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Organic matter breakdown in streams in a region of contrasting anthropogenic land use

K. Voß *, D. Fernández, R.B. Schäfer

Quantitative Landscape Ecology, Institute for Environmental Science, University of Koblenz-Landau, Fortstraße 7, D-76829 Landau, Germany

HIGHLIGHTS

- Investigated land use effects on organic matter breakdown
- Only microbial breakdown differed between land use types
- Tree cover correlated with invertebrate-mediated breakdown
- pH correlated with microbial breakdown
- Land use insufficient to distinguish differences in breakdown

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ABSTRACT

Streams provide ecosystem services to humans that depend on ecosystem functions, such as organic matter breakdown (OMB). OMB can be affected by land use-related disturbance. We measured OMB in 29 low-order streams in a region of contrasting land use in south-west Germany to quantify land use effects on OMB. We deployed fine and coarse mesh leaf bags in streams of forest, agricultural, vinicultural and urban catchments to determine the microbial and invertebrate-mediated OMB, respectively. Furthermore, we monitored physico-chemical, geographical and habitat parameters to explain potential differences in OMB among land use types and sites. Only microbial OMB differed between land use types. Microbial OMB was negatively correlated with pH and invertebrate-mediated OMB was positively correlated with tree cover. Generally, OMB responded to stressor gradients rather than directly to land use. Therefore, the monitoring of specific stressors may be more relevant than land use to detect effects on ecosystem functions, and to extrapolate effects on functions, e.g. in the context of assessing ecosystem services.

1. Introduction

Freshwater ecosystems provide central ecosystem services to humans (Palmer et al., 2004), such as clean drinking water or food (e.g. fishery products: MEA, 2005). Human activities in river basins can seriously impair streams through reach and catchment-wide influences that may result in large scale alterations in stream ecosystem functions (Allan et al., 1997). Ecosystem functions such as primary production and organic matter breakdown (OMB) are pivotal for ecosystem services. OMB represents the most important energy source in terms of organic carbon in the first 10 km from the stream source, especially in forested headwaters (Wallace et al., 1997). After the input of organic matter from the riparian vegetation the benthic stream community (i.e. microbial decomposers and invertebrate detritivores) colonize, degrade and then shred the leaf material which provides energy for local and downstream food webs (Webster et al., 1999). This fundamental ecosystem function is impacted by the individual and joint occurrence of stressors such as pollution, anthropogenic nutrient enrichment and hydromorphological changes (e.g. Allan and Castillo, 2007; Paul et al., 2006; Aristi et al., 2012; Piggott et al., 2012).

According to the Millennium Ecosystem Assessment (MEA, 2005), anthropogenic land use is an important stressor for freshwater ecosystems. For instance, in the course of agricultural intensification the riparian vegetation has been increasingly logged, resulting in erosion, reduced input of coarse organic matter and consequently a reduction of available energy (Campbell et al., 1992). Furthermore, several studies reported land use effects on the benthic stream community (e.g. Delong and Brusven, 1998; Urban et al., 2006) or large-scale effects on OMB (e.g. Woodward et al., 2012), though the effects of land use on OMB were often contrasting. As an example for agricultural land use effects, Magbanua et al. (2010) found no significant differences in OMB between conventional, integrated and organic farming, while Piscart et al. (2009) observed a decrease in the OMB rate with increasing

* Corresponding author.
E-mail address: voss@uni-landau.de (K. Voß).

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agricultural intensity. Similarly, inconsistent results were reported for effects of urbanization (cf. Sponseller and Benfield, 2001 vs. Pascoal et al., 2005). These differences between studies could originate from variable associations of stressors with specific land use types as well as from variable responses to stressors. Moreover, most of the previous studies were conducted on a local scale (i.e. ≤5 streams per land use type), considered only one land use type and investigated effects during spring, while the peak of litter fall in regions dominated by deciduous trees occurs in autumn. The local approach, i.e. strong influence of particular environmental variables in a specific catchment, as well as the relatively small sample size (total n ≤ 10 in most studies) could further explain the inconsistent effects and high variability between the studies. Based on these findings, it is difficult to extrapolate land use effects to larger spatial scales, for example to determine losses of ecosystem functions and services, which has been done on several occasions in ecosystem service assessment (Maes et al., 2012). Extrapolation would require the establishment of a link between land use and ecosystem function on the respective scale, also considering that anthropogenic stressors operate over multiple spatial scales (Allan, 2004). Moreover, regulatory frameworks such as the Water Framework Directive require freshwater management from a catchment perspective, stipulating regional scale studies that include an assessment of stream ecosystem functions (Vighi et al., 2006).

In this study, we aimed to quantify the influence of multiple land use types (i.e. agriculture, viniculture, urbanization) on OMB at a regional scale (i.e. sampling sites located in different catchments) during leaf-fall in autumn. In addition, we intended to identify environmental factors that explain this influence. We hypothesized that OMB differed among land use types and that OMB is higher in anthropogenically-influenced sites due to elevated nutrient discharge.

2. Material and methods

2.1. Study area and sampling sites

The study area was located in the south-west of Germany, between the Palatinate forest nature park in the North and West, the River Rhine in the East and the Vosges in the South (Fig. S1 in the Supplementary information). To quantify the effect of land use on the OMB, we selected sampling sites in each of eight first-to-third-order streams in forested (F), agricultural (A) and vinicultural (V) areas and in seven streams in urban (U) areas. F sites were considered as relatively pristine, whereas A, V and U sites were considered as subject to different anthropogenic stressors. All streams originated in the Palatinate forest nature park, except for two agricultural streams (cf. Fig. S1), and were mainly small, fine substrate-dominated siliceous highland streams. Thus, geology and climate were very similar for the streams and their upstream catchments were relatively pristine. Sampling sites of A, V and U were located within a distance of 5 km from the edge of the nature park and selected to mainly represent the respective land use type. In addition, the sites were located upstream from drainages or discharges. U sites were chosen downstream from settlements with at least 2500 inhabitants, but upstream from wastewater treatment plants to avoid variability originating from differences in chemical input. The settlements were rather small (between 2500 and 10,000 inhabitants) but categorized as urban because the streams were hydromorphologically modified (streambed was channelized). Each site was located in a different catchment and in a different stream, except for three forested sites that were in the same catchment or stream as two urban and one vinicultural site (Fig. S1). Given that most streams originated in the Nature Park, the land cover in the stream catchments was mainly forest. Therefore, we based our site selection on the land use within a 100-m wide buffer zone of 3 km length upstream of each sampling site. This buffer zone was selected based on a recent study that found a similar or a slightly stronger ecological response to riparian (buffer zone 100-m wide) compared to catchment-scale land use (Feld, 2013). We assessed land use employing Corine Land Cover (CLC) maps (Büttner and Kosztra, 2007) and created buffers using a geographical information system (QGIS Development Team, 2014). We assigned each site the dominant land use type (>50% areal cover) within each buffer zone. The assigned land use categories were confirmed during site visits. Overall, we suggest that our site selection process guaranteed that potential effects can be attributed to the selected land use type. Two sites had to be omitted due to drying out (one vinicultural stream) and vandalism (one urban site).

2.2. Leaf deployment and calculation of breakdown rates

The study was conducted in autumn 2012 from September to October. Approximately 3 ± 0.07 g of oven-dried (60 °C for 24-h) black alder leaves (*Alnus glutinosa* — collected from a locally common riparian tree species) were placed into coarse polyethylene mesh bags (mesh size: 8 mm, bag size: 20 × 20 cm) accessible to microbial decomposers and invertebrate detritivores and into fine cylindrical nylon bags (mesh size: 250 μm, cylinder length: 15 cm) accessible only to microbial decomposers. Five replicates of each bag type were deployed in each sampling site. Leaf bags were fixed a few centimeters above the stream bottom. After 21 days the leaf bags were recovered from the streams, remaining leaf material was carefully removed from the bags, rinsed to remove mineral particles, oven-dried at 60 °C (24-h), reweighed and averaged for each type of bags (i.e. coarse and fine) for every site. To correct for handling loss, five replicates of each bag type were treated the same way as the others but returned to the laboratory immediately after brief immersion in the stream. To correct for leaching loss the same amount of bags was retrieved after a 24-h stream deployment.

Furthermore, the water temperature was recorded hourly with Synotech HOBO© temperature/data loggers 64K (Hückelhoven, Germany). This allowed us to calculate the sum of degree days (ddays^{-1}) for the deployment period of leaves, which was used to standardize the breakdown rate for temperature. The breakdown rate k in a site i was calculated based on the exponential mass loss per ddays (dd^{-1}):

$$k_i = \frac{-\ln\left(\frac{S_i(t)}{S_i(0)}\right)}{\sum_{j=1}^{t} \overline{T}_i(j)}$$

where S is the mass as a function of deployment time t (in days) and $\overline{T}$ is the mean temperature for a day j. $S_i(t)$ of leaves in fine and coarse mesh bags was corrected for handling and leaching losses. Moreover, $S_i(t)$ of leaves in coarse mesh bags was corrected for microbial breakdown to determine the contribution of invertebrates to breakdown (for details see: Benfield, 2007). Finally, we calculated the proportion (in percentage) of invertebrate-mediated OMB ($\mathrm{OMB_{macro}}$) and microbial OMB ($\mathrm{OMB_{micro}}$) of total OMB to assess the respective contribution to OMB.

2.3. Environmental variables

To identify the environmental variables that could explain the influence of the selected land use types, we measured the following physicochemical parameters: water temperature, pH, oxygen, electrical conductivity (EC), flow velocity and nutrient concentration (nitrate, nitrite, ammonium, phosphate) by on-site analysis with Macherey-Nagel visocolor® (Düren, Germany) kits. Following a protocol of EPA (2003), three defined geographical parameters were monitored within a 50-m stream section at each sampling site: mean percent of shading, width of riparian zone in meters (mean of left and right bank) and percent tree cover within the riparian zone (left and right bank maximum). Furthermore, the area of the upstream catchment of each sampling site was