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Research paper

Effects of wood addition on stream benthic invertebrates differed among seasons at both habitat and reach scales

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ABSTRACT

Addition of large wood (LW, wood pieces longer than 1 m and wider than 10 cm) into stream channels is a common restoration practice aimed at enhancing habitat diversity and fish populations, as well as the abundance and diversity of benthic invertebrates. Nevertheless, there is some controversy regarding the effects of LW restoration on invertebrate assemblages, and the effects could differ depending on the season of the year as well as on the scale of observation (habitat vs reach scale). In this study we analysed the effects of a LW restoration experiment performed in 4 mountain streams following a BACI (Before-After/Control-Impact) design. We sampled benthic invertebrates in 3 main habitats (fine inorganic sediments, called gravel; coarse inorganic sediments, called cobbles; and particulate organic matter, called POM) in winter and summer before and after the addition of LW into experimental reaches, and compared the results to those obtained from upstream control reaches. LW addition promoted the retention of gravel and organic matter, resulting in an overall decrease in the areal cover of cobbles and a significant increase in the cover of organic matter in summer. Invertebrate richness was highest in cobbles and lowest in POM, whereas density and biomass were highest in POM and lowest in gravel. At the habitat scale, LW addition promoted invertebrate and shredder richness, diversity and biomass in summer, but the opposite effect was found in winter. Community composition changed significantly with wood addition, most notably as a result of increased density of elmids, limnephilids and limoniids, and decreased density of baetids. Density of limnephilids increased 20-fold and that of limoniids 5-fold. At the reach scale, LW addition enhanced the biomass of invertebrates and shredders in summer but the effects were opposite in winter. LW addition did not affect invertebrate density. The results show the effects of LW restoration on invertebrates to differ among seasons. Positive effects on biomass occur in summer, when retention by LW enhances food availability compared to unrestored reaches, whereas effects are slightly negative in winter, a period of large food availability.

1. Introduction

Freshwaters are among the most threatened ecosystems (Dudgeon, 2010), as they are affected by multiple stressors, such as pollution, exploitation and regulation of water resources, alteration of riparian zones, or channel modification (Walsh et al., 2005; Carpenter et al., 2011). These threats typically lead to declines in biodiversity as well as to impairment of ecosystem processes (Sweeney et al., 2004; Collier and Clements, 2011; von Schiller et al., 2015), which are the basis of important ecosystem services (Costanza et al., 2007). In many regions, legal regulations and massive economic investment resulted in an improvement of water quality in the last decades (Skjelkvåle et al., 2005; EEA, 2012), although river biodiversity continues decreasing (Vörösmarty et al., 2010). Most European rivers show an altered channel form or hydrological regime (Lorenz et al., 2004). Therefore, hydromorphological restoration has gained momentum in the last years, accompanied by a shift from water quality issues to a perspective centred on ecosystem services (Palmer and McDonough, 2013). In particular, many river rehabilitation projects attempted to restore channel complexity (Kail and Hering, 2005; Jähnig et al., 2010) to improve fish habitat (e.g. Roni et al., 2002) or recreate more natural channels (Kail et al., 2007). One frequent habitat restoration tech-

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nique is reintroduction of large wood (LW) into the river channels (Bernhardt et al., 2005). LW has been defined as pieces of wood of a minimum length of 1–3 m and a minimum diameter of 5–20 cm (Wohl et al., 2010). LW restoration has been reported to improve hydraulic retention, enhance accumulation of organic matter (OM), reduce the impact of large floods and provide habitat for many organisms (Gregory et al., 2003). Among others, LW serves as substrate for algae and invertebrates and is used by some insects for oviposition and as an emergence site (Dudley and Anderson, 1982).

Geomorphologic and hydraulic conditions affect the structure and functioning of stream biological communities (Lamouroux et al., 2004; Elosegi and Sabater, 2013). Macroinvertebrates in particular are sensitive to changes in environmental characteristics (Gore et al., 2001; Mackie et al., 2013), and show strong microhabitat preferences (Tachet et al., 2002). Therefore, since LW has been demonstrated to increase physical heterogeneity (Wohl, 2011), many authors assumed LW restoration should increase the diversity of benthic invertebrates (Harrison et al., 2004). Nevertheless, many LW restoration projects failed to promote invertebrate diversity despite increased habitat complexity, thus casting doubts on the effectiveness of these restoration practices (Palmer et al., 2010).

Irrespective of its effects on diversity, LW addition can cause other effects on macroinvertebrate assemblages, such as promoting shredder biomass as a consequence of enhanced OM retention (Entrekin et al., 2009). These effects can be caused by different mechanisms. On the one hand, LW might enhance the quality of existing stream habitats, for instance, by increasing the availability of food or refugia, which would promote invertebrate density at the habitat scale. On the other hand, it might affect the availability of aquatic habitats by promoting the areal cover of some habitats at the detriment of some others, which would result in changes at the reach scale, but not at the habitat scale. It is, of course, also possible that LW addition affects invertebrates at both the habitat and the reach scales, or that the effects at one scale are compensated by those at the other. Therefore, it is important to understand the effects of LW addition at both the habitat and the reach scale. Even more, it is necessary to check the effects in different seasons, as the enhanced retention of OM could be most important in summer, when this food resource is less abundant (Flores et al., 2011).

The objective of the present study was to analyse the effects of a LW restoration project on invertebrate assemblages at both the habitat and the reach scale. At the habitat scale, we predicted that LW addition will:

- increase invertebrate diversity as a consequence of enhanced availability of refugia;
- promote shredder density and biomass as a consequence of increased OM availability;
- At the reach scale, we predicted:
- an increase in total number and biomass of invertebrates.

In addition we predicted:

- these effects to be greater in summer when OM is less abundant in temperate streams, and thus differences between restored and control reaches are likely to be higher.

2. Methods

The study was carried out in 4 headwater streams draining to Añarbe, a reservoir located in the north of the Iberian Peninsula (43°13′N, 1°52′W) within the Aiako Harria Natural Park. It is a rugged area, with mountains of 900 m a.s.l., annual rainfall over 2500 mm and covered by extensive oak and beech forests. Channel width of the studied streams ranged from 3.5 to 13 m and annual average discharge from 0.026 to 2.5 $m^3\ s^{-1}$ (see Flores et al., 2011 for a detailed information of each of the 4 studied streams). Streams in the area have excellent water quality although they lack LW as a legacy of historic harvesting for charcoal and timber. Two reaches were delimited in each stream, with lengths ranging from 100 m in the small streams to 400 m in the largest stream (Fig. 1). The two reaches were separated by at least 200 m. The downstream (E, experimental) reaches were subject to LW restoration, while the upstream (C, control) reaches were used as control. Logs were added into E reaches in volumes ranging from 33 to 239 $m^3\ ha^{-1}$, mimicking the type of structures commonly found in streams in the region (Elosegi et al., 2017). These included isolated logs in the channel, dams (logs or groups of logs spanning the entire stream channel) and deflectors (diagonal logs blocking partially the channel, Kail et al., 2007). Overall, the average length of logs added to the streams was slightly larger than the width of the streams, although in some cases it exceeded channel width in 4–5 m (see Elosegi et al., 2016, 2017 for more details on the added structures). The amount of LW to add was fixed from a regression between channel size and LW abundance published by Bailey et al. (2008) in pristine New Zealand streams under similar climate, topography and vegetation, which corresponds very well to LW abundance found in the few undisturbed streams remaining in the region (unpublished data). No cables or other artificial devices were used to fix LW in place, so the logs were free to move with changes in discharge.

Our study followed a before–after/control–impact (BACI) design: all 8 reaches were sampled in winter and summer one year before wood addition. The sampling campaigns before wood addition were carried out in December 2006 (winter) and July 2007 (summer). On January 2008 wood was added into experimental reaches and one year after all reaches were sampled again (December 2008, July 2009), allowing invertebrate communities to stabilize in experimental reaches during 1 year. Because in the small streams most of the channel was very shallow and flow characteristics varied markedly with discharge, the distinction between pools, runs and riffles was considered inappropriate. Instead, we distinguished invertebrate *habitats* depending on the main substrate types present in the study reaches: coarse inorganic substrate (particles equal or larger than cobbles), fine inorganic substrate (particles equal or finer than pebbles) and organic substrate (OM accumulations, mainly comprising leaf litter and twigs), which we called cobble, gravel and POM, respectively, from the most abundant substrate type. Bedrock cover was negligible in the study reaches. A stratified random approach was followed to sample benthic invertebrates within the 3 main habitats. Ten samples were taken per reach with a Surber net (0.09 m^2 area, 500 μm mesh size), the number of samples per habitat being determined by the relative areal cover of each habitat type. Samples were preserved in 70% ethanol until analysed. In the laboratory, all macroinvertebrates in the samples were sorted, identified and counted. Organisms were identified to family level (except Oligochaeta) and assigned to functional feeding groups following Merritt and Cummins (1996) and Tachet et al. (2002). Previous studies in the same streams (Flores et al., 2011, 2013) showed that most individuals of one family belonged to a single genus. Therefore, individuals were grouped into families and dried at 70 °C over 72 h and weighed in an analytical balance (precision 0.01 mg) to estimate dry mass. Caddisfly cases were removed before drying.

2.1. Data analysis

2.1.1. Habitat level

Diversity was estimated by means of Shannon index. All the statistical analyses in this study had the same structure: *stream* and *habitat* were considered as random factors in the analyses as testing differences among streams and habitat was not an objective for this study, but controlling the variability associated to these