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1 **Effects of sulfonamide antibiotics on aquatic microbial community composition and functions**

2

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22 **HIGHLIGHTS**

- 23 • Periphyton slowed down microbial leaf-litter decomposition
- 24 • Sulfonamide antibiotics did not affect decomposition rates
- 25 • Sulfonamides reduced bacterial biomass accrual in leaves
- 26 • Sulfonamides stimulated β -glucosidase activity in leaves
- 27 • Periphyton did not damp the impact of sulfonamides on decomposers

28

29 **ABSTRACT**

30 Knowledge on interactions among microbial communities colonizing various streambed substrata
31 (e.g. cobbles, sediment, leaf-litter etc.) is essential when investigating the functioning of stream
32 ecosystems. However, these interactions are often forgotten when assessing the responses of
33 aquatic microbial communities to chemical contamination. Using a stream microcosm approach, the
34 respective impact of two sulfonamide antibiotics (sulfamethoxazole and sulfamethazine) on the
35 ability of microbial heterotrophs to decompose alder leaves was investigated in the presence or
36 absence of periphyton. Our hypothesis suggested that sulfonamides would negatively impair
37 microbial litter decomposition and that periphyton could possibly alleviate this effect by stimulating
38 microbial decomposer activity through a priming effect. Results showed that the presence of
39 periphyton enriched water with oxygen and labile dissolved organic carbon forms. However, these
40 labile organic carbon sources did not stimulate leaf-litter decomposition but mostly decoupled
41 microbial decomposer activity from particulate organic matter to dissolved organic matter through
42 negative priming. Also, the two sulfonamide molecules did not affect the leaf-litter decomposition

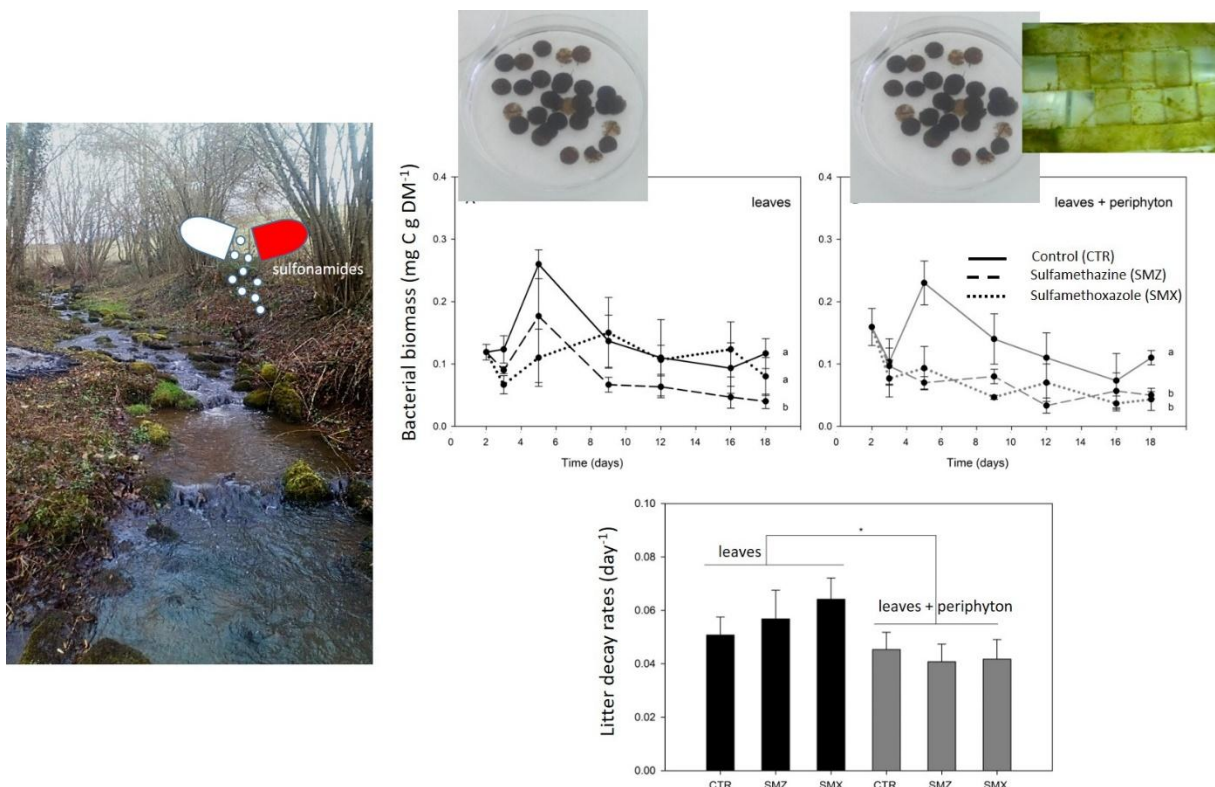
1 process but significantly decreased bacterial biomass accrual on leaves. The reduction of bacteria
2 was concomitant with an increase in biomass-specific β -glucosidase activity and this was attributed
3 to a stress response from bacteria to sulfonamides. Further research looking at microbial interactions
4 would provide for better assessment of chemical contamination effects in communities and
5 processes in stream ecosystems.

6

7 **Keywords:** leaf-litter decomposition, sulfamethazine, sulfamethoxazole, periphyton, negative
8 priming, β -glucosidase activity.

9

10 GRAPHICAL ABSTRACT



11

12 1. INTRODUCTION

13 Microbial communities are pivotal in stream ecosystem biogeochemical cycles and actively
14 participate in organic matter degradation. The leaf-litter decomposition process is characteristic of
15 forested streams and has been proposed as a tool to assess stream ecosystem integrity (Young *et al.*,
16 2008; Gessner *et al.*, 2010) though its application is rare. Leaf-litter decomposers are mainly
17 represented by fungi and bacteria, the fungal compartment being predominant and representing on
18 average 96% of the total microbial biomass accrual in leaves (Baldy *et al.*, 2002). However, their
19 relative importance in the degradation process is more balanced. For alder leaves (*Alnus glutinosa*), it
20 has been estimated that fungi contributed to 15% and bacteria to 7% of total leaf mass loss (Hieber
21 and Gessner, 2002), the remaining leaf mass being degraded by the action of macrofauna and/or
22 abiotic processes. Microbial leaf decay is mediated by specific extracellular enzymes excreted by
23 fungi and bacteria (Romani *et al.*, 2012) and the activity of these enzymes is regulated depending on

1 the quality and quantity of the dissolved organic matter available in stream water (Sinsabaugh and
2 Follstad Shah, 2012).

3 Under specific environmental conditions, leaf-litter decomposition in forested streams can be
4 modulated by primary production according to the priming effect, defined as the increase in
5 recalcitrant organic matter mineralization rate in the presence of labile organic matter (Guenet *et al.*,
6 2010). Some complex macromolecules present in leaves (i. e. phenolics) may be recalcitrant to
7 mineralization because microorganisms are unable to synthesize the enzymes that would degrade
8 them. However, the capabilities of leaf-associated microorganisms to increase the mineralization
9 rate of these complex molecules can be enhanced if extra energy is supplied as labile organic matter
10 (i. e. soluble carbohydrates). Primary producers in streams are mostly accumulated in the
11 periphyton, a type of microbial community colonizing inert substrates and being characterized by a
12 predominant autotrophic metabolism capable of releasing C-labile exudates to stream water (Lock *et al.*,
13 1984; Romaní & Sabater, 2000). Photosynthetic extracellular release has been estimated at
14 between 0.9 and 48% of primary production, greater release being measured in lentic sections of
15 rivers (e.g. side arms, Sieczko *et al.*, 2015) and under oligotrophic conditions where nutrient-limited
16 algae are forced to increase their release of carbon exudates due to stoichiometric constraints
17 (Ziegler and Lyon 2010). For microbial decomposers, this labile organic carbon supply from
18 periphyton has been shown to stimulate leaf-litter degradation (Daufresne and Loreau, 2001) but the
19 intensity of the priming effect may be modified by shifts in environmental conditions (Guenet *et al.*,
20 2010). Nutrient availability has been seen to influence the nature of the interactions between
21 decomposers of labile and recalcitrant organic matter (bacteria and fungi, respectively) and their
22 capacities to degrade leaf-litter (Carney *et al.*, 2007). Low nutrient concentrations in stream water
23 have been seen to increase leaf-litter decomposition in the presence of benthic algae but not in their
24 absence, while this priming effect was not observed under high-nutrient conditions (Danger *et al.*,
25 2013). Also, low quality litter species have been described as less palatable for leaf-litter decomposer
26 which shifts their activity from particulate organic matter (POM) degradation to dissolved organic
27 matter (DOM) degradation (Pastor *et al.*, 2014). Also, a recent study describes the influence of leaf-
28 litter on the primary producer community, i.e. changing microalgae taxonomic composition (Allen *et al.*,
29 2020). Since considering microbial interactions becomes central when investigating the
30 functioning of stream ecosystems, this study investigates how the priming effect can temper adverse
31 effects generated by antibiotics in leaf-litter decomposer communities for the first time.

32 Antibiotics are used worldwide for human and animal health. When delivered, antibiotics are
33 far from being completely absorbed by organisms. For antibiotics used in animal husbandry, between
34 30% and 90% of the parent compound is excreted unmetabolized in animal urine and feces (Sarmah
35 *et al.*, 2006). Due to this incomplete absorption, antibiotics are likely to reach the environment
36 mainly through manure and wastewater (Boxall *et al.*, 2003; Kümmerer, 2009). Sulfonamides (SMs)
37 form a class of antibiotics with a common mode of action. They inhibit multiplication of bacteria by
38 inhibiting folic acid production and therefore the synthesis of purines and DNA (Seydel, 1968). SMs
39 are mainly used for animal health. In France, in 2018, they represented 17.8% of antibiotic sales for
40 veterinary use (Urban *et al.*, 2019). SMs have low sorption properties in soils (Boxall *et al.*, 2002;
41 Thiele-Bruhn *et al.*, 2004) and high water solubilities (Sukul & Spiteller, 2006). Several publications
42 attempted to assess their degradation in the environment. In soils, half-lives range between 10 and
43 30 days (García-Galán *et al.*, 2008). The degradation of three different SMs was studied in lake water,
44 where half-lives varied between 10.5 and 12.9 days (Zhang *et al.*, 2013). Their high mobility in the
45 environment and their relative persistence are confirmed by many studies highlighting their
46 occurrence in ground and surface waters worldwide (Batt *et al.*, 2006; Peng *et al.*, 2008; Matongo *et al.*,
47 2015). In a wide study conducted in the USA, Kolpin *et al.* (2002) examined the occurrence of

1 pharmaceuticals in 139 streams and rivers susceptible to contamination (biased selection). They
2 found that sulfamethazine (SMZ) and sulfamethoxazole (SMX) were detected at the respective
3 frequencies of 4.8% and 12.5% and respective median concentrations of 0.02 $\mu\text{g L}^{-1}$ and 0.15 $\mu\text{g L}^{-1}$
4 (maximum concentrations: 0.12 $\mu\text{g L}^{-1}$ and 1.9 $\mu\text{g L}^{-1}$ respectively). Sulfonamides have been detected
5 worldwide, their concentrations in surface waters generally varying between a few and several dozen
6 ng L^{-1} (Martin-Laurent *et al.*, 2019).

7 The impact of SMs was mainly studied on microbial communities from activated sludge.
8 Changes in bacterial community structure were noticed but no significant modification was observed
9 in organic matter microbial degradation (Xia *et al.*, 2012; Collado *et al.*, 2013). However, in soils, at
10 environmentally realistic concentrations, it has been shown that SMs inhibit microbial respiration
11 and possibly reduce specific enzymatic activities like phosphatase or urease (Liu *et al.*, 2009;
12 Gutierrez *et al.*, 2010; Conkle & White, 2012; Roose-Amsaleg & Laverman, 2016). This reduction in
13 microbial activity could negatively impact nutrient renewal and potentially alter biogeochemical
14 cycles in soils. Also, when exposed to SMs, microbial communities in soils show a reduction in the
15 bacteria/fungi biomass ratio (Gutierrez *et al.*, 2010; Pinna *et al.*, 2012). To our knowledge, no studies
16 have attempted to investigate the impact of SMs on natural leaf-litter decomposer communities in
17 streams. Therefore, the objective of this study is to assess the impact of two SM antibiotics
18 (sulfamethazine and sulfamethoxazole) on leaf-litter decomposers, and more precisely, to determine
19 whether the presence of periphyton could modulate the response of litter decomposers to the
20 antibiotics or not. To that end, a microcosm experiment was conducted on previously-colonized alder
21 leaves which were exposed (or not) to an antibiotic in the presence (or in absence) of periphyton.
22 Physicochemical characteristics of water and the biomass and functional responses of leaf-associated
23 microbial communities were monitored over three weeks. Thus, this study aims to answer the
24 following specific questions: i) Does the presence of periphyton increase the activity of leaf-litter
25 decomposers? ii) What are the specific impacts of SMs on the leaf-litter decomposer community? iii)
26 Can the impact of antibiotics on leaf-litter decomposers be alleviated by the presence of periphyton?

27 Our hypotheses suggest that primary production in periphyton would release labile DOM in
28 water. These labile carbon forms could increase the activity of both labile and recalcitrant organic
29 matter decomposers in leaf-litter through a priming effect. Due to their bacteriostatic activity, SM
30 antibiotics are expected to negatively impact bacterial biomass as well as the decomposition of labile
31 organic matter through decreases in the activity of glucosidases. However, one could expect fungi to
32 take advantage of impaired bacteria by increasing their biomass and their capacity to degrade both
33 labile and complex carbon polymers available in water and of which alder leaves are made up.
34 Finally, the greater availability of labile DOM could enhance the resistance of bacterial populations in
35 leaves to SM antibiotics.

36

37 **2. MATERIALS AND METHODS**

38

39 **2.1. Microbial communities**

40 Two different substrates were used to grow natural microbial communities: (1) glass slides (7.6 \times 2.6
41 cm) glued onto three independent flagstones to collect periphyton and (2) dried freshly-fallen alder
42 leaves (*Alnus glutinosa* L. (Gaertn.)) divided into three nylon bags of 0.5 mm mesh size to collect leaf-
43 litter decomposer communities. Both the glass slides and the leaves were incubated for one week in
44 spring 2019 in a first-order forested stream (Artière, Centre France; 45° 43' N, 3° 01' E) before being
45 transferred to microcosms in the laboratory. Aquatic microorganisms can colonize, within a few days,

1 deciduous leaves submerged in streams (*i. e.* aquatic hyphomycetes; Nikolcheva and Bärlocher,
 2 2004). The 500-meter section in which the microbial communities were grown was characterized by
 3 a stream channel 2-4 m wide and 10-20 cm deep. The streambed alternated sand pools and riffles
 4 with cobbles of granitic nature and the riparian vegetation was mainly composed of alder and
 5 chestnut with a nearly absent shrubby stratum. This section is located within a “zero pesticide area”
 6 which means that the use of phytosanitary products is forbidden. Indeed, levels of contamination
 7 were low for pesticides and pharmaceuticals at this site (Rossi *et al.*, 2019). More specifically, no SMs
 8 were detected in the stream water at this study site.

9 2.2. Microcosm study

10 The microcosms used consisted of 20 L glass aquariums (rectangular parallelepiped length 40
 11 cm × width 20 cm × height 25 cm). Previously-colonized alder leaves were cut into 1 cm² disks and
 12 placed in nylon bags (0.5 mm mesh) before being divided into 18 independent microcosms. Each
 13 microcosm contained 10 litter bags and each bag contained 35 leaf disks. Among these 18
 14 microcosms containing litter bags, 9 were filled each with 12 glass slides colonized with periphyton.
 15 Since early life forms of macrofauna could be still contained in periphyton (*i. e.* eggs, larvae)
 16 deployed in aquariums, we decided to enclose all leaf disks in fine mesh bags in order to prevent
 17 their decomposing activity.

18 Six different treatments were tested each with three replicates in our experiment (Table 1).
 19 Leaves alone (L) and leaves incubated in the presence of periphyton (L+P) exposed to sulfamethazine
 20 (SMZ), sulfamethoxazole (SMX) or without antibiotic (CTR). A nominal concentration of 5 µg L⁻¹ in
 21 water was chosen for the antibiotics. These nominal concentrations were based on SMZ and SMX
 22 concentrations detected in agricultural and urban streams worldwide, which can reach a few µg L⁻¹
 23 during acute exposure scenarios (Matongo *et al.*, 2015, Peng *et al.*, 2008).

ANTIBIOTICS BIOFILMS	CTR	SMZ 5 µg L ⁻¹	SMX 5 µg L ⁻¹
L	a	b	c
L + P	d	e	f

24
 25 **Table 1.** Experimental design. Six treatments were tested with leaves alone a) without
 26 contamination, b) with SMZ in water at 5 µg L⁻¹, c) with SMX in water at 5 µg L⁻¹, or with leaves and
 27 periphyton d) without contamination, e) with SMZ in water at 5 µg L⁻¹, f) with SMX in water at 5 µg L⁻¹.
 28
 29

30 The water in the microcosms consisted in 12 L of dechlorinated tap water plus 3 L of natural stream
 31 water taken fresh from the study site in the Artièrre. The tap water was filtrated using a BRITA PURITY
 32 C Quell ST (Brita, Germany). Every week, the water was fully replaced in all the microcosms to ensure
 33 supply of nutrients and microbial species succession during leaves degradation. Antibiotics were
 34 reintroduced at their nominal concentration (5 µg L⁻¹) into each microcosm at the same time the
 35 water was replaced. Flow velocity was set at 6.6 L min⁻¹ using NEWA Jet NJ1200 submersible pumps
 36 (Newa, Italy) and the temperature maintained at 20 °C, whereas the light regime followed the

1 natural light (1054 ± 122 Lux) dark (0 Lux) cycles. The experiment was conducted for two weeks with,
2 in total, seven sampling days (d): d0, d1, d3, d7, d10, d14, and d16.

3

4 **2.3. Physical and chemical characteristics of water and leaves**

5 pH (Accumet AE150, Fisher Scientific, USA), water temperature (ProODO, YSI, USA) and light (Testo
6 545, Testo AG, Germany) were measured (Table S1) on each day of sampling. Dissolved oxygen
7 concentration in the water was measured each hour between 9:00 a.m. to 6:00 p.m. in each of the
8 18 microcosms over four sampling days (d1, d7, d10, and d14) with a Pro ODO (YSI, USA) optical
9 oximeter. The average of the dissolved oxygen measures between 12:00 p.m. and 4:00 p.m.
10 (maximum irradiation, Young and Huryn 1999) in each microcosm was used to calculate the
11 differences between L and L+P exposure treatments (Table 2).

12 Dissolved nutrient concentrations in the water were assessed on filtered (0.45 μm , cellulose
13 acetate, Dutscher) samples, in order to remove the particulate fraction. Dissolved organic carbon
14 (DOC) was determined by the combustion catalytic oxidation method and total dissolved nitrogen
15 (TN) by chemiluminescence (TOC VCPN Analyzer, Shimadzu, Japan). Total dissolved phosphorus (TP)
16 concentration was measured by the molybdenum-blue colorimetric method (Murphy & Riley, 1962)
17 after basic digestion of water samples (Grasshoff *et al.*, 1983).

18 The SUVA_{254} index was used to assess the quality of dissolved organic matter (DOM). This index is
19 defined as the ratio between the absorption of a water sample at 254 nm and its corresponding
20 concentration of dissolved organic carbon (Weishaar *et al.*, 2003). SUVA_{254} is positively correlated
21 with the aromaticity of dissolved organic matter. A high SUVA_{254} may indicate the predominance of
22 terrestrially-derived DOM that is commonly more recalcitrant than autochthonous-derived DOM
23 (more labile). Measurements were performed on filtered water samples (0.45 μm) with a UV
24 spectrophotometer (Ultrospec 2000, Pharmacia Biotech, United-Kingdom) and combined with the
25 DOC results described above.

26 For antibiotic concentrations in water, samples (100 mL) were taken from microcosms thirty
27 minutes before and after each water renewal. After adjusting the pH to 4 and adding an internal
28 standard (50 μL of a 500 $\mu\text{g L}^{-1}$ SMX-d₄ solution), samples were concentrated by a factor 200 on Oasis
29 HLB-200mg cartridges (Waters™) according to the manufacturer's recommendations (elution
30 solvent: methanol). The antibiotic concentrations were determined by LC/ESI-MS on a Thermo
31 Scientific UHPLC Ultimate 3000 RSLC coupled with an Orbitrap Q-Exactive analyzer. The analyses
32 were carried out in positive mode. The UHPLC was equipped with a Luna Omega Polar C18 column;
33 100 x 2.1 mm; 1.6 μm (Phenomenex) at 30 °C with acetonitrile gradient + 0.1% formic acid (Solvent
34 A) and water + 0.1 % formic acid (Solvent B): 0–2.5 min: 30–64.5 % A (linear); 2.5– 2.6 min: 64.5–99 %
35 A (linear); 2.6–5 min: 99 % A; 5–5.1 min: 99–30 % A; 5.1–8 min: 30 % A. Flow: 0.45 mL min⁻¹. For the
36 mass spectrometer, gaseous N₂ was used as nebulizer gas (50 L h⁻¹). The spray voltage was 3.0 kV.

37 Nutrient content in leaves (including its associated decomposers community) was
38 determined at each sampling time and treatment in triplicate. Three leaf disks were mixed with 1.3
39 mL of deionized water and placed in previously-rinsed (deionized water x 2) Lysing Matrix E tubes
40 (MP Biomedicals, USA). Samples were then homogenized with a Precellys 24 tissue homogenizer
41 (three times for 30s, Bertin Technologies, France). After centrifugation of the homogenate (30s at
42 17000 g), the supernatant was collected and separated into two different aliquots. The first aliquot
43 (diluted 1:50) was used to determine total carbon and nitrogen content following the same method
44 described above for water (TOC VCPN Analyzer, Shimadzu, Japan). Total phosphorus content was

1 assessed using the second aliquot (diluted 1:16) following the molybdenum-blue colorimetric
2 method previously described. Eight blanks containing only deionized water were run to assess
3 carbon, nitrogen and phosphorus contaminations in Lysing Matrix E tubes. Results were expressed as
4 C:nutrient molar ratios.

5 **2.4. Microbial biomass and diversity in leaves**

6 Fungal biomass was estimated through ergosterol concentration determination. Extraction and
7 determination of ergosterol concentration were performed on fourteen alder leaf-disks per sample
8 following the protocol described by Artigas *et al.* (2012). Finally, ergosterol concentrations were
9 converted into fungal carbon using conversion factors described by Gessner and Chauvet (1993) and
10 Baldy *et al.* (1995). The ergosterol results were therefore expressed in mg fungal carbon per g leaf
11 dry mass (DM)⁻¹.

12 Bacterial biomass was assessed through bacterial density analyses using flow cytometry
13 (Borrel *et al.*, 2012). At each sampling time, one leaf disk per replicate was placed in sterile Ringer's
14 solution and immediately fixed with paraformaldehyde (2% final concentration). Before cytometric
15 counts, samples were incubated with sodium pyrophosphate (10 mM final concentration) and
16 sonicated twice 30 s + 30 s to improve cell disaggregation (Artigas *et al.*, 2008). Later, the bacterial
17 suspension was centrifuged for one minute at 800g in order to pellet large debris and the
18 supernatant was diluted 100-fold in TE buffer (10 mM Tris, 1 mM EDTA). Finally, bacterial cells were
19 stained with SYBR GREEN (1X, Molecular Probes, USA) and counted with a BD FACSCalibur (15 mW at
20 488 nm, Becton Dickinson, USA). Bacterial density results were converted into mg of bacterial carbon
21 per g of DM using conversion factors described in Theil-Nielsen & Sondergaard (1998) and Bratbak &
22 Dundas (1984).

23 Fungal and bacterial species' richness was analyzed by Polymerase Chain Reaction and
24 Denaturing Gradient Gel Electrophoresis (PCR-DGGE) analysis. DNA was extracted from leaves at
25 days 0, 1, 3, 7, and 14 using the FastDNA 2 SPIN Kit for soil (MP Biomedicals, Santa Ana, California,
26 USA) following the manufacturer's instructions. Extracted DNA was amplified targeting the regions i)
27 Internal Transcribed Spacer (ITS) of the fungal rRNA and ii) the V6-V8 regions of bacterial 16S rDNA
28 according to the primers and conditions described in Crouzet *et al.* (2010). DGGE was run in a TIGEK
29 2401-220 device (CBS Scientific, San Diego, North Carolina, USA) in 1X TAE buffer for 16 h, at a
30 temperature of 59 °C and a voltage of 90 V. Denaturing gradient urea conditions were 20-50% for
31 fungi and 35-55% for bacteria. At the end of the electrophoresis, gels were stained in 1X TAE buffer
32 containing 1/20000 dilution of Gel star (Lonza, Rockland, USA) and digitized using a BioSpectrum AC
33 Imaging System (UVP, Upland, California, US). DGGE band (operational taxonomical unit, OTU)
34 profiles were analyzed using Gelcompar II software (Applied Maths, Belgium) in order to obtain the
35 fungal and bacterial OTU richness.

36 **2.5. Microbial degrading-activity in leaves**

37 At each sampling date, five leaf disks per experimental treatment were dried at 37 °C for 72h and
38 weighed with an analytical balance with 0.1 mg sensitivity (Precisa 80A 200M, Precisa, Switzerland).
39 The amount of mass loss over time in these disks was used to determine microbial decay rates in
40 alder leaves.

41 A total of five different extracellular enzymatic activities were assessed in leaf-associated
42 microbial communities using fluorogenic or chromogenic substrate analogs (Sigma-Aldrich, USA).β-
43 glucosidase (EC 3.2.1.21), cellobiohydrolase (EC 3.2.1.91) and phosphatase (EC 3.1.3.1–2) enzymatic
44 activities were measured using methylumbelliferyl (MUF) substrates (Sigma).Leucine-aminopeptidase

1 (EC 3.4.11.1) enzymatic activity was determined using L-leucine-4-methyl-7-coumarinylamide, a
2 substrate labelled with aminomethyl coumarin (AMC). Finally, phenol oxidase (EC 1.10.3) enzymatic
3 activity was assessed using 3,4-dihydroxy-L-phenylalanine (L-DOPA).

4 All enzymatic activity bioassays consisted in one leaf disk immersed in 1 mL of Ringer's media
5 supplemented with the corresponding substrate analog concentration expected to saturate
6 enzymatic activity (0.3 mM for β -glucosidase, phosphatase, leucine-aminopeptidase and 1.5 mM for
7 cellobiohydrolase, phenol oxidase) (Artigas *et al.*, 2008 and 2012). Samples were incubated for one
8 hour at 20 °C in the dark under agitation. After incubation, the enzymatic reaction was stopped by
9 adding glycine buffer (pH 10.4) and substrate turnover was determined fluorometrically (360/460 nm
10 excitation/emission) for MUF and AMC substrates with a Fluoroskan (Thermo Fisher Scientific, USA).
11 For phenol oxidase, the activity was determined spectrophotometrically at 460 nm (Ultrospec 2000,
12 Pharmacia Biotech, United-Kingdom). After enzymatic activity bioassays, disks were dried (37 °C for
13 72h) and weighed to correct enzymatic activities by the corresponding DM and further corrected by
14 the microbial biomass (fungal + bacterial) accumulated in the disk.

15 Dose-response experiments on β -glucosidase activity in alder leaves were run afterwards in
16 order to assess the interaction between the increasing concentrations of the two antibiotics (SMZ
17 and SMX) and that of glucose in the media. We selected glucose to assess to which extent β -
18 glucosidase activity rates decreased when increasing the concentration of an end product of β -
19 cellobiose hydrolysis (Desphande and Eriksson, 1988) in the presence of SMZ or SMX. Incubations
20 were performed in triplicate per each glucose and antibiotic concentration tested and run for 72h at
21 the same temperature and light conditions described for the microcosm experiment. After 72h, β -
22 glucosidase activity was determined and corrected by the corresponding bacterial cell density
23 present in the leaf disk (as described above). Dose-response curves were plotted for each of the
24 antibiotic concentrations tested and after expressing β -glucosidase activity values as the % response
25 against the treatment without glucose.

26

27 2.6. Data analyses

28 Differences in water (dissolved oxygen and organic carbon concentrations, SUVA₂₅₄, C:N and C:P
29 molar ratios) and leaf-litter (biomass of fungi and bacteria, C:N and C:P molar ratios) variables among
30 experimental treatments were assessed by two-way repeated measures ANOVA. Between
31 treatments, effects were tested for the following factors: incubation (I; including L and L+P levels),
32 the antibiotic treatment (A; including CTR, SMZ, SMX levels) as well as their interaction (I \times A). Within
33 treatments, effects were tested for the factor Time and the interaction between Time and the factors
34 described above (I, A, and I \times A).

35 Microbial decay rates of alder leaves were determined fitting an exponential decay model on disk
36 mass loss over the time of the experiment and using the following equation: $M_t = M_0 e^{-kt}$, where M_0 is
37 the initial DM (g), M_t is the DM (g) at time t and k is the decay rate coefficient (d^{-1}) (Petersen &
38 Cummins, 1974).

39 Accumulated enzymatic activity (AEA) was calculated for β -glucosidase and phenol oxidase activities
40 (both responding to I and A factors) by means of areas under the time curve calculations in each
41 microcosm using Origin 9.1 software (OriginLab, USA). To assess the efficiency of these enzymes in
42 degrading alder leaves, turnover activities were calculated according to the method described by
43 Mora-Gómez *et al.* (2016) with few modifications. Briefly, Turnover Activities (TA), i.e. $TA = 1/k$,
44 where k is the slope of ln ash free dry mass (AFDM) remaining regressed to accumulated enzyme
45 activities (AEA) (Simon *et al.*, 2009). However, leaf mass loss was not studied in terms of ash-free-dry-
46 mass (AFDM) but in terms of dry mass (DM) in our experiment. Higher TA values would mean low

1 enzyme efficiencies for alder leave decomposition while lower TA indicates the opposite. TA was
2 expressed as μmol of enzyme produced to decompose one gram of alder litter DM.

3 Integrated variables on microbial communities' degrading activity (leaf-litter decay rates, enzyme
4 activity AUC or TA) were tested for differences among I, A, and $I \times A$ factors using two-way ANOVA.
5 Post-hoc Tukey's test was used to determine differences among levels within the factor A (CTR, SMZ,
6 SMX) as well as on dose-response curves between antibiotics and glucose.

7 All ANOVA tests were run on response variables having equality of variances and normality. The
8 different statistical tests were run with SPSS Statistics Ver. 17.0 (IBM, USA).

9

10 **3. RESULTS**

11

12 **3.1. Impact of periphyton on leaf-litter decomposers**

13 Microcosms containing leaf-litter and periphyton had greater average dissolved oxygen
14 concentrations in water than microcosms containing leaf-litter alone ($P < 0.005$, Table 2 and S2). The
15 moderate increase in dissolved oxygen concentrations was accompanied by slightly higher dissolved
16 organic carbon (DOC) concentrations and lower DOM aromaticity indices ($P < 0.05$ and $P < 0.005$ for
17 DOC and SUVA_{254} , respectively; Tables 2 and S2) compared to L treatment, suggesting the presence
18 of photosynthetic extracellular release in L+P treatment. Carbon: nutrient molar ratios in water (C:N
19 and C:P) were in line with the enhanced C production by the periphyton since C:P and C:N were
20 greater in L+P than in L (Table 2), although results were only statistically significant for the C:P ratio
21 ($P < 0.05$ and $P = 0.098$, respectively; Table S2). Average water C:P ratio (days 1 to 16) was 383.9 ± 9.8
22 in L and 614.6 ± 54.5 in L+P. Water N:P molar ratio was also higher in L+P treatments (days 7, 14 and
23 16, Table 2) indicating greater availability of dissolved N over P molecules for leaf-litter decomposers.

24 The microbial decomposition of alder leaves was lower in L+P ($0.042 \pm 0.004 \text{ day}^{-1}$) compared to L
25 ($0.057 \pm 0.005 \text{ day}^{-1}$) treatments (Figure 1). In average, the microbial decomposition of leaves was
26 reduced by 25% in L+P compared to L treatments.

1

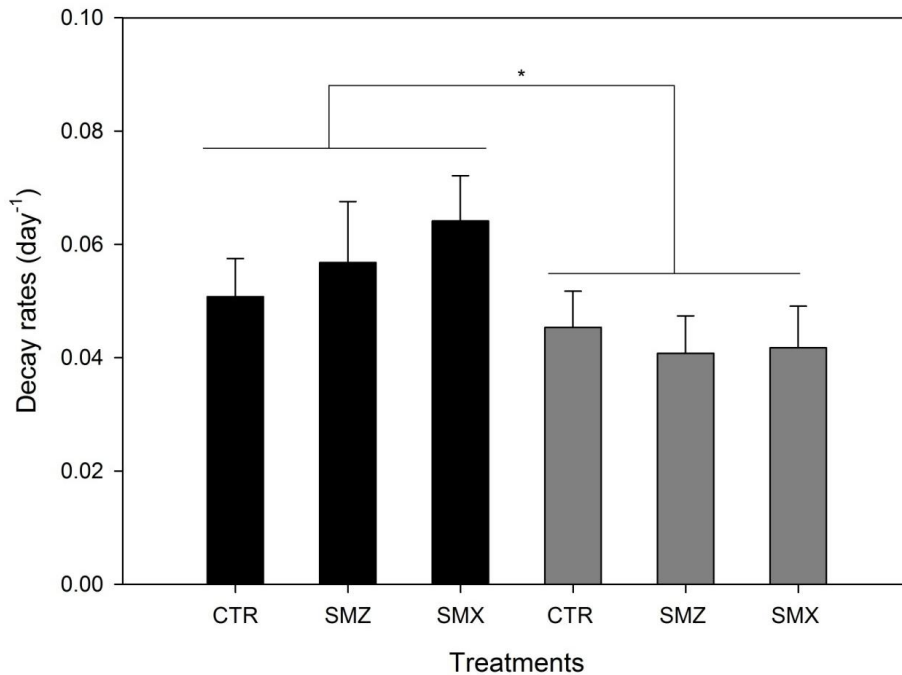
2 Table 2. Comparison of water characteristics (including dissolved oxygen (DO) and organic carbon concentrations, SUVA₂₅₄ index and C:N and C:P molar
 3 ratios) between aquariums with leaves alone (L) and in the presence of periphyton (L+P). Oxygen₁₂₋₁₆ values correspond to the average of hourly dissolved
 4 oxygen measures taken in L and L+P microcosms (including controls and contaminated microcosms) between 12h and 16h. Values are means (n=9, including
 5 CTR, SMZ and SMX treatments) and standard errors (*in italic*) in L and L+P aquariums at days 1, 7, 10 and 14 of the experiment. Asterisks represent
 6 significant differences between L and L+P (Tukey's test, $P < 0.05$). n.a. means not analyzed.

7

	Day 1		Day 3		Day 7		Day 10		Day 14		Day 16	
	L	L+P	L	L+P	L	L+P	L	L+P	L	L+P	L	L+P
DO (mg L ⁻¹)	8.21 <i>0.01</i>	8.34 <i>0.02</i>	n.a.	n.a.	8.13 <i>0.01</i>	8.29* <i>0.01</i>	8.18 <i>0.01</i>	8.50* <i>0.05</i>	8.23 <i>0.01</i>	8.71* <i>0.08</i>	n.a.	n.a.
DOC (mg L ⁻¹)	0.33 <i>0.06</i>	0.63 <i>0.15</i>	0.44 <i>0.10</i>	1.08* <i>0.13</i>	0.69 <i>0.06</i>	0.93* <i>0.12</i>	0.48 <i>0.09</i>	1.64* <i>0.43</i>	0.66 <i>0.06</i>	1.21* <i>0.16</i>	0.55 <i>0.03</i>	1.68* <i>1</i>
SUVA ₂₅₄ (L mg ⁻¹ m ⁻¹)	14.74 <i>2.57</i>	10.24 <i>1.85</i>	14.03 <i>4.67</i>	4.22* <i>0.74</i>	8.67 <i>1.43</i>	6.01* <i>0.64</i>	8.84 <i>2.25</i>	3.96* <i>0.62</i>	8.14 <i>1.10</i>	5.19* <i>0.54</i>	11.37 <i>1.90</i>	8.04* <i>1.91</i>
C:N _{water}	16.07 <i>0.14</i>	16.36 <i>0.92</i>	15.45 <i>0.17</i>	15.89 <i>0.64</i>	14.34 <i>0.13</i>	14.25 <i>0.49</i>	14.32 <i>0.70</i>	17.07 <i>1.67</i>	14.96 <i>0.11</i>	21.66 <i>3.25</i>	16.88 <i>0.16</i>	23.25* <i>3.87</i>
C:P _{water}	370.5 <i>11.4</i>	433.7 <i>61.1</i>	380.5 <i>15.1</i>	427.2 <i>45.9</i>	464.4 <i>26.8</i>	564.3* <i>42.6</i>	373.2 <i>30.2</i>	438.3 <i>29.2</i>	338.5 <i>12.9</i>	954.3* <i>222.7</i>	383.3 <i>26.5</i>	869.7* <i>156.7</i>
N:P _{water}	26.92 <i>0.88</i>	30.11 <i>2.09</i>	28.69 <i>0.96</i>	30.90 <i>2.21</i>	37.63 <i>1.86</i>	46.04* <i>3.01</i>	30.93 <i>2.68</i>	30.88 <i>2.20</i>	26.43 <i>1.09</i>	42.60* <i>5.05</i>	26.51 <i>1.86</i>	43.66* <i>5.12</i>

8

9



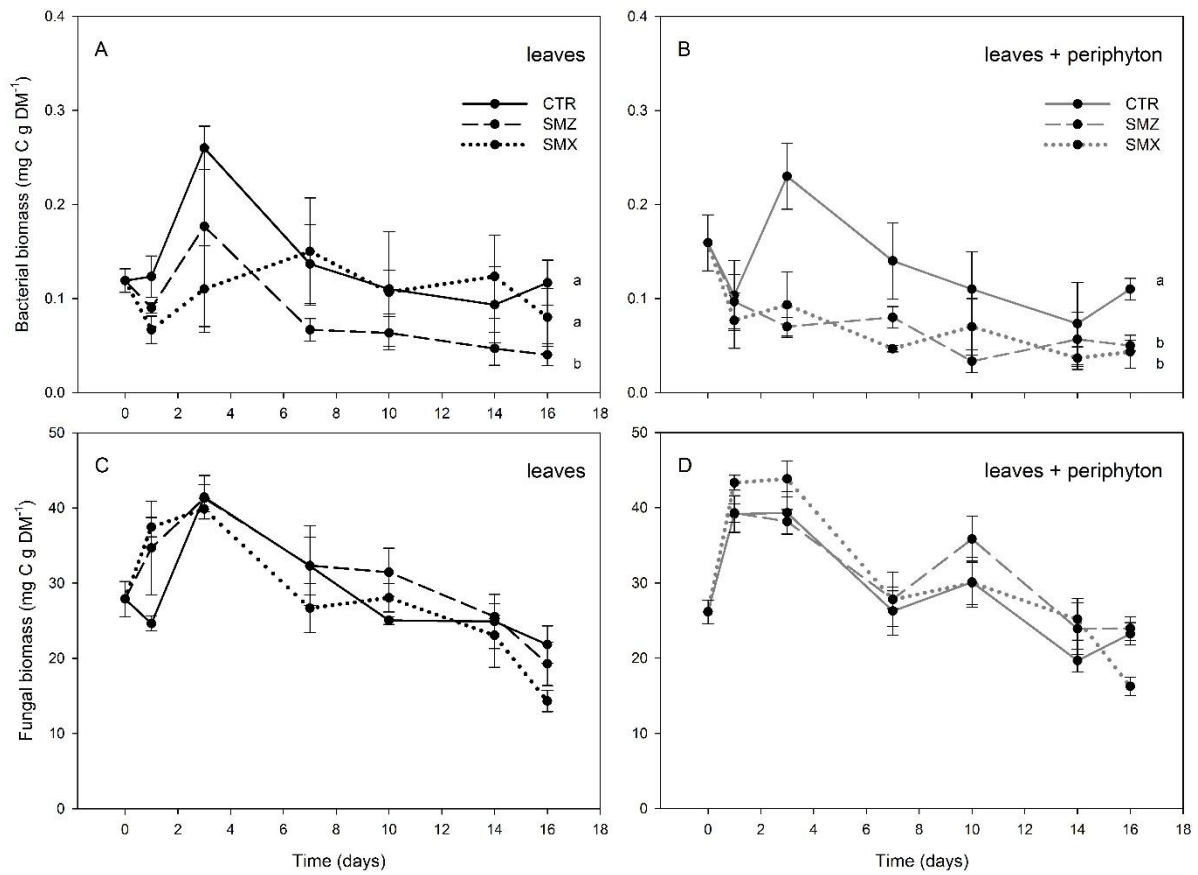
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2 Figure 1. Decomposition rates of alder leaves during the microcosm experiment. Control (CTR),
 3 sulfamethazine (SMZ) and sulfamethoxazole (SMX) treatments for leaves exposed alone (black bars)
 4 or in the presence of periphyton (grey bars). Values correspond to means (n=3) and standard errors.
 5 Significant differences between tested treatments are represented by an asterisk (ANOVA test, $P <$
 6 0.05).

7

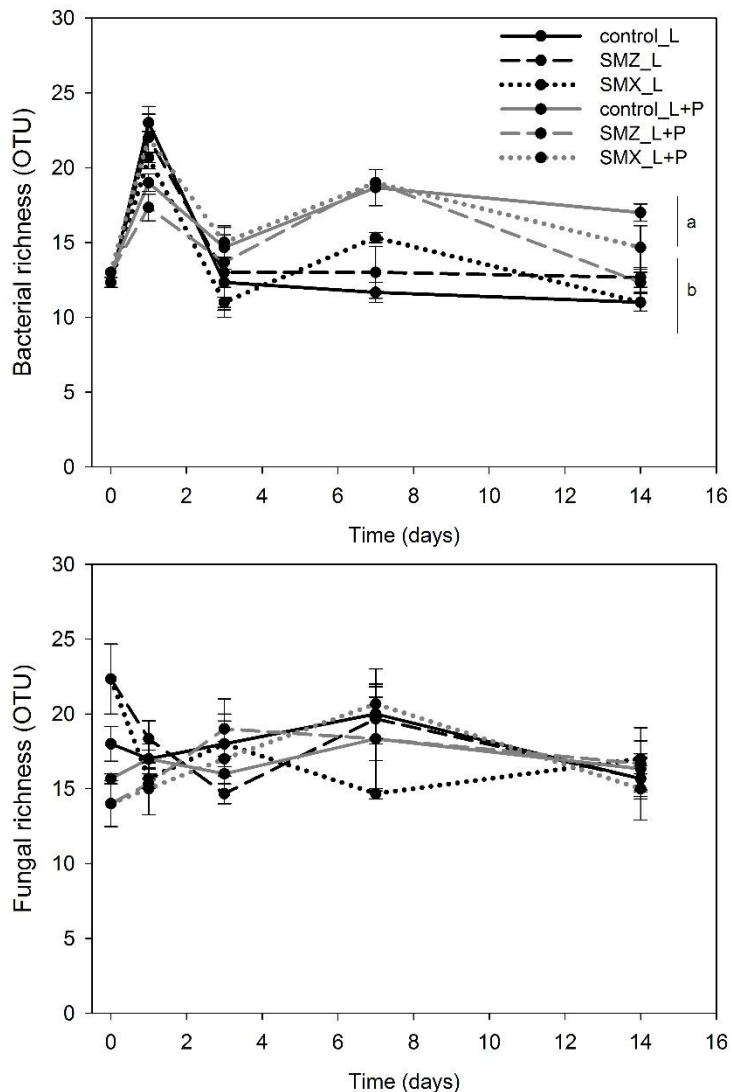
8 The extra labile C available in L+P treatments was not allocated to the growth of leaf-litter
 9 decomposers. Fungal biomass accrual values in leaves were similar between L (29.1 ± 1.2 mg fungal C
 10 g DM^{-1} , average days 1 to 16 including control, SMZ and SMX treatments) and L+P (30.7 ± 1.2 mg
 11 fungal C g DM^{-1}) treatments ($P = 0.181$, Figures 2C and 2D) and that of bacteria were moderately
 12 higher in L (0.109 ± 0.011 mg bacterial C g DM^{-1}) than in L+P (0.084 ± 0.008 mg bacterial C g DM^{-1}) (P
 13 < 0.05 , Figures 2A and 2B). Among the five different extracellular enzymatic activities measured (β -
 14 glucosidase, cellobiohydrolase, phenol oxidase, alkaline phosphatase and leucine-aminopeptidase),
 15 only the phenol oxidase activity calculated as the Area Under the time Curve (AUC) was influenced by
 16 the treatment (Table S2, Figure S1). Phenol oxidase was higher in L treatment (617 ± 68 mmolDIQC g
 17 microbial C⁻¹) than in L+P treatment (480 ± 30 mmolDIQC g microbial C⁻¹) evidencing faster
 18 breakdown of recalcitrant polymers per unit of microbial biomass in L, since phenol oxidase is
 19 responsible for the degradation of lignin polymers from leaves.

20 The average species richness of fungi (15.2 ± 0.6 OTUs) and bacteria (17.2 ± 0.5 OTUs) in leaves were
 21 relatively similar when considering together all treatments and times (Figure 3). However, the
 22 species richness of bacteria tended to be greater in L+P (16.0 ± 0.6 OTUs, average day 3 to 16) than in
 23 L (12.3 ± 0.4 OTUs) from day 3 and until the end of experiment. During the same time period, fungal
 24 species richness remained similar between L+P (17.5 ± 0.6 OTUs) and L (17.0 ± 0.6 OTUs) treatments.



1

2 Figure 2. Bacterial (plots A, B) and fungal (plots C, D) biomass accretion in alder leaves during the 16-
3 day experiment. The three experimental treatments (control, SMZ, SMX) are represented for leaves
4 incubated alone (left plots) and in the presence of periphyton (right plots). Values are means (n=3)
5 and standard errors for each sampling time, treatment and exposure condition. Letters indicate
6 statistically significant differences between treatments including the whole sampling dates (a>b,
7 Tukey's test, $P < 0.05$).



1

2 Figure 3. Band richness (as operational taxonomic unit, OTU) in leaf-associated bacterial and fungal
 3 communities during the experiment is represented. Controls, SMZ and SMX treatments are
 4 represented for L (black-colored line) and L+P (grey-colored line) treatments. Values are means (n=3)
 5 and standard errors per treatment and sampling time. Statistical differences between treatments are
 6 marked by letters (a > b, $P < 0.001$, repeated measures ANOVA test).

7

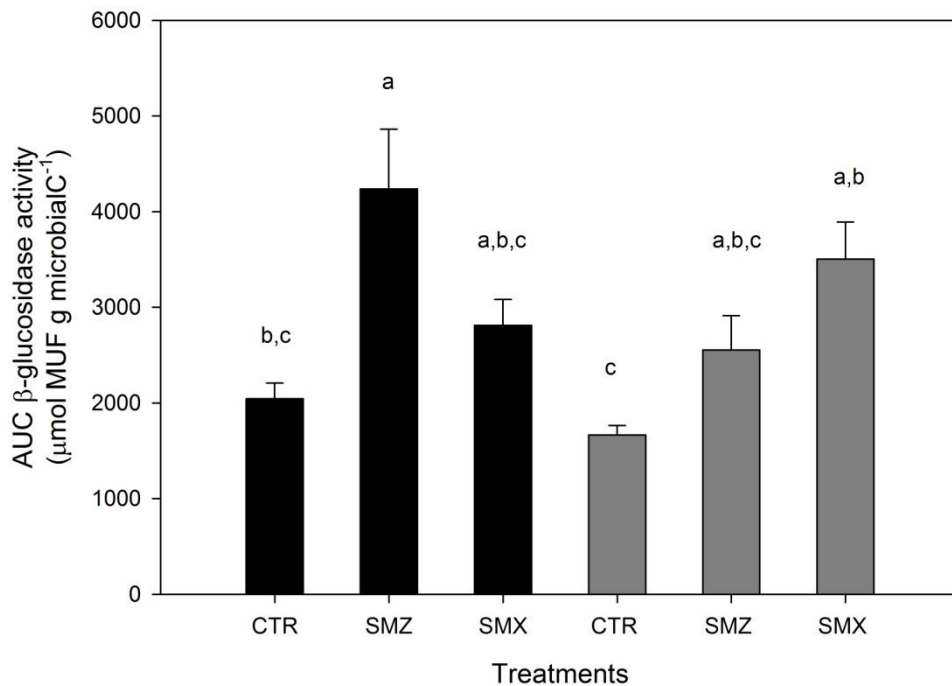
8 3.2. Impact of antibiotics on leaf-litter decomposers

9 Average SMZ and SMX concentrations in water from microcosms measured 30 minutes after weekly
 10 contamination at days 1 and 7 (5.1 ± 0.2 and $4.6 \pm 0.2 \mu\text{g L}^{-1}$, respectively) were relatively close to the
 11 expected nominal concentration of $5 \mu\text{g L}^{-1}$. Chemical monitoring in water from microcosms revealed
 12 different dissipation rates between SMZ and SMX. Dissipation percentages were calculated during
 13 the second week of the experiment (between days 7 and 14) and resulted in almost double the
 14 percentage in SMZ ($53.8 \pm 6.4\%$) compared to SMX ($34.8 \pm 8.0\%$) treatment (Table S3). The greater
 15 dissipation of SMZ coincided with the detection of the metabolite, 2-amino-4,6-dimethylpyrimidine

1 (ADMP). The lower dissipation percentage of SMX was the consequence of the low dissipation values
2 obtained in the treatment with leaves incubated in the presence of periphyton ($15.44 \pm 1.36 \%$)
3 compared to those incubated alone ($54.19 \pm 3.22 \%$).

4 Alder leaf-litter decomposition rates were neither affected by the exposure to antibiotics ($P = 0.798$)
5 nor by the interaction between antibiotics and the presence of periphyton ($P = 0.562$). Breakdown
6 rates in L and L+P averaged $0.045 \pm 0.004 \text{ day}^{-1}$ in control and $0.049 \pm 0.004 \text{ day}^{-1}$ in antibiotic
7 treatments (Figure 1). However, the antibiotics significantly reduced bacterial biomass accrual in
8 leaves between 3 and 7 days of exposure and onwards compared to controls ($P < 0.005$, Figure 2A
9 and B). While the effect of SMZ and SMX was similar in bacterial communities from L+P treatment
10 (Figure 2B), their effects varied in communities from the L treatment (Figure 2A). In the L+P
11 treatment, bacterial biomass accrual in leaves was 2.5 times lower in both SMZ and SMX treatments
12 ($0.058 \pm 0.006 \text{ mg bacterial C g DM}^{-1}$) compared to the control (0.143 ± 0.02 ; average days 1 to 16,
13 Figure 2B). In the L treatment, bacterial biomass accrual in leaves was reduced 2-fold by SMZ ($0.079 \pm$
14 $0.023 \text{ mg bacterial C g DM}^{-1}$, average days 1 to 16) whereas only a transitory reduction (2.5-fold) was
15 seen between days 1 to 7 for SMX (0.066 ± 0.034) comparing to the control (0.150 ± 0.022 , Figure
16 2A). Fungal biomass accumulation in alder leaves followed a similar pattern than that observed for
17 bacterial biomass (Time $P < 0.0001$, Figures 2C and 2D), though they were neither impacted by
18 antibiotics ($P = 0.329$) nor by the interaction between antibiotics and periphyton ($P = 0.747$, Table
19 S2).

20 Among the five extracellular enzymatic activities measured in alder leaves (β -glucosidase,
21 cellobiohydrolase, phenol oxidase, alkaline phosphatase and leucine-aminopeptidase), only β -
22 glucosidase activity was affected by antibiotics ($P < 0.005$, Table S2, Figure 4). Activity expressed both
23 over time (Figure S1) or integrated as the AUC (Figure 4) showed higher scores in the antibiotic
24 treatments than in the controls (Tukey's test, $P < 0.05$, Figure 4). However, these higher activity
25 scores did not suggest more efficient decomposition of alder leaves since turnover activity ($TA = 1/k$)
26 calculations were higher on average in the antibiotic treatments ($111.4 \pm 18.8 \text{ mmol MUF gDM}^{-1}$)
27 than in the controls ($81.0 \pm 11.2 \text{ mmol MUF gDM}^{-1}$).



1

2 Figure 4. Area under the curve (AUC) of biomass-specific β -glucosidase activity in alder leaves
 3 incubated alone (black bars) or in the presence (grey bars) of periphyton subjected to control, SMZ
 4 and SMX treatments. Values are means and standard errors of triplicate AUC measures per
 5 experimental treatment. Differences between treatments are marked by letters (a>b>c, Tukey's test,
 6 $P < 0.05$).

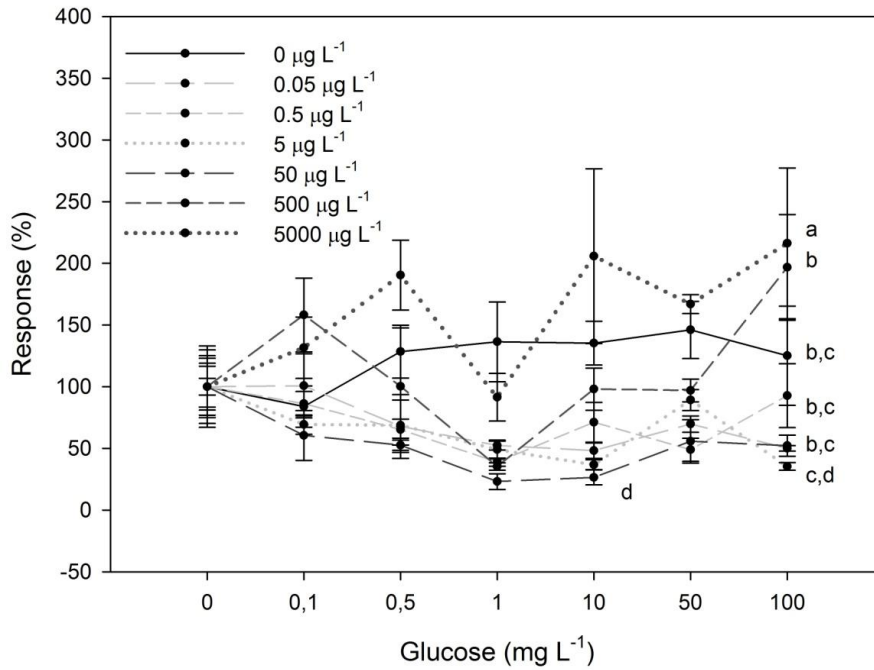
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8 3.3. Interaction between periphyton and antibiotics on leaf-litter decomposers

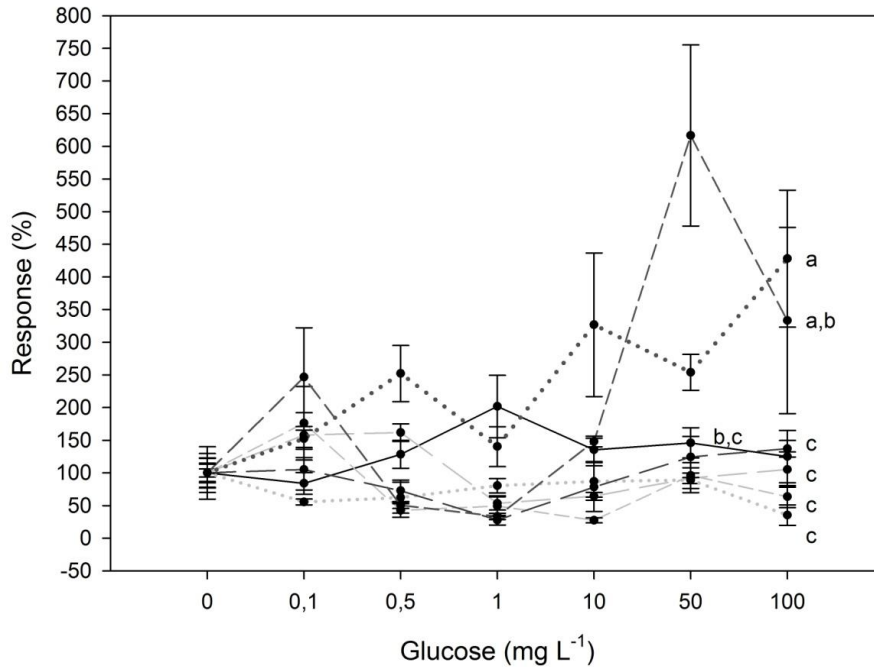
9 The only interaction between exposure to periphyton (I) and antibiotic (A) treatments was observed
 10 for β -glucosidase activity (I x A interaction, $P < 0.05$, Table S2). The highest activity values were
 11 recorded in the presence of SMZ in the L treatment and in the presence of SMX in the L+P treatment
 12 (Tukey's test, $P < 0.05$; Figures 4 and S1).

13 Dose-response curves for β -glucosidase revealed different activity responses to glucose
 14 concentrations depending on the concentration of SMZ ($P < 0.001$) and SMX ($P < 0.001$) tested. While
 15 increase in glucose concentration tends to repress progressively β -glucosidase activity at SMZ and
 16 SMX concentrations ranging from 0.05 to 50 $\mu\text{g L}^{-1}$, these relationships are the opposite for antibiotic
 17 concentrations ranging from 500 to 5000 $\mu\text{g L}^{-1}$. Dose-response results confirm that at antibiotic
 18 concentrations $\geq 500 \mu\text{g L}^{-1}$, and despite the increase in glucose concentration in the media, leaf-litter
 19 decomposers continue to increase β -glucosidase activity. This response was more marked for SMX
 20 than for SMZ (Figure 5), which contrasted with the results obtained in Figure 4 where SMZ had a
 21 stronger effect on β -glucosidase than SMX.

SMZ



SMX



1

2 Figure 5. Dose-response curves for biomass-specific β -glucosidase activity depending on antibiotic
 3 concentration (from 0 to 5000 $\mu\text{g L}^{-1}$ of SMZ and SMX) and glucose concentration (from 0 to 100 mg L^{-1}). Activity response after 72h incubation is expressed as the % response against treatment without
 4 glucose. Values correspond to means ($n=3$) and standard errors of the activity response variable.
 5 Differences between treatments are marked by letters ($a>b>c>d$, Tukey's test, $P < 0.05$).
 6

1

2 4. DISCUSSION

3

4 4.1. Periphyton effect on leaf-litter decomposers

5 Photosynthetic extracellular release in our microcosm experiment was moderate (2.3 times more
6 DOC in L+P than in L) and this can be explained by the high dissolved nutrient conditions (averaging
7 $1.94 \pm 0.03 \text{ mg N L}^{-1}$ and $0.16 \pm 0.01 \text{ mg P L}^{-1}$ in this study for all treatments and sampling times)
8 which are proved to repress labile carbon excretion from algae (Daufresne and Loreau, 2001; Danger
9 *et al.*, 2013). Photosynthetic extracellular release is essentially favored by nutrient-limited algae
10 subjected to stoichiometric constraints (Ziegler and Lyon, 2010). Algal-mediated priming intensities
11 tend to be negative under low dissolved N:P and positive under high dissolved N:P conditions,
12 exhibiting an apparent switchpoint between molar N:P of 16 and 64 (Halvorson *et al.*, 2019).
13 Considering that N:P molar ratios in the water from L+P microcosms (including control, SMZ and
14 SMX treatments) averaged 37.4 ± 1.7 , conditions reached in our study were at the switchpoint
15 between the positive and the negative algal-mediated priming.

16 The decomposition rates of alder leaves measured in this study (from 0.018 to 0.072 day⁻¹) were
17 higher compared to those measured in field studies in temperate streams (0.0166, Gonçalves *et al.*,
18 2006; 0.0169 +/- 0.0027, Ferreira *et al.*, 2012; 0.0193 +/- 0.0052, Bruder *et al.*, 2014). These
19 differences in alder decay rates can be explained by the relatively high and constant temperature
20 conditions setup in our experiment (19 °C 24h/24h) comparing to those observed in the literature.
21 These higher temperatures lead to an increase in microbial processing of alder leaves.

22 In the presence of periphyton (L+P treatment), the microbial decomposition of alder leaves was
23 reduced by 25% compared to treatments without periphyton (L treatment), which contradicts our
24 initial hypothesis of positive priming effect on leaf-litter decomposition in the presence of periphyton
25 (Guenet *et al.*, 2010, Hotchkiss *et al.*, 2014, Danger *et al.*, 2013). Some authors describe this negative
26 priming as leaf-litter decomposers exposed to high dissolved N and P likely to allocate labile algal C
27 towards growth, respiration or reproduction instead of degradative enzymes and decomposition
28 (Halvorson *et al.*, 2018, Catalán *et al.*, 2015). This is probably due to a preferential substrate use
29 (Guenet *et al.*, 2010). However, decomposer responses in our study are contradictory to the negative
30 priming observations made in the literature. For instance, Halvorson *et al.* (2018) observed that the
31 presence of active periphyton increased bacterial abundance in leaves of *Liriodendron tulipifera* and
32 *Quercus nigra* species by 141% and 733%, respectively. Moreover, active periphytic algae could
33 potentially enrich the microbial biomass associated with leaves in N and P (Danger *et al.*, 2013,
34 Halvorson *et al.*, 2016) thus modifying their corresponding C:N and C:P molar ratios. In our study, the
35 periphyton neither stimulated bacterial abundance nor decreased C:nutrient molar ratios in leaves.
36 An explanation for such inconsistencies is the duration of the experiments. Whereas the bacterial
37 biomass and C:P ratio responses to the presence of algae were observed after 30 days and 42 days
38 exposure in previous studies (Danger *et al.*, 2013; Halvorson *et al.*, 2018), the responses in our
39 experiment appeared much earlier between 7 and 16 days of incubation. This faster response could
40 be linked to the fast decaying character of alder leaves (N-rich species, Chauvet 1987), though this
41 cannot be confirmed since the same litter species were used to prove algal priming in the study of
42 Danger *et al.* (2013). Another possible explanation is that litter decomposers might compete with
43 algae for dissolved inorganic nitrogen in the L+P treatment. The N supply may be insufficient to
44 support heterotrophic investment in N-costly degradative enzymes, thus limiting degradation of the
45 litter (Jabiol *et al.*, 2018; Halvorson *et al.* 2019) and biomass growth. Finally, our study revealed an

1 increase in the extent of bacterial species richness in leaves accompanied by periphyton, suggesting
2 an increase in the accommodation capacity of the community, whereas the presence of periphyton
3 did not modify fungal species richness.

4 **4.2. Antibiotics effect on leaf-litter decomposers**

5 Sulfonamides reduced bacterial biomass accrual in leaves but did not affect litter decomposition
6 rates. It is recognized that sulfonamides (including SMZ and SMX molecules) exert a bacteriostatic
7 effect characterized by the inhibition of folic acid synthesis, and therefore amino acid synthesis, in
8 both Gram-negative and Gram-positive bacteria (Petri *et al.*, 2001). This bacteriostatic effect can
9 explain the lower biomass accumulation in leaves subjected to SMZ and SMX. However, the effect of
10 the two sulfonamides varied between L+P and L. The recovery of bacterial biomass in L exposed to
11 SMX could be attributed to adaptation and resistance processes setup by microorganisms as
12 observed in other studies (Thiele-Bruhn and Beck, 2005; Liu *et al.*, 2009). The greater dissipation of
13 SMX in L (41.4 ± 4.8 %) compared to L+P (20.5 ± 13.8 %) and the transitory increase in bacterial OTU
14 richness in L compared to L+P (at day 7) could confirm the better adaptation of the former
15 decomposer community to SMX. The dissipation percentages in our study were in the same range as
16 those obtained in the study by Collado *et al.* (2013) at $50 \mu\text{g SMX L}^{-1}$ in a lab scale sequencing batch
17 reactor bacterial community seeded with activated sludge. Concerning SMZ dissipation, Topp *et al.*
18 (2013) reported the isolation of *Microbacterium* sp. C448, a bacterium able to mineralize the benzylic
19 portion of SMZ and to excrete stoichiometric amounts of ADMP. The ability of the white-rot fungus
20 *Trametes versicolor* to degrade SMZ has also been reported through the action of laccase-type
21 enzymes (García-Galán *et al.*, 2011). A similar SMZ-degradation mechanism occurring in the leaf-litter
22 decomposer community cannot be ruled out. The differences observed in sulfonamides-dissipation
23 percentages between treatments can be explained by a range of biotic and abiotic factors that would
24 need to be investigated in further experiments focusing on sulfonamide biodegradation.

25 As expected, the mode of action of the two antibiotics tested did not alter fungal biomass accrual in
26 leaves. Potential indirect effects of antibiotics enhancing fungal biomass accrual in leaves due to
27 reduced competition with bacteria for organic matter use (*e. g.* Romani *et al.*, 2006) were
28 hypothesized, but not observed in this experiment.

29 Sulfonamides increased β -glucosidase activity in leaves. β -glucosidase enzyme degrades cellobiose
30 and other relatively small oligomers containing β -d-glucose linkages by splitting off the terminal β -d-
31 glucose residue (Desphande and Eriksson, 1988). Then, the β -glucosidase increase was perhaps
32 linked to the use of an external C-supply in the antibiotic treatments of both L and L+P. This external
33 C-source can be linked to C-oligomers release from bacterial cells impaired by the antibiotics and
34 later taken up by fungi. This enzymatic response resulting from fungal-bacterial interactions has also
35 been suggested in other studies (Cycón *et al.*, 2006, Artigas *et al.*, 2012). However, this external C-
36 supply was not noticeable on the C:N and C:P molar ratios of water or biofilms in antibiotic
37 treatments compared to controls (Table S2). Furthermore, the increase in β -glucosidase could also be
38 associated with a general stress response from the decomposer community to resist antibiotics.
39 Although the molecular mechanism controlling this enzymatic response would deserve further
40 research, one could hypothesize that β -glucosidase activity will increase the glucose available for the
41 metabolism of microbial decomposers which is probably impaired by the presence of antibiotics.

42 **4.3. Interaction between periphyton and antibiotics**

43 The presence of periphyton did not alleviate the impact of antibiotics on leaf-litter decomposers (*i. e.*
44 bacterial biomass) which contradicts our hypothesis. Only β -glucosidase activity in leaves was

1 affected by the interaction between periphyton and antibiotics. The highest activity values were
2 recorded in the presence of SMZ in the L treatment and in the presence of SMX in the L+P treatment.
3 These differences are probably explained by the different concentration of antibiotics and C-sources
4 tested in both experiments. The in-depth study on β -glucosidase response to antibiotics and glucose
5 (dose-response curves) revealed that sulfonamide concentrations $\geq 500 \mu\text{g L}^{-1}$ stimulated β -
6 glucosidase even when increasing the concentration of the end-product (glucose) of β -glucosidase
7 enzymatic reaction. One could expect an increase in bacterial biomass in leaves concomitant with the
8 increase in glucose concentration in the media, thus leading to enhanced β -glucosidase activity
9 values as observed in other studies (e.g. Ylla *et al.*, 2009). Unexpectedly, neither antibiotic nor
10 glucose adjustment significantly modified bacterial biomass accrual in alder leaves during the 72h
11 experiment (data not shown). After correction of β -glucosidase by the corresponding bacterial cell
12 density present in the leaf disk, we can conclude that at high sulfonamide concentrations ($\geq 500 \mu\text{g L}^{-1}$),
13 SMX produced greater stress than SMZ on the metabolism of litter-decomposer communities.

14

15 **5. CONCLUSIONS**

16 Photosynthetic extracellular release from the periphyton neither accelerated microbial
17 decomposition of alder leaves nor buffered the impact of sulfonamides on the bacterial decomposer
18 community, and this can be explained by the relatively high N:P ratio in stream water. Further
19 experiments mimicking positive algal-mediated priming (low water N:P ratios) should be conducted
20 in order to assess whether the effect of sulfonamides is alleviated on the bacterial decomposer
21 community. The concentration of antibiotics tested in this experiment ($5 \mu\text{g L}^{-1}$) does not represent
22 an ecotoxicological risk for the microbial litter decomposition process itself but consistently impaired
23 the bacterial community which sustains higher trophic levels in the aquatic food web. Special
24 attention should be paid to β -glucosidase activity response to sulfonamides and its potential impact
25 in the stability of dissolved organic matter recycling in stream ecosystems.

26

27 **6. ACKNOWLEDGMENTS**

28

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33 **7. REFERENCES**

34 Allen, J., Laviale, M., Cellamare, M., Bachelet, Q., Felten, V., Danger, M., 2020. Do leaf-litter
35 decomposers control biofilm primary production and benthic algal community structure in
36 forest streams? Insights from an outdoor mesocosm experiment. *Freshwat. Biol.* 65, 1256-
37 1269.

38

39 Artigas, J., Majerholc, J., Foulquier, A., Margoum, C., Volat, B., Neyra, M., Pesce, S., 2012. Effects of
40 the fungicide tebuconazole on microbial capacities for litter breakdown in streams.
41 *Aquat.Tox.* 122-123, 197-205.

42

- 1 Artigas, J., Romani, A.M., Sabater, S. 2008. Relating nutrient molar ratios of microbial attached
2 communities to organic matter utilization in a forested stream. *Fund. Appl. Limnol.* 173, 255-
3 267.
4
- 5 Baldy, V., Chauvet, E., Charcosset, J.Y., Gessner, M.O. 2002. Microbial dynamics associated with
6 leaves decomposing in the mainstem and floodplain pond of a large river. *Aquat. Microb.*
7 *Ecol.* 28, 25-36.
8
- 9 Baldy, V., Gessner, M.O., Chauvet, E. 1995. Bacteria, fungi and the breakdown of leaf litter in a large
10 river. *Oikos*, 93-102.
11
- 12 Batt, A.L., Snow, D.D., Aga, D.S. 2006. Occurrence of sulfonamide antimicrobials in private water
13 wells in Washington County, Idaho, USA. *Chemosphere.* 64, 1963-1971.
14
- 15 Borrel, G., Colombet, J., Robin, A., Lehours, A.C., Prangishvili, D., Sime-Ngando, T. 2012. Unexpected
16 and novel putative viruses in the sediments of a deep-dark permanently anoxic freshwater
17 habitat. *ISME J.* 6,2119-2127.
18
- 19 Boxall, A.B., Blackwell, P., Cavallo, R., Kay, P., Tolls, J. 2002. The sorption and transport of a
20 sulphonamide antibiotic in soil systems. *Toxicol Lett.* 131, 19-28.
21
- 22 Boxall, A.B., Kolpin, D.W., Halling-Sørensen, B., Tolls, J. 2003. Peer reviewed: are veterinary medicines
23 causing environmental risks? *Env. Sci. Technol.* 37, 286-294.
24
- 25 Bratbak, G., Dundas, I. 1984. Bacterial dry matter content and biomass estimations. *Appl. Environ.*
26 *Microbiol.* 48, 755-757.
27
- 28 Bruder, A., Schindler, M.H., Moretti, M.S., Gessner, M.O. 2014. Litter decomposition in a temperate
29 and a tropical stream: the effects of species mixing, litter quality and shredders. *Freshwat.*
30 *Biol.* 59, 438-449.
31
- 32 Carney, M., Hungate, B.A., Drake, B.G., Megonigal, J.P. 2007. Altered soil microbial community at
33 elevated CO₂ leads to loss of soil carbon. *PNAS USA.* 104, 4990-4995.
34
- 35 Catalán, N., Kellerman, A.M., Peter, H., Carmona, F., Tranvik, L.J. 2015. Absence of a priming effect on
36 dissolved organic carbon degradation in lake water. *Limnol. Oceanogr.* 60, 159-168.
37
- 38 Chauvet, E. 1987. Changes in the chemical composition of alder, poplar and willow leaves during
39 decomposition in a river. *Hydrobiologia*, 148: 35-44.
40
- 41 Collado, N., Buttiglieri, G., Marti, E., Ferrando-Climent, L., Rodriguez-Mozaz, S., Barcelo, D., Comas, J.,
42 Rodriguez-Roda, I. 2013. Effects on activated sludge bacterial community exposed to
43 sulfamethoxazole. *Chemosphere.* 93, 99-106.
44
- 45 Conkle, J.L., White, J.R. 2012. An initial screening of antibiotic effects on microbial respiration in
46 wetland soils. *J. Environ. Sci. Health A.* 47, 1381-1390.
47
- 48 Crouzet, O., Batisson, I., Besse-Hoggan, P., Bonnemoy, F., Bardot, C., Poly, F., Bohatier, J., Mallet, C.
49 2010. Response of soil microbial communities to the herbicide mesotrione: a dose-effect
50 microcosm approach. *Soil Biol. Biochem.* 42, 193-202.
51

- 1 Cycón, M., Piotrowska-Seget, Z., Kaczyńska, A., Kozdrój, J. 2006. Microbiological characteristics of a
2 Sandy loam soil exposed to tebuconazole and λ -cyhalothrin under laboratory conditions.
3 Ecotoxicology. 15, 639–646.
4
- 5 Danger, M., Cornut, J., Chauvet, E., Chavez, P., Elger, A., Lecerf, A. 2013. Benthic algae stimulate leaf
6 litter decomposition in detritus-based headwater streams: a case of aquatic priming effect?
7 Ecology. 94, 1604-1613.
8
- 9 Daufresne, T., Loreau, M. 2010. Ecological stoichiometry, primary producer-decomposer interactions,
10 and ecosystem persistence. Ecology. 82, 3069-308.
11
- 12 Deshpande, V., and Eriksson, K.E. 1988. 1,4- β -Glucosidases of *Sporotrichum pulverulentum*. In
13 Methods in Enzymology Volume 160 Part A Cellulose and Hemicellulose, Wood, W.A., and
14 Kellog, S.T., eds. (AcademicPress, Inc.), pp. 415–424.
15
- 16 Ferreira, V., Encalada, A.C., Graça, M.A.S. 2012. Effects of litter diversity on decomposition and
17 biological colonization of submerged litter in temperate and tropical streams. Freshwat. Sci.
18 31, 945-962.
19
- 20 García-Galán, M.J., Rodríguez-Rodríguez, C.E., Vicent, T., Caminal, G., Díaz-Cruz, M., Barceló, D. 2011.
21 Biodegradation of sulfamethazine by *Trametes versicolor*: Removal from sewage sludge and
22 identification of intermediate products by UPLC–QqTOF-MS. Sci. Total Environ. 409, 5505-
23 5512.
24
- 25 García-Galán, M.J., Díaz-Cruz, M., Barceló, D. 2008. Identification and determination of metabolites
26 and degradation products of sulfonamide antibiotics. TRAC Trend. Anal. Chem. 27, 1008-
27 1022.
28
- 29 Gessner, M.O., Chauvet, E. 1993. Ergosterol-to-biomass conversion factors for aquatic
30 hyphomycetes. Appl. Environ. Microbiol. 59, 502-507.
31
- 32 Gessner, M.O., Swan, C.M., Dang, C.K., McKie, B.G., Bardgett, R.D., Wall, D.H., Hättenschwiler, S.
33 2010. Diversity meets decomposition. Trends Ecol. Evol. 25, 372-380.
34
- 35 Gonçalves, J.F., Graça, M.A.S., Callisto, M. 2006. Leaf-litter breakdown in 3 streams in temperate,
36 Mediterranean, and tropical Cerrado climates. J. N. Am. Benthol. Soc. 25, 344–355.
37
- 38 Grasshoff, K., Ehrhardt, M., Kremling, K. 1983. Methods of seawater analysis. Second, revised and
39 extended edition. In: Grasshoff, K., Ehrhardt, M., Kremling, K. (eds). – VerlagChemie,
40 Weinheim, Germany, 419 p.
41
- 42 Guenet, B., Danger, M., Abbadie, L., Lacroix, G. 2010. Priming effect: bridging the gap between
43 terrestrial and aquatic ecology. Ecology. 91, 2850-2861.
44
- 45 Gutierrez, I.R., Watanabe, N., Harter, T., Glaser, B., Radke, M. 2010. Effect of sulfonamide antibiotics
46 on microbial diversity and activity in a Californian *MollicHaploxera*lf. J. Soil Sediment. 10,
47 537–544.
48
- 49 Halvorson, H.M., Francoeur, S.N., Findlay, R.H., Kuehn, K.A. 2019. Algal-mediated priming effects on
50 the ecological stoichiometry of leaf-litter decomposition: meta-analysis. Front. Earth Sci. 7,
51 Article 76.
52

- 1 Halvorson, H.M., Barry, J.R., Lodato, M.B., Findlay, R.H., Francoeur, S.N., Kuehn, K.A. 2018. Periphytic
2 algae decouple fungal activity from leaf litter decomposition via negative priming. *Funct.*
3 *Ecol.* 33, 188-201.
4
- 5 Halvorson, H.M., Scott, E.E., Entrekin, S.A., Evans-White, M.A., Scott, J.T. 2016. Light and dissolved
6 phosphorus interactively affect microbial metabolism, stoichiometry and decomposition of
7 leaf litter. *Freshwat. Biol.* 61, 1006–1019.
8
- 9 Hieber, M., Gessner, M.O. 2002. Contribution of stream detritivores, fungi, and bacteria to leaf
10 breakdown based on biomass estimates. *Ecology.* 83, 1026-1038.
11
- 12 Hotchkiss, E.R., Hall, R.O., Baker, M.A., Rosi-Marshall, E.J., Tank, J.L. 2014. Modeling priming effects
13 on microbial consumption of dissolved organic carbon in rivers. *J. Geophys. Res-Biogeo.* 119,
14 982–995.
15
- 16 Jabiol, J., Cornut, J., Tlili, A., Gessner, M.O. 2018. Interactive effects of dissolved nitrogen, phosphorus
17 and litter chemistry on stream fungal decomposers. *FEMS Microbiol. Ecol.* doi:
18 10.1093/femsec/fiy151.
19
- 20 Kolpin, D.W., Furlong, E.T., Meyer, M.T., Thurman, E.M., Zaugg, S.D., Barber, L.B., Buxton, H.T. 2002.
21 Pharmaceuticals, hormones, and other organic wastewater contaminants in US streams,
22 1999-2000: A national reconnaissance. *Environ. Sci. Technol.* 36, 1202-1211.
23
- 24 Kümmerer, K. 2009. Antibiotics in the aquatic environment—a review—part I. *Chemosphere*, 75, 417-
25 434.
26
- 27 Liu, F., Ying, G.G., Tao, R., Zhao, J.L., Yang, J.F., Zhao, L.F. 2009. Effects of six selected antibiotics on
28 plant growth and soil microbial and enzymatic activities. *Environ. Poll.* 157, 1636-1642.
29
- 30 Lock, M.A., Wallace, R.R., Costerton, J.W., Ventullo, R.M., Charlton, S.E. 1984. River epilithon: toward
31 a structural-functional model. *Oikos.* 42, 10-22
32
- 33 Martin-Laurent, F., Topp, E., Billet, L., Batisson, I., Malandain, C., Besse-Hoggan, P., ... Devers-
34 Lamrani, M. 2019. Environmental risk assessment of antibiotics in agroecosystems:
35 ecotoxicological effects on aquatic microbial communities and dissemination of antimicrobial
36 resistances and antibiotic biodegradation potential along the soil-water continuum. *Environ.*
37 *Sci. Pollut. Res.* 26, 18930-18937.
38
- 39 Matongo, S., Birungi, G., Moodley, B., Ndungu, P. 2015. Occurrence of selected pharmaceuticals in
40 water and sediment of Umgeni River, KwaZulu-Natal, South Africa. *Environ. Sci. Pollut. Res.*
41 22, 10298–10308.
42
- 43 Mora-Gómez, J., Elosegí, A., Duarte, S., Cássio, F., Pascoal, C., Romani, A.M. 2016. Differences in the
44 sensitivity of fungi and bacteria to season and invertebrates affect leaf litter decomposition
45 in a Mediterranean stream. *FEMS Microbiol. Ecol.* 92, 1-13.
46
- 47 Murphy, J., Riley, J.P. 1962. A modified single solution method for the determination of phosphate in
48 natural waters. *Anal. Chim. Acta.* 27, 31-36.
49
- 50 Nikolcheva, L.G., Bärlocher, F. 2004. Seasonal substrate preferences of fungi colonizing leaves in
51 streams : traditional versus molecular evidence. *Environ. Microbiol.* 7, 270-280.
52

- 1
2 Pastor, A., Compson, Z.G., Dijkstra, P., Riera J.L., Martí, E., Sabater, F., Hungate, B.A., Marks, J.C.
3 2014. Stream carbon and nitrogen supplements during leaf litter decomposition: contrasting
4 patterns for two foundation species. *Oecologia*. 176, 1111-1121.
5
6 Peng, X., Tan, J., Yu, Y., Wang, Z. 2008. Multiresidue determination of fluoroquinolone, sulfonamide,
7 trimethoprim, and chloramphenicol antibiotics in urbanwaters in China. *Environ. Toxicol.*
8 *Chem.* 27, 73-79.
9
10 Petersen, R.C., Cummins, K.W. 1974. Leaf processing in a woodland stream. *Freshwat. Biol.* 4, 343-
11 368.
12
13 Petri, W. 2001. Goodman and Gilman's the Pharmacological Basis of Therapeutics, 10th ed.; Hardman
14 J. G., Limbird L. E., Eds.; McGrawHill: New York, pp. 1171-1188.
15
16 Pinna, M.V., Castaldi, P., Deiana, P., Pusino, A., Garau, G. 2012. Sorption behavior of sulfamethazine
17 on unamended and manure-amended soils and short-term impact on soil microbial
18 community. *Ecotox. Environ. Safe.* 84, 234-242.
19
20 Romani, A.M., Artigas, J., Ylla, I. 2012. Extracellular enzymes in aquatic biofilms: microbial
21 interactions vs water quality effects in the use of organic matter, In: Lear G, Lewis G (Eds)
22 *Microbial Biofilms: Current Research and Applications*. Caister, ISBN: 978-1-904455-96-7, pp
23 153-174.
24
25 Romani, A.M., Fischer, H., Mille-Lindblom, C., Tranvik, L.J. 2006. Interactions of bacteria and fungi on
26 decomposing litter: differential extracellular enzyme activities. *Ecology*. 87, 2559-2569.
27
28 Romani, A.M., Sabater, S. 2000. Influence of algal biomass on extracellular enzyme activity in river
29 biofilms. *Microbial Ecol.* 40, 16-24.
30
31 Roose-Amsaleg, C., Laverman, A.M. 2016. Do antibiotics have environmental side-effects? Impact of
32 synthetic antibiotics on biogeochemical processes. *Environ. Sci. Pollut. Res.* 23, 4000-4012.
33
34 Rossi, F., Mallet, C., Portelli, C., Donnadieu, F., Bonnemoy, F., Artigas, J. 2019. Stimulation or
35 inhibition: leaf microbial decomposition in streams subjected to complex chemical
36 contamination. *Sci. Total Environ.* 648, 1371-1383.
37
38 Sarmah, A.K., Meyer, M.T., Boxall, A.B. 2006. A global perspective on the use, sales, exposure
39 pathways, occurrence, fate and effects of veterinary antibiotics (VAs) in the
40 environment. *Chemosphere* 65, 725-759.
41
42 Scieczko, A., Maschek, M., Peduzzi, P. 2015. Algal extracellular release in river-floodplain dissolved
43 organic matter: response of extracellular enzymatic activity during a post-flood period. *Front.*
44 *Microbiol.* 80. doi.org/10.3389/fmicb.2015.00080
45
46 Seydel, J.K. 1968. Sulfonamides, Structure-Activity Relationship, and Mode of Action: Structural
47 Problems of the Antibacterial Action of 4-Aminobenzoic Acid (PABA) Antagonists. *J Pharm.*
48 *Sci.* 57: 1455-1478.
49
50 Simon, K.S., Simon, M.A., Benfield, E.F. 2009. Variation in ecosystem function in Appalachian streams
51 along an acidity gradient. *Ecol. Appl.* 19, 1147-11460
52

- 1 Sinsabaugh, R.L., FollstadShah, J.J. 2012. Eoenzymatic stoichiometry and ecological theory. *Annu.*
2 *Rev. Ecol. Evol. Syst.* 43, 313-343.
3
4
- 5 Sukul, P., Spiteller, M. 2006. Sulfonamides in the environment as veterinary drugs. In: *Reviews of*
6 *environmental contamination and toxicology* (pp. 67-101). Springer, New York, NY.
7
8
- 9 Theil-Nielsen, J., Søndergaard, M. 1998. Bacterial carbon biomass calculated from biovolumes.
10 *Archiv. Hydrobiol.* 141, 195-207.
11
- 12 Thiele-Bruhn, S., Beck, I.C. 2005. Effects of sulfonamide and tetracycline antibiotics on soil microbial
13 activity and microbial biomass. *Chemosphere.* 59, 457-465.
14
- 15 Thiele-Bruhn, S., Seibicke, T., Schulten, H.R., Leinweber, P. 2004. Sorption of sulfonamide
16 pharmaceutical antibiotics on whole soils and particle-size fractions. *J. Environ. Qual.* 33,
17 1331-1342.
18
- 19 Topp, E., Chapman, R., Devers-Lamrani, M., Hartmann, A., Martin-Laurent, F., Marti, R., Sabourin, L.,
20 Scott, A., Sumaraha, M. 2013. Accelerated biodegradation of veterinary antibiotics in
21 agricultural soil following long-term exposure, and isolation of a sulfamethazine-
22 degrading *Microbacterium* sp. *J. Environ. Qual.* 42, 173-178.
23
- 24 Urban, D., Chevance, A., Moulin, G. 2019. Sales survey of veterinary medicinal products containing
25 antimicrobials in France in 2018. Annual report, French Agency for Food, Environmental and
26 Occupational Health & Safety (ANSES) – French Agency for Veterinary Medicinal Products
27 (ANMV). 100 pp.
28
- 29 Weishaar, J.L., Aiken, G.R., Bergamaschi, B.A., Fram, M.S., Fujii, R., Mopper, K. 2003. Evaluation of
30 specific ultraviolet absorbance as an indicator of the chemical composition and reactivity of
31 dissolved organic carbon. *Environ. Sci. Technol.* 37, 4702-4708.
32
- 33 Xia, S., Jia, R., Feng, F., Xie, K., Li, H., Jing, D., Xu, X. 2012. Effect of solids retention time on antibiotics
34 removal performance and microbial communities in an A/O-MBR process. *Bioresour.*
35 *Technol.* 106: 36-43.
36
- 37 Ylla, I., Borrego, C., Romani, A.M., Sabater, S. 2009. Availability of glucose and light modulates the
38 structure and function of a microbial biofilm. *FEMS Microbiol. Ecol.* 69, 27-42.
39
- 40 Young, R.G., Matthaei, C.D., Townsend, C.R. 2008. Organic matter breakdown and ecosystem
41 metabolism: functional indicators for assessing river ecosystem health. *J. North. Am. Benthol.*
42 *Soc.* 27, 605-625.
43
- 44 Zhang, Y., Xu, J., Zhong, Z., Guo, C., Li, L., He, Y., Chen, Y. 2013. Degradation of sulfonamides
45 antibiotics in lake water and sediment. *Environ. Sci. Poll. Res.* 20, 2372-2380.
46
- 47 Ziegler, S.E., Lyon, D.R. 2010. Factors regulating epilithic biofilm carbon cycling and release with
48 nutrient enrichment in headwater streams. *Hydrobiologia.* 657, 71-88.
- 49 Table S1. Average water temperature, pH and light measured in L and L+P microcosms at the
50 different sampling times (n=108).

Temperature (°C)	22.2 ± 0.66
pH	8.14 ± 0.19
Light (Lux)	1054 ± 121.9

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1 Table S2. Statistical analyses performed on water and microbial descriptors measured during the
 2 microcosm experiment.

3 **Two-way ANOVA**

Descriptor	Factor	F-value	P-value
Decayrates	I	5.3 _{1,12}	< 0.05
	A	0.2 _{1,12}	0.798
	I x A	0.6 _{1,12}	0.562
β-glucosidase activity (AUC)	I	2.4 _{1,12}	0.146
	A	10.7 _{2,12}	< 0.005
	I x A	5.5 _{2,12}	< 0.05
Phenol oxidase activity (AUC)	I	3.5 _{1,12}	0.085
	A	0.3 _{2,12}	0.745
	I x A	1.9 _{2,12}	0.185
TA β-glucosidase	I	1.3 _{1,12}	0.269
	A	1.9 _{2,12}	0.197
	I x A	0.1 _{2,12}	0.955

4 **Repeated-measures ANOVA**

Descriptor	Within-treatments effects			Between-treatment effects			
	Factor	F-value	P-value	Factor	F-value	P-value	
Water	Dissolved oxygen	T	6.5 _{1,13}	< 0.05	I	15.3 _{1,12}	< 0.005
		T x I	3.0 _{1,13}	0.101	A	0.4 _{2,12}	0.692
		T x A	0.6 _{2,13}	0.558	I x A	1.0 _{2,12}	0.393
		T x I x A	1.0 _{2,13}	0.371			
DOC	DOC	T	1.3 _{1,13}	0.271	I	5.2 _{1,12}	< 0.05
		T x I	1.0 _{1,13}	0.346	A	0.3 _{2,12}	0.752
		T x A	1.4 _{2,13}	0.286	I x A	1.3 _{2,12}	0.577
		T x I x A	1.3 _{2,13}	0.307			
SUVA ₂₅₄	SUVA ₂₅₄	T	3.3 _{2,21}	0.062	I	21.1 _{1,12}	< 0.005
		T x I	1.1 _{2,21}	0.349	A	2.9 _{2,12}	0.095
		T x A	1.5 _{4,21}	0.243	I x A	1.5 _{2,12}	0.253
		T x I x A	1.8 _{4,21}	0.162			
C:N water	C:N water	T	4.2 _{2,19}	< 0.05	I	3.2 _{1,12}	0.098
		T x I	2.3 _{2,19}	0.135	A	0.2 _{2,12}	0.810
		T x A	0.5 _{3,19}	0.719	I x A	0.3 _{2,12}	0.759
		T x I x A	0.8 _{3,19}	0.494			
C:P water	C:P water	T	4.5 _{1,15}	< 0.05	I	6.5 _{1,12}	< 0.05
		T x I	5.5 _{1,15}	< 0.05	A	0.1 _{2,12}	0.974
		T x A	0.3 _{3,15}	0.782	I x A	0.1 _{2,12}	0.941
		T x I x A	0.4 _{3,15}	0.754			
N:P water	N:P water	T	5.4 _{3,32}	< 0.005	I	13.4 _{1,11}	< 0.005
		T x I	3.5 _{3,32}	< 0.05	A	0.2 _{2,11}	0.857
		T x A	0.8 _{6,32}	0.541	I x A	0.4 _{2,11}	0.679
		T x I x A	0.6 _{6,32}	0.699			
Leaves	Bacterial biomass	T	4.9 _{3,36}	< 0.01	I	4.8 _{1,12}	< 0.05
		T x I	0.4 _{3,36}	0.779	A	11.8 _{2,12}	< 0.005
		T x A	1.25 _{6,36}	0.302	I x A	0.8 _{2,12}	0.464
		T x I x A	0.6 _{6,36}	0.690			
Fungal biomass	T	52.6 _{3,36}	< 0.001	I	2.0 _{1,12}	0.181	

	T x I	3.9 _{3,36}	< 0.05	A	1.2 _{2,12}	0.329
	T x A	2.4 _{6,36}	< 0.05	I x A	0.3 _{2,12}	0.747
	T x I x A	1.0 _{6,36}	0.424			
C:N leaves	T	0.9 _{1,15}	0.371	I	2.0 _{1,12}	0.181
	T x I	1.4 _{1,15}	0.269	A	0.8 _{2,12}	0.488
	T x A	1.3 _{2,15}	0.298	I x A	1.4 _{2,12}	0.284
	T x I x A	1.4 _{2,15}	0.279			
C:P leaves	T	10.2 _{3,31}	< 0.001	I	0.2 _{1,12}	0.664
	T x I	2.7 _{3,31}	0.072	A	1.5 _{2,12}	0.262
	T x A	1.6 _{5,31}	0.181	I x A	2.5 _{2,12}	0.125
	T x I x A	1.8 _{5,31}	0.140			

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1 Table S3. Antibiotic concentrations (mean and standard errors) after 7 and 14 days of experiment.

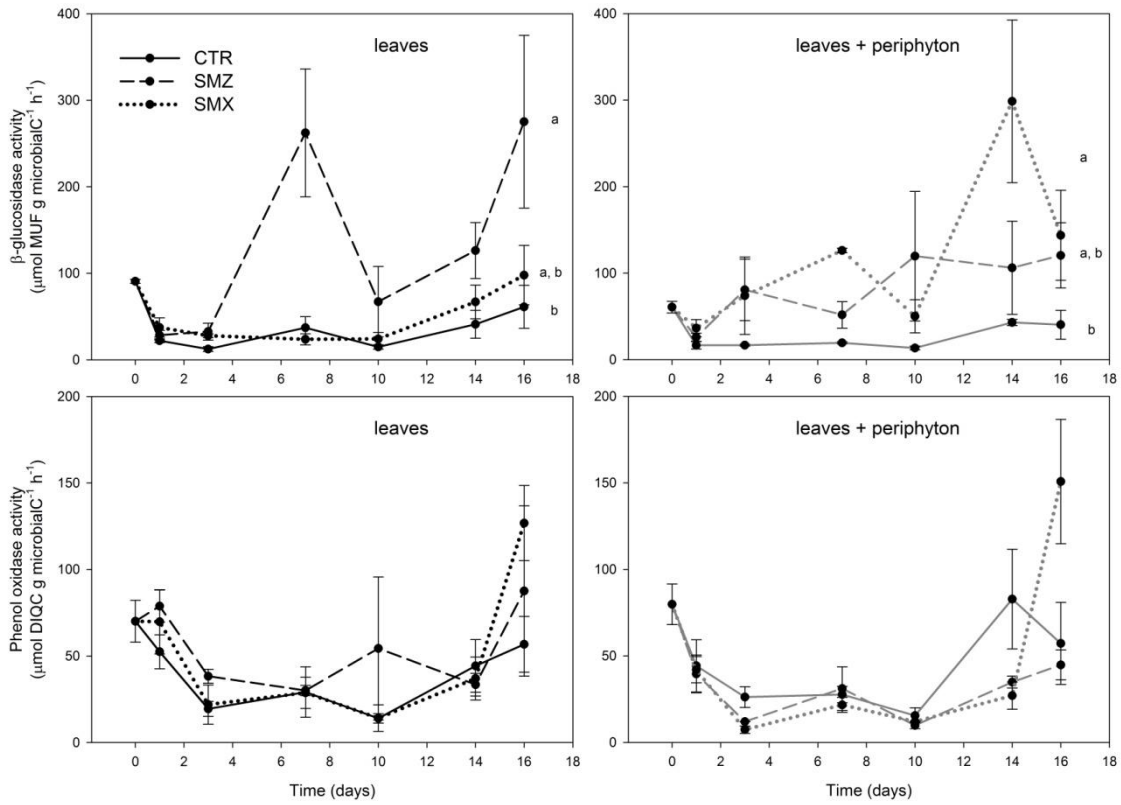
Day	I	A	Concentration ($\mu\text{g L}^{-1}$)
7	L	SMZ	4.86 (0.38)
	L+P	SMZ	5.73 (0.08)
	L	SMX	5.91 (0.14)
	L+P	SMX	4.57 (0.04)
14	L	SMZ	2.20 (0.21)
	L+P	SMZ	2.65 (0.04)
	L	SMX	2.70 (0.09)
	L+P	SMX	3.86 (0.04)

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1 Figure S1. Biomass-specific activity of β -glucosidase and phenol oxidase enzymes measured in L (left-
 2 side plots) and L+P (right-side plots) treatments subjected to controls without antibiotics (solid line),
 3 SMZ (dashed-line) and SMX (dotted line) treatments. Values are means (n=3) and standard errors per
 4 each treatment at each sampling time. Differences between experimental treatments are marked by
 5 letters (a > b, $P < 0.05$, Tukey's test)



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