and date, stored in plastic bags and kept at -20°C until analyzed. Periphyton was removed from the surface of each cobble using a metal brush and rinsed with deionized water. Two subsamples of the suspension were filtered through Whatman GF/F filters. One filter was dried at 60°C for 24h, weighed, combusted at 450°C for 4 h and reweighed for determination of ash-free dry mass. The other filter was used for chlorophyll-*a* analysis. Chlorophyll-*a* was extracted with hot ethanol and analysed by HPLC (Meyns, Illi & Ribí, 1994). The surface area of each cobble was calculated by wrapping the stone with aluminium foil and using a weight-to-area relationship (Bergey & Getty, 2006).

Three benthic samples were collected from each site/date using a Hess sampler (250-µm mesh, 0.04 m² area) and preserved in the field with 70% ethanol. Concurrently, samples from tributaries were collected in 2015 to use as a reference for sites un-affected by dam or SBT releases. In the laboratory, macroinvertebrates were handpicked using a stereomicroscope (10x magnification), identified to family level (Ephemeroptera, Plecoptera,

Trichoptera, Diptera, Crustacea) or order level (Oligochaeta) and counted. Macroinvertebrate density and taxa richness were calculated for each sample.

Data analysis

The coefficient of variation (CV) was used to examine the temporal variation of physico-chemical characteristics among sites. Physico-chemical measures were analyzed comparing all data among sites using principal component analysis (PCA) on log-transformed data. Normal data ellipses (main stream versus tributary) were generated with the standard probability of 0.69.

To calculate the percentage rate of change after each event, rates of measured ecosystem properties (SR, periphyton biomass and chl-a, and macroinvertebrates density and taxa richness) at each site were compared to rates at Site 1 under residual flow (May 2014 and May 2015, respectively). A two-Way ANOVA on $\log(x+1)$ transformed data followed by post hoc Tukey's test was used to compare rates at affected sites (2-6) with non-affected Site 1. A stepwise multiple regression on $\log(x+1)$ transformed data was used to determine what predictive factors (distance to tunnel outlet, Max SBT Q, Max tot Q, Cum. water volume, event duration and Cum. sediment volume) influenced measured ecosystem properties. Variation inflation Factor was used to check for collinearity among the explanatory variables. Due to collinearity, Cum. water volume and event duration were dropped from the model. Data from Site 1 were excluded in this analysis, since this site was not affected by SBT events. Variables were excluded in a backward direction when the Akaike Information Criterion (AIC) values of the alternative models were lower and the variables involved were significant (based on ANOVA). Relative variable importance was calculated according to the metric '*LMG*' using the package '*relaimpo*' in R (Grömping, 2006).

Changes in macroinvertebrate composition were analysed using non-metric multidimensional scaling (NMDS) based on Bray-Curtis distance and calculated on $\log(x+1)$ transformed densities. Two NMDS analyses were carried out, one by-site to examine longitudinal patterns, and one by-date to examine temporal patterns. Samples from sites non-affected by SBT events (Tributaries and Site 1) were excluded in the by-date analysis to avoid interference with the affected samples. Analysis of Similarity (ANOSIM) was conducted to test for among-group differences. All analyses were carried out using R software (R Development Core Team, 2015).

Results

Flow regime and physico-chemical properties

Timing and magnitude of the events matched the occurrence of floods in the system (Fig. 2). Sub-daily fluctuations in flow were caused by hydropeaking by an upstream dam in the Julia river and other regulated inflows from upstream reservoirs. The main physico-chemistry differences were found between main stream and tributaries. DOC and TOC concentrations were similar among stream sites and tributaries, although stream sites showed greater temporal variation (CV stream: DOC 36%, TOC 45%; CV tributaries: DOC 17%, TOC 22%) (Table II). Nitrate concentrations were high in both tributaries (on average 0.57 mg N/l) and temporally stable (CV = 17% on average for both tributaries), thus causing higher values at Sites 5 and 6 (on average 0.45 mg N/l) than sites upstream (on average 0.31 mg N/l). Electrical conductivity was higher in the main river (on average 430 $\mu\text{S}/\text{cm}$) than in the tributaries (on average 376 $\mu\text{S}/\text{cm}$) and temporally constant at sites not affected by tributaries. Phosphate in tributaries showed the greatest CV (71%). The PCA revealed 2 different groups with main stream samples situated in the upper part of the plot and tributaries on the left-down side of the plot (Fig. 3). The first PCA axis (PC1) explained 41.5% of the variance and was correlated with DOC and TOC, while PC2 explained 19.9% of the variance and was correlated to NO_3 and TP.

Sediment respiration, benthic organic matter and sediment size distribution

SR showed high spatial heterogeneity under residual flow conditions, ranging from 0.5 to 1 $\text{mg O}_2/\text{kg sed h}^{-1}$, but with no significant differences between Site 1 and Sites 2-6 ($p = 0.96$). All sites showed a decrease in sediment respiration after the first SBT opening in 2014 (Fig. 4a). Sites downstream of the SBT showed decreases of 27% on average, while Site 1 decreased 15% due to flushing flows ($p = 0.33$). In 2014, the reduction in SR was greatest after the summer SBT events, the highest magnitude flows during this study. Sites below the SBT outlet decreased by ca. 60% of initial SR rates while Site 1 showed a significantly lower decrease of only 28% ($p = 0.048$). In September 2014, SR at Site 1 increased two-fold to 1.29 $\text{mg O}_2/\text{kg sed h}^{-1}$, while SR increased by only 20% at Sites 2-6 ($p < 0.001$) (Fig. 4a). A flushing release in October 2014 caused a decrease in SR at Sites 1-3 and an increase at Sites 5-6.

SR declined significantly at Sites 2-3 after the first SBT event in 2015 (54% and 85%, respectively), while SR increased by 9% at Site 1 ($p = 0.01$) (Fig. 4b). A month later, SR at Site 1 increased to ca. 30% greater than rates before the first SBT event in 2015, showing significant differences with affected sites ($p = 0.03$). Here, SR at Sites 2-3 were 15% and 50% of initial rates, whereas SR at Site 4 decreased to $0.28 \text{ mg O}_2/\text{kg sed h}^{-1}$. In September 2015, SR at Site 4 almost doubled and remained the same until the end of the study. Sites 5-6 showed a similar pattern with increases in SR at the end of the study period. Stepwise regression showed that respiration was related to maximum SBT Q and sediment amount of each event ($R^2 = 0.22$), with both variables showing similar relative importance (Table III).

FPOM levels were an order of magnitude greater than CPOM levels in hyporheic sediments (Fig. 4c-f). FPOM differed among sites, whereas CPOM varied little among sites with similar values at all sites over time; an exception was a peak value in CPOM at Site 3 in September 2015 ($0.8 \text{ g AFDM/kg sed}$). The d50 values increased at Sites 2-4 after the first SBT event in 2014 (Fig. 4g), but returned to pre-event d50 values by the next sample date. Sediment size at Sites 1, 5 and 6 remained essentially the same through 2014. After the first SBT event in 2015, sediment sizes at Sites 2-3 decreased from 2 and 3 mm to 0.78 and 0.25 mm, respectively, which persisted until the end of the study at Site 3. The d50 remained basically the same at the other sites throughout 2015, and d50 at Site 2 increased to pre-event values within 2 months (Fig. 4h). FPOM amounts were related to the distance to the tunnel outlet (95% of importance) and the amount of sediment in each event (5% of importance) ($R^2 = 0.37$), while CPOM levels were related with the distance to the tunnel and to maximum discharge ($R^2 = 0.13$). The variation in d50 was explained only by sediment content ($R^2 = 0.19$) (Table III).

Periphyton biomass

Periphyton AFDM and chlorophyll-a showed similar response patterns over time to SBT events. Periphyton AFDM decreased at all sites following the first SBT event in 2014 to values near zero (Fig. 5a). Site 1, only affected by flushing releases from the dam, showed the least reduction in periphyton AFDM, decreasing by 25% (from 8.6 to 6.4 g AFDM/m^2 ; $p < 0.001$). The reduction in periphyton AFDM was dramatic at all sites ($p = 0.99$) following the large SBT events in summer 2014 (average $< 1 \text{ g AFDM/m}^2$), followed by a rapid recovery to pre-event levels, showing differences between the non-affected site and affected sites ($p = 0.003$). Periphyton AFDM then showed a general decrease at all sites in October 2014,

displaying significant differences between non-affected site and affected sites ($p < 0.001$). After May 2015, all sites showed a reduction in periphyton biomass ($p = 1.0$). Afterwards, periphyton showed a recovery at all sites until the end of the study period regardless of the June SBT events (all $p = 1.0$). Stepwise regression indicated that periphyton AFDM was related to distance to the tunnel (8% of importance), maximum discharge (47%) and sediment amount of each event (44%) ($R^2 = 0.43$) (Table III).

Periphyton chlorophyll-a decreased at all sites after the first SBT event in 2014 showing at Site 1 a significantly lower decrease than affected sites ($p < 0.001$). Periphyton chlorophyll-a remained low at all sites throughout summer ($p = 1.0$) before increasing in September (Fig. 5c). In October-November 2014, periphyton chl-a increased at Sites 1 and 5, decreased at Sites 2 and 6, and remained the same at Sites 3-4. Periphyton chl-a decreased at all sites below the SBT after the first event in 2015, showing differences between non-affected site and affected sites ($p < 0.001$) (Fig. 5d). The second SBT event in 2015 did not reduce periphyton chl-a at any site, with Sites 2-3 actually showing increases in chl-a to >250 mg/m². By October, periphyton biomass (AFDM, chl-a) increased to values greater than before the SBT events of 2015 at most sites. Variation of periphyton chl-a was best explained by maximum SBT discharge ($R^2 = 0.46$) (Table III).

Macroinvertebrates

Macroinvertebrate densities in the river ranged between 658 ind/m² (Site 4) to 10,000 ind/m² (Site 2) before the first SBT event in 2014 (Fig. 5e). Densities at Sites 2 and 3 decreased by 89 and 81%, respectively, after the first SBT event in 2014, but increased at Site 4 by 124% (from 658 to 1475 ind/m²). Site 1 showed a decrease of 43%, but was not significantly different than affected sites ($p = 0.99$). Densities decreased by 98% at all sites after the summer events of 2014 relative to densities in May ($p = 0.50$). Most sites still showed low densities two months after the SBT events of 2014, except for Site 4 (Fig. 5e). In 2015, macroinvertebrate density at Site 1 decreased after flushing flows in May and June, remaining at low numbers until the end of the study. Sites 2-6 showed a similar trend, decreasing after the May and June SBT events and then increasing to values greater than before the SBT events. Site 4 showed a peak in July and Site 6 in July and October, reaching densities up to 15,000 ind/m².

Macroinvertebrate taxa richness showed a similar pattern as density in 2014, decreasing on average 22 and 73%, respectively, to the sequential events (Fig. 5g). Taxa

richness increased after the SBT events of 2014 to pre-event levels, being especially evident at Sites 5-6. Taxa richness showed no severe response to SBT events in 2015, although a slight decrease was evident at Site 3 after the first SBT event. Both macroinvertebrate density and richness were related to maximum SBT discharge as well to distance to the tunnel outlet, with distance being relatively less important than maximum discharge (90%) (density, $R^2 = 0.34$ and taxa richness, $R^2 = 0.25$, respectively) (Table III).

The effects of SBT events on macroinvertebrate assemblages differed between 2014 and 2015 (Fig. 6). Macroinvertebrate composition was similar among sites in 2014 (ANOSIM, $R = 0.0$, $p = 0.70$), whereas two distinct groups were evident in 2015 (ANOSIM, $R = 0.5$, $p = 0.001$). In 2015, Sites 1-3 were grouped together and Sites 4-6 were clustered with the two sampled tributary assemblages. Macroinvertebrate assemblages also changed over time, showing a significant temporal shift after SBT events with a recovery to previous assemblage compositions after the SBT events (Fig. 6c, d). For example, assemblages in 2014 shifted to a dominance of Oligochaeta and Chironomidae initially after the SBT events, followed by another shift to a more diverse assemblage as the system recovered (ANOSIM, $R = 0.6$, $p = 0.001$). In contrast, after the first SBT event 2015, assemblages also shifted but returned to pre-event assemblage composition within a month after the events (ANOSIM, $R = 0.3$, $p = 0.001$).

Discussion

One main advantage of SBTs is that their operation characteristics (timing, frequency, magnitude, duration) can simulate natural flow in dam-regulated rivers since they are operated only during natural high flow events. This is essential in river management since flow pluses are important drivers of biotic and abiotic interactions, organism life histories and ecosystem functioning (Bunn & Arthington, 2002; Lytle & Poff, 2004). SBTs can re-activate the longitudinal dimension of sediment regimes as well, providing sediments to downstream systems impacted by the dam. Therefore, simulating the characteristics of the natural flow/sediment regime is essential if the aim of management is to simulate a near natural ecosystem response to floods (Wohl *et al.* 2015). Importantly, our results clearly show that specific ecosystem properties respond quite differently to SBT events in relation to flow/sediment related parameters. Hyporheic properties, such as sediment respiration, organic matter content or sediment size, were mostly influenced by the maximum discharge released through the SBT, sediment released through the tunnel and the distance to the SBT outlet,

whereas periphyton and macroinvertebrates characteristics were mainly affected by the maximum discharge released through the SBT.

SBT events had little influence on physico-chemical properties along the study reach, although we observed a tributary influence on water chemistry that clearly reflected the relative flow contribution from tributaries. As shown in other experimental flood studies (Robinson & Uehlinger, 2008), a major change in the chemical make-up of receiving waters from flow releases is not expected as no changes were made to alter the water chemistry upstream. Some short-term changes may occur during the flow event depending on the origin of the released water (e.g., hypolimnetic versus hyperlimnetic releases), but such changes were not recorded in our study.

More than 150,000 m³ of sediment passed through the tunnel into the downstream stretch during the study. It is likely that most of the sediment was transported downstream of the study stretch due to the narrow configuration of the canyon that enhanced flow competence. Similarly, water release after SBT operations ensure a minimum deposition within the wet channel. However, we observed sediment accumulations in gravel bars in wider parts of the canyon within the study section, which can induce the formation of new habitats. These freshly deposited sediments can be later re-distributed along the canyon or colonized by riparian vegetation. Only the large events influenced hyporheic respiration, suggesting small sediment-laden events lacked the competence to mobilize deeper layers of the stream bed (Uehlinger, Naegeli & Fisher, 2002), as seen in Site 1 affected by flushing operations. Further, SBT alterations on hyporheic respiration were unrelated to distance from the SBT outlet, showing similar impacts at all sites below the SBT. However, changes in sediment organic matter were associated with distance from the SBT outlet, possibly indicating deposition of organic matter in areas near the outlet. Our hypothesis of tributaries providing organic matter to the river could not be confirmed.

All SBT events reduced periphyton levels and macroinvertebrate densities in the top layer of the stream bed, likely caused by the scouring effects of high flows. This result was expected as similar responses have been observed in other studies of flood (natural and e-flows) effects on rivers (Robinson, Aebischer & Uehlinger, 2004; Espa *et al.*, 2015; McMullen & Lytle, 2012). The effect of medium and high discharge peaks on macroinvertebrates seemed to be similar regardless the content of sediment. During the first operation of 2015, even though more than 95.000 m³ of sediment were released, the reduction on density on Site 1 (only affected by overflow) was similar to the effect on sites affected by

SBT sediment releases. Similar results were shown by the NMDS, where Sites 1-3 clustered together, showing a similar response to operations despite the differences in flow and sediment conditions. Furthermore, biotic response patterns were associated with event magnitude, with larger events having greater impacts and recovery to pre-event communities being faster after smaller events (Robinson, 2012). Community composition showed a greater shift from June to August compared to May to June in 2014, and the recovery to a pre-event community was faster in 2015, when the discharge peak was lower, regardless of the high amount of sediments released in the May 2015 event. Tributaries in the section acted as a source of invertebrates, as they appeared to enhance the recovery of macroinvertebrate assemblages following SBT events (Robinson, Uehlinger & Monaghan, 2003).

As shown in this study, SBT events can modify key components of ecosystem structure and function. An extreme magnitude or frequency of events could highly modify organic matter decomposition processes, and primary or secondary production in the system, thus having severe ecosystem consequences (Aristi *et al.*, 2014; Ponsatí *et al.*, 2015; Jones *et al.*, 2012). However, the near-natural characteristics of events in our study case avoided dramatic unexpected ecosystem impacts. In the short term, SBT operations can increase the flow/sediment variability that often is lost in flow-regulated rivers (Naiman *et al.*, 2008; Petts, 2009). A permanent positive change in the system would take several years of adaptive management operations, similarly to experimental floods (Robinson, 2012). Our results already suggest the potential of SBTs in better balancing between technical and ecosystem demands, and thereby improving the sediment regime in rivers downstream of storage reservoirs suffering siltation (after Wohl *et al.*, 2015). In order to reduce the risk of permanent ecological impacts in the long-term management of such riverine ecosystems, the possibility of defining thresholds for SBT flow events (e.g., maximum discharge, sediment concentration) should be considered, as has been implemented in other rivers in terms of sediment management (Crosa *et al.*, 2010).

In conclusion, we found that SBTs, if ecologically implemented (i.e., having the operational characteristics similar to the natural flood features of a system), can improve the longitudinal connectivity of sediments of rivers impounded by dams. Indeed, SBT events can be used as environmental flows to simulate more natural flow/sediment regimes of receiving waters. Determination of SBT event thresholds for each system are recommended to reduce ecological risk from extreme magnitude events that are analogous to a catastrophic flood disturbance (Olden *et al.*, 2014; Konrad *et al.*, 2011). Knowledge regarding the ecological

impacts of SBT events is still poor and currently based on other e-flow studies concerning sediment releases from reservoirs. Consequently, further studies are needed to better understand ecological response patterns and assist learning in the adaptive management of flow-regulated rivers.

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625 **Tables**

626 Table I: Sediment bypass tunnel events in 2014-2015. Maximum discharge through the
 627 bypass tunnel (Max SBT Q, m³/s), maximum discharge during an event at sites S2 and S3,
 628 including overflow release from the dam (Max tot Q, m³/s), cumulative water volume
 629 transported through the SBT (Cum. water volume, m³), event duration (hours), and
 630 cumulative sediment volume transported through the SBT (Cum. sediment volume, m³).

Event	Max SBT Q (m ³ /s)	Max tot Q (m ³ /s)	Cum. water volume (m ³)	Event duration (h)	Cum. sediment volume (m ³)
23/05/2014	87.2	115.3	2,763,764	10.5	9,973
29/06/2014	113.9	173.4	1,708,560	5.8	12,228
13/08/2014	179.0	273.7	7,407,582	14.5	31,762
15/05/2015	107.6	107.6	2,432,669	11.2	96,775
9/06/2015	62.0	62.0	1,117,974	8.5	51

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632 Table II: Chemical characterization of each study site (2014-2015).

Parameter	<i>n</i>	DOC mg C/l	TOC mg C/l	NO ₃ mg N/l	PO ₄ µg P/l	TP µg P/l	Conductivity µS/cm 20°C
Site 1	mean	10	0.73	0.9	0.3	2.3	10.8
	SD		0.29	0.3	0.06	0.6	5.4
	CV		40	33	20	26	50
Site 2	mean	10	0.76	1.03	0.31	1.9	8.7
	SD		0.22	0.45	0.06	0.7	7.1
	CV		29	44	19	37	82
Site 3	mean	10	0.75	1.09	0.32	1.7	16.4
	SD		0.27	0.52	0.07	0.6	12.4
	CV		36	48	22	35	76
Site 4	mean	7	0.84	1.14	0.36	2.9	12.9
	SD		0.26	0.4	0.05	2.3	6.4
	CV		31	35	14	79	50
Site 5	mean	7	0.89	1.16	0.45	1.7	16.9
	SD		0.35	0.49	0.16	1.2	11.8
	CV		39	42	36	71	70
Site 6	mean	8	0.75	0.98	0.44	1.4	13.4
	SD		0.34	0.33	0.18	1.1	13.9
	CV		45	34	41	79	104
Tributary 1	mean	6	0.94	1.14	0.5	2.6	10.7
	SD		0.11	0.12	0.06	1.6	5.6
	CV		12	11	12	62	52
Tributary 2	mean	7	0.86	1.09	0.64	2.9	10.5
	SD		0.19	0.36	0.14	2.3	4.8
	CV		22	33	22	79	46

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Table III: Stepwise multiple regression on ecosystem properties (response variable) against predictive physical variables. Significance level of single factors: *p <0.05, **p <0.01, ***p <0.001.

Response variable	R ²	Predictive variables	Relative importance %	p-value
SR	0.22	Max SBT Q	56	*
		Cum. sediment vol.	44	***
FPOM	0.37	Distance from SBT outlet	95	***
		Cum. sediment vol.	5	*
CPOM	0.15	Distance from SBT outlet	55	**
		Max SBT Q	44	**
d50	0.19	Cum. sediment vol.	100	***
Periphyton Biomass	0.43	Distance from SBT outlet	8	*
		Max SBT Q	47	***
		Cum. sediment vol.	44	**
Periphyton Chlorophyll a	0.46	Max SBT Q	100	***
Macroinvertebrate Density	0.34	Max SBT Q	100	***
Macroinvertebrate Richness	0.25	Distance from SBT outlet	10	*
		Max SBT Q	90	***

1 **Figures**

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3 Figure 1: Map of the study stretch below Solis reservoir showing the study sites. SBT =
4 sediment bypass tunnel.

Figure 2: Inflow of Solis reservoir and discharge of the Albula River in 2014-2015 below Solis reservoir. Upper panel shows inflow of the reservoir, middle panel shows discharge from the Sediment Bypass Tunnel (SBT) and Spillwater + Bottom Outlet (Flushing Q). Lower panel shows total discharge (Total Q) through the canyon and sampling dates (triangles).

Figure 3: Scatterplot of the principal components analysis. The first two axes explained 61.4% of the variation among sites and dates. Normal data ellipses were generated with a standard probability of 0.69.

Figure 4: Temporal changes in sediment respiration (SR) per kg sediment < 8 mm ($\text{mg O}_2/\text{kg sed h}^{-1}$) (A, B), fine particulate organic matter (FPOM, g AFDM/kg sed) (C, D), coarse particulate organic matter (CPOM, g AFDM/kg sed) (E, F) and median grain size (d50, mm) (G, H). All values are means $\pm$ SE. Arrows represent the date of each SBT event.

Figure 5: Temporal changes in periphyton biomass (g AFDM/m²) (A, B) and chlorophyll-a (mg/m²) (C, D), and macroinvertebrate density (individuals/m²) (E, F) and richness (number of taxa/m²) (G, H). All values are means $\pm$ SE. Arrows represent the date of each SBT event.

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Figure 6: Non-metric multidimensional scaling (NMDS) plots based on benthic macroinvertebrate densities. Plots showing spatial variation among sites (A and B, all dates combined) and temporal variation among dates (C and D, all sites combined) by year. Legend in B applies for panel A, and legend in D applies for panel C. Dotted arrows in panel C and D illustrate temporal changes.