Journal of Hazardous Materials 263P (2013) 139–145

Contents lists available at ScienceDirect

Journal of Hazardous Materials

journal homepage: www.elsevier.com/locate/jhazmat

Irrigation agriculture affects organic matter decomposition in semi-arid terrestrial and aquatic ecosystems

Maite Arroita[a,*], Jesús Causapé[b], Francisco A. Comín[c], Joserra Díez[d], Juan José Jimenez[c], Juan Lacarta[c], Carmen Lorente[c], Daniel Merchán[b], Selene Muñiz[c], Enrique Navarro[c], Jonatan Val[c], Arturo Elosegi[a]

[a] Faculty of Science and Technology, University of the Basque Country, PO Box 644, 48080 Bilbao, Spain
[b] Geological Survey of Spain, C/Manuel Lasala n° 44, 50006 Zaragoza, Spain
[c] Instituto Pirenaico de Ecología (CSIC), Av. Montañana 1005, 50059 Zaragoza, Spain
[d] Faculty of Education, University of the Basque Country, J. Ibañez Sto. Domingo 1, 01006 Vitoria-Gasteiz, Spain

HIGHLIGHTS

- Irrigation agriculture accelerated breakdown of organic matter.
- Changes were higher in terrestrial than in aquatic ecosystems.
- Our results have implications for global carbon budgets.

ARTICLE INFO

Article history:
Received 6 April 2013
Received in revised form 30 May 2013
Accepted 19 June 2013
Available online 28 June 2013

Keywords:
Irrigation agriculture
Ecosystem functioning
Breakdown
Stream
Soil

ABSTRACT

Many dryland areas are being converted into intensively managed irrigation crops, what can disrupt the hydrological regime, degrade soil and water quality, enhance siltation, erosion and bank instability, and affect biological communities. Still, the impacts of irrigation schemes on the functioning of terrestrial and aquatic ecosystems are poorly understood. Here we assess the effects of irrigation agriculture on breakdown of coarse organic matter in soil and water. We measured breakdown rates of alder and holm oak leaves, and of poplar sticks in terrestrial and aquatic sites following a gradient of increasing irrigation agriculture in a semi-arid Mediterranean basin transformed into irrigation agriculture in 50% of its surface. Spatial patterns of stick breakdown paralleled those of leaf breakdown. In soil, stick breakdown rates were extremely low in non-irrigated sites (0.0001–0.0003 day^{-1}), and increased with the intensity of agriculture (0.0018–0.0044 day^{-1}). In water, stick breakdown rates ranged from 0.0005 to 0.001 day^{-1}, and increased with the area of the basin subject to irrigation agriculture. Results showed that irrigation agriculture affects functioning of both terrestrial and aquatic ecosystems, accelerating decomposition of organic matter, especially in soil. These changes can have important consequences for global carbon budgets.

1. Introduction

Human activities are transforming dramatically the world's landscape [1], what probably represents the most important component of global environmental change [2]. Extensive natural areas are being converted for human use, and management practices are intensifying in already human-dominated lands [3]. The area of cultivated land increased globally 466% from 1700 to 1980 [4], to such an extent that croplands and pastures have become one of the largest biomes on the planet [5,6]. More recently, this expansion has slowed down, but, even so, yields keep increasing considerably [7]. This increase is a consequence of the so-called "Green Revolution", which promoted the expansion of high-yielding crops that depend on the use of potentially hazardous materials such as synthetic fertilizers and pesticides, and on the implementation of irrigation and mechanization. As a result, the irrigated surface has doubled during the last 50 years [8,9] and the use of fertilizers increased sevenfold [10]. Moreover, future projections related with global change claim for further expansion of irrigated lands [1,11] in order to compensate the rising temperatures [12], the altered seasonality [13] and the enhanced torrentiality [14]. Highly populated areas with a shortage of water availability, like the Mediterranean region, will be most dependent on the increase of irrigation to ensure their

* Corresponding author at: Faculty of Sciences and Technology, University of the Basque Country, 48080 Bilbao, Spain. Tel.: +34 946015939.
E-mail address: maite.arroita@ehu.es (M. Arroita).

0304-3894/$ – see front matter © 2013 Elsevier B.V. All rights reserved.
http://dx.doi.org/10.1016/j.jhazmat.2013.06.049

agricultural supply, and this will likely imply transformation of non-irrigated croplands as well as of non-agricultural lands.

Modern agricultural techniques result in increased productivity, but often at a high environmental cost leading to unacceptable environmental alterations [3,15]. Irrigation disrupts the hydrological regime [16,17], whereas fertilizers and pesticides pollute soils and nearby aquatic ecosystems [18–20]. These impacts can lead to soil acidification [21], salinization [22], eutrophication and hypoxia [23], water quality issues [24], as well as being a critical source of greenhouse gases [1,3,25]. Moreover, agricultural streams are often associated with high siltation, erosion and bank instability [22,26], which reduce habitat quality [27] and affect the composition and structure of biological communities [28,29]. These environmental impacts could affect the functioning of aquatic and terrestrial ecosystems, eroding their resilience and undermining many ecosystem services [30,31]. Therefore, for a sustainable future it is crucial to understand how ecosystem functioning is altered by irrigation agriculture.

Ecosystem functioning includes a wide variety of processes that change at different spatial and temporal scales and respond to environmental changes specifically [32]. Decomposition, usually measured in terms of leaf litter breakdown, is one of the most broadly used functional variables to assess the impacts of environmental changes on the functioning of both terrestrial [33,34] and aquatic [35,36] ecosystems. Standard wooden sticks are a cost-effective alternative to leaf bags to assess the functional impairment of ecosystems [37–39]. Breakdown of organic matter integrates physical abrasion, microbial colonization, and invertebrate fragmentation [40] and is a key process in the cycling and storage of carbon and nutrients in terrestrial and aquatic ecosystems [41,42]. In addition, it is sensitive to many anthropogenic stressors including flow regulation [43], pollution [44,45], eutrophication [46], changes in riparian vegetation [47,48], or loss of biodiversity [49,50]. However, because each stressor can push breakdown in a different direction and interactions between stressors are common in agro-ecosystems exposed to multiple stressors simultaneously, it is difficult to forecast the overall effect on decomposition [38].

This study assesses the impacts of intensive irrigation agriculture on the functioning of terrestrial and aquatic ecosystems in a semi-arid landscape. The main goal is to compare the breakdown of standard wooden sticks in terrestrial and aquatic sites exposed to a gradient of irrigation intensity. Because this tool has only been validated recently, we also compared the response of wooden sticks with that of classical leaf bags. We hypothesized that: (a) the breakdown of wooden sticks will follow the same patterns than leaf bags; (b) irrigation will promote breakdown in soils, as water availability is strongly limiting in semi-arid regions; and (c) irrigation will also result in higher breakdown in aquatic ecosystems because of higher nutrient contents in the returning water from agricultural fields to streams.

2. Materials and methods

2.1. Study area

Breakdown experiments were performed in the Lerma Creek basin, in the Ebro Depression (Spain). It is located south from the pre-Pyrenees, in the semi-arid Bardenas sector of the Mediterranean region. The climate is continental with an annual precipitation of 468 mm, occurring mostly in spring and fall, whereas severe droughts occur in summer. The annual mean temperature is 14 °C, and the monthly mean varies between 5 °C in January/February and 23 °C in July/August (data from the nearby town of Ejea de los Caballeros).

The Lerma Creek drains a basin of 752 ha that has been recently transformed to irrigation agriculture in almost 50% of its surface. Sprinkler (86%) and drip irrigation (14%) are used to grow mainly maize, winter cereals and tomato. Irrigation in the area enabled double cropping, thanks to an increase in the amount of water, fertilizers and pesticides applied annually per hectare. The common agricultural practices consist of sowing fertilization with NPK compounds, followed by urea (46% N) and/or multiple applications of liquid fertilizers (16% urea, 8% ammonia and 8% nitrate). The predominant active principles of the pesticides used are metolachlor, terbuthylazine and chlorpyrifos, and also included athrazine in the recent past [51].

Irrigation had deep effects on the hydrological regime. The median of daily flow at the Lerma basin increased significantly, the intermittent main creek became perennial and seasonality shifted from a rain-driven to an irrigation-driven pattern [52]. This increased the export of salt from the basin, reduced water salinity due to dilution, and increased the concentration and export of nitrate [53]. Abrahao et al. [51] found slightly elevated values of endrin in the soil, pp'-DDT in water and Ni and Zn in the sediments. However, overall, no serious contamination was detected related to the substances they analyzed (44 pesticides and metabolites, 11 organochlorinated compounds, 17 polycyclic aromatic hydrocarbons, 13 polychlorinated biphenyls and several metals and metalloids).

We selected terrestrial and aquatic sites following a gradient of increasing irrigation intensity (Fig. 1). Terrestrial sites ranged from natural shrubby vegetation above the irrigated area (Shrubland) to irrigated crops (Cropland), passing through non-irrigated pine plantations (Pine Plant), with some differences in soil characteristics (Table 1). Non-irrigated sites were set in the valleys of the Lerma gully, predominated by emerging tertiary materials with a small effective depth limited by limestone and tabular gypsum, and slow drainage. On the contrary, croplands were located in the quaternary glacis consisting of deeper layers of gravels with a sandy loam matrix and layers of tertiary materials [52]. Aquatic sites included the main irrigation canal, and 3 stream reaches with increasing area of irrigation agriculture in their drainage basins, and considerable differences in conductivity and nutrient concentration (Table 2).

2.2. Breakdown experiment

Breakdown experiments were carried out with tongue depressors (15 cm × 1.8 cm × 0.2 cm) made of untreated Canadian poplar wood (*Populus nigra* × *canadiensis*, Moench 1785). Wooden sticks were numbered, oven-dried (70 °C, 72 h), weighed and assembled in strings of 5 with fishing line. Sticks were buried (5–10 cm) in terrestrial sites, and tied to metal bars or roots in water. In order to check whether the response of wooden sticks was consistent to that of traditional leaf bags, we also incubated fine (100 μm) and coarse (5 mm) mesh bags with leaves of holm oak (*Quercus rotundifolia* Lam.) in two of the sampling points in soil (Gully and Cropland2), and fine (100 μm) and coarse (5 mm) mesh bags with leaves of alder (*Alnus glutinosa* (L.) Gaertner) in the four sampling points in water. Fine mesh bags exclude invertebrates, enabling to discern the contribution of microbial and invertebrate communities to the overall decomposition. Freshly fallen holm oak and alder leaves were collected in autumn 2010, air-dried at room temperature (21 °C), enclosed in labelled bags (3 ± 0.05 g in holm oak bags and in fine alder bags; 5 ± 0.05 g in coarse alder bags) and buried and tied like wooden sticks.

All materials were retrieved after 43, 75, 219 and 371 days (5 replicates per site and material), except for alder bags which were collected after 14, 43 and 75 days due to their faster decomposition. The rainfall during the incubation period was 260 mm, lower than the long-term average but within the natural interannual

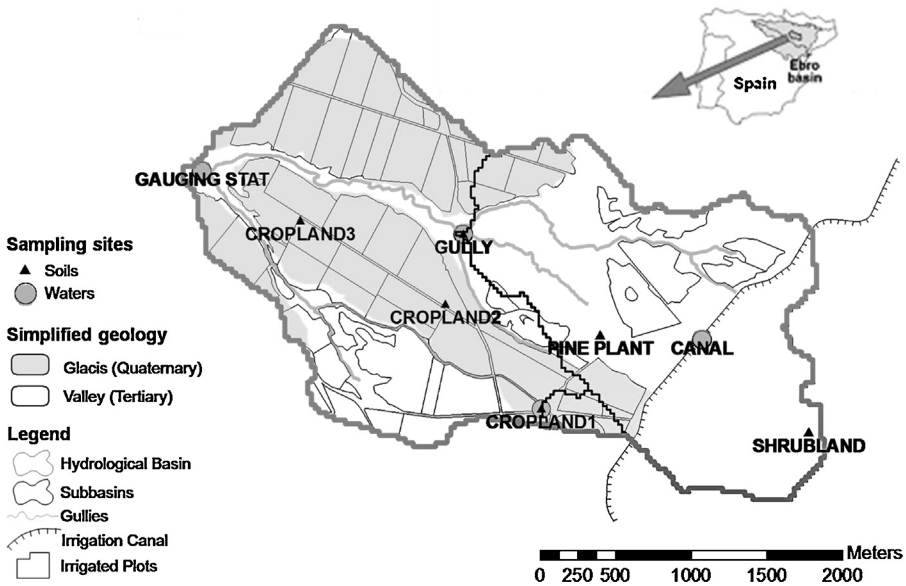

Fig. 1. Physical setting of the Lerma basin and sampling sites.

Table 1
Irrigation water applied, water availability, content of soil organic carbon (SOC), soil inorganic carbon (SIC), total nitrogen (N_{TOT}), $CaCO_3$, sand, silt and clay, pH and number of bacteria alive (B_A) and bacteria harmed or dead (B_{HD}) in the sampling points in soil.

Site	Irrigation (mm)	Wat. avail. (mm)	SOC (%)	SIC (%)	N_{TOT} (%)	$CaCO_3$ (%)	Sand (%)	Silt (%)	Clay (%)	pH	B_A (cell g^{-1})	B_{HD} (cell g^{-1})	B_{HD} (%)
Shrubland	0	7.15	4.39	2.32	0.167	34.6	46.8	33.1	20.1	8.41	7.88E+06	9.55E+04	1.2
Pine	0	7.15	4.81	2.49	0.285	34.1	57.8	31.2	10.9	5.74	3.05E+07	1.14E+06	3.6
Gully	0	10.8	4.10	2.14	0.060	34.3	47.9	38.0	14.1	7.98	9.01E+06	1.53E+05	1.67
Cropland1	726	17.4	3.67	1.55	0.114	29.7	65.2	22.9	11.9	7.98	2.47E+07	8.63E+06	25.9
Cropland2	683	14.3	4.01	1.64	0.101	29.0	73.5	17.4	9.1	8.23	1.25E+07	2.40E+06	16.1
Cropland3	706	12.0	3.31	1.31	0.141	28.3	66.4	22.8	10.9	8.28	4.57E+07	9.86E+06	17.8

variability. Upon removal, sticks and bags were stored in individual zip-lock bags and carried to the laboratory on ice. Samples were rinsed with tap water to remove invertebrates and mineral particles. The material was oven-dried (70 °C, 72 h) and ashed (500 °C, 5 h) to determine the ash free dry mass (AFDM). Initial dry mass of sticks and alder bags incubated in water was corrected for leaching to exclude the effect of this site-independent chemical process, and the remaining AFDM was fitted to the negative exponential model to calculate breakdown rates [54].

2.3. Data treatment

The response of different materials was compared by two-way ANCOVAs (*k* rate as dependent variable, site and material as fixed factors, and time as covariable) and by computing ratios of breakdown rates between different sites for each material [35]. Differences in breakdown rates of wooden sticks were analyzed by means of one-way ANCOVAs (k rate as dependent variable, site as fixed factor, and time as covariable) and post hoc Bonferroni tests. Pearson's correlation coefficients were calculated between breakdown rates and the rest of the variables, and a Principal Component Analysis (PCA) was also performed to identify the main factors explaining the differences among breakdown rates in soil.

3. Results

3.1. Wooden sticks versus leaves

There were large differences among materials in breakdown rates in both terrestrial and aquatic ecosystems (soil: $F_{1,132} = 132.6$, $p < 0.0001$; water: $F_{2,206} = 115.1$, $p < 0.0001$; Fig. 2). Leaf litter in coarse mesh bags broke down the fastest and wooden sticks the slowest, except for Cropland2, where holm oak in fine mesh bags broke down slower than sticks. Differences among sites were also statistically significant (soil: $F_{2,132} = 27.03.6$, $p < 0.0001$; water: $F_{3,206} = 4.35$, $p = 0.005$), but spatial variations depended on the material. The increase in breakdown rate was much higher for coarse mesh than for fine mesh bags, especially in soil, and the interaction site × material was significant (soil: $F_{2,132} = 21.7$, $p < 0.0001$; water: $F_{6,206} = 2.27$, $p = 0.038$). Moreover, ratios of

Table 2
Characteristics of study sites in water (mean ± SE).

SITE	Irrigated area		Cond. (μS cm^{-1})	pH	O_2 (mg L^{-1})	NO_3^- (μg N L^{-1})	PO_4^{-3} (μg P L^{-1})
	ha	%					
Canal	0	0	246 ± 10	8.50 ± 0.08	9.0 ± 1.89	269 ± 89	2.21 ± 0.8
Cropland1	22	87	1906 ± 140	8.45 ± 0.04	7.8 ± 1.87	1689 ± 73	5.38 ± 0.9
Gully	36	35	295 ± 31	7.40 ± 0.90	10.6 ± 0.79	124 ± 95	8.16 ± 3.2
GaugingStat	320	43	3461 ± 319	8.37 ± 0.03	12.1 ± 0.63	22,265 ± 380	3.32 ± 1.5

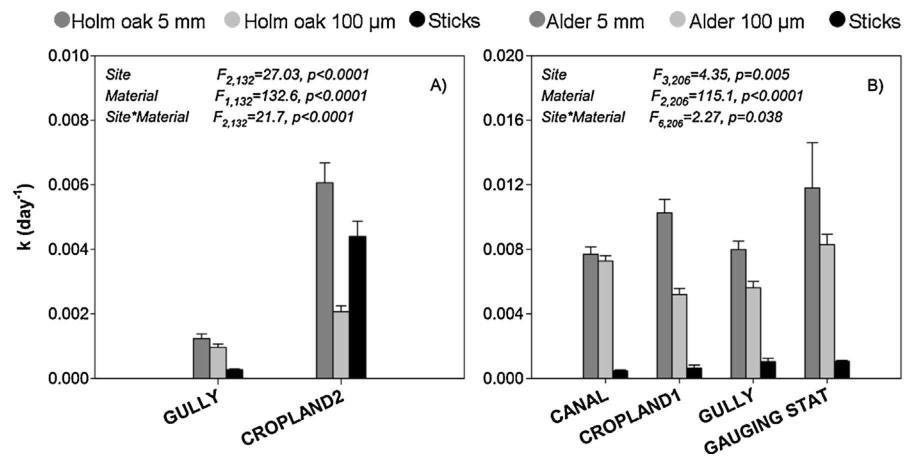

Fig. 2. (A) Breakdown rates in soil (day^{-1}) of holm oak in fine and coarse mesh bags, and of poplar sticks. (B) Breakdown rates in water (day^{-1}) of alder in fine and coarse mesh bags, and of poplar sticks. Error bars show SE. Results from two-way (site × material) ANCOVA are also shown.

Table 3
Ratios of breakdown rates between different sites for each material.

	Bag 5 mm	Bag 100 µm	Sticks
Gully:Cropland2	0.20	0.46	0.06
Canal:Cropland1	0.69	1.56	0.42
Canal:Gully	0.92	1.30	0.20
Canal:Gauging Stat	0.59	0.94	0.43
Cropland1:Gully	1.34	0.83	0.47
Cropland1:GaugingStat	0.85	0.60	1.02
Gully:GaugingStat	0.64	0.72	2.18

breakdown rates computed with different materials were highly variable and showed no consistent pattern (Table 3). Nevertheless, coarse mesh bags and sticks displayed consistent patterns and the site × material interaction became non-significant when fine mesh bags were removed from the analysis (soil: $F_{1,86} = 0.595$, $p = 0.442$; water: $F_{3,134} = 2.09$, $p = 0.105$).

3.2. Impacts of irrigation agriculture

In the terrestrial environment irrigation agriculture accelerated decomposition. Stick breakdown rates were extremely low in non-irrigated sites (0.0001–0.0003 day^{-1}), and increased significantly with the intensity of irrigation (0.0018–0.0044 day^{-1}; Fig. 3). According to the post hoc Bonferroni test, the sampling points could be grouped in three groups regarding breakdown rates: they were slowest in non-irrigated sites, 5–16 times faster in Cropland3, and fastest in Cropland1 and Cropland2.

The regression between water availability and breakdown rates was only marginally significant ($p = 0.054$), being content of $CaCO_3$, sand and silt the only variables significantly correlated with stick breakdown rates in soil, all of them strongly inter-correlated ($p < 0.05$). However, breakdown rates were strongly correlated with the first of the two main components extracted for the PCA analysis, which together accounted for 81.8% of the variance (Fig. 4). The first axis was significantly correlated with water availability, carbon content, granulometry and number of bacteria (both alive and harmed or dead) in soils. The second axis was correlated with N content and pH. Differences between the non-irrigated sites and the three croplands were mainly influenced by the main component, whereas the second axis explained the differences between the non-irrigated sites.

In the aquatic environment stick breakdown rates ranged from 0.0005 to 0.001 day^{-1}, and increased significantly downstream, with the area of the basin subject to irrigation agriculture (one-way ANCOVA: $F_{3,62} = 18.8$, $p < 0.0001$; Fig. 5). The only variable significantly correlated with stick breakdown rates in water was PO_4^{-3}

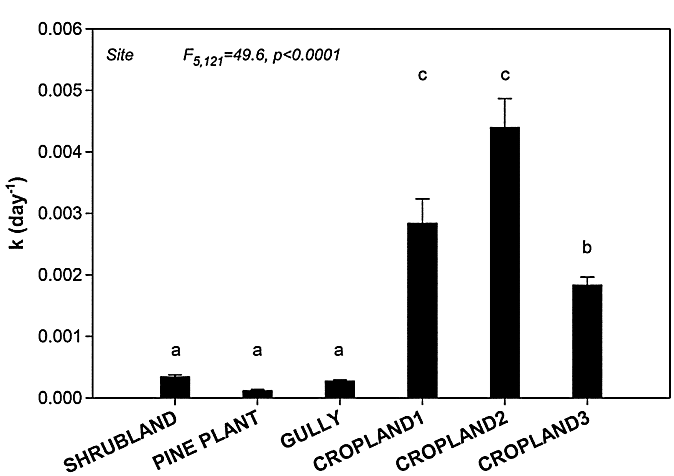

Fig. 3. Breakdown rates (day^{-1}) of wooden sticks in soil. Error bars show SE. Results from post hoc Bonferroni test after one-way ANCOVA with site as a factor are also shown.

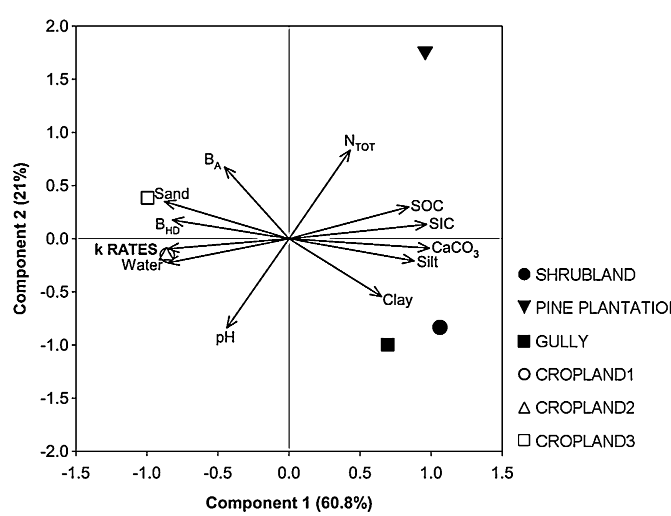

Fig. 4. Principal Component Analysis with breakdown rates, water availability, soil organic carbon (SOC), soil inorganic carbon (SIC), total nitrogen (N_{TOT}), $CaCO_3$, content of sand, silt and clay, pH and number of bacteria alive (B_A) and bacteria harmed or dead (B_{HD}) in each site.

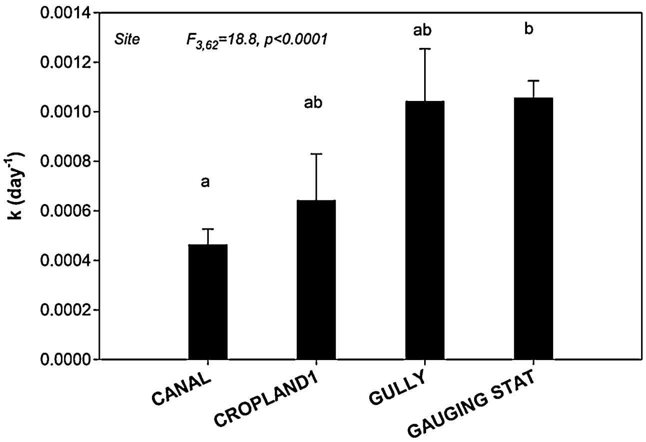

Fig. 5. Breakdown rates (day^{-1}) of wooden sticks in water. Error bars show SE. Results from post hoc Bonferroni test after one-way ANCOVA with site as a factor are also shown.

(R^2 = 0.9, p = 0.049). The regression between NO_3^- and the area under irrigation was statistically significant (R^2 = 0.98, p = 0.007), but neither NO_3^- nor the area under irrigation were strongly correlated with breakdown rates (p > 0.05). Besides, spatial differences in water were not as abrupt as in soil, and according to the post hoc Bonferroni test the sampling points could be grouped in two highly overlapping groups (Fig. 5). Maximum differences in breakdown rate were also smaller in water (5×) than in soil (44×).

4. Discussion

4.1. Wooden sticks versus leaves

The response of wooden sticks to environmental stressors caused by irrigation agriculture was consistent to that of coarse mesh bags in both terrestrial and aquatic ecosystems, thus suggesting their breakdown integrates the effects of microbial activity and consumption by invertebrates. Sticks have been used in streams as indicators of functional impairment [37,38], as they are a cheaper and less time consuming alternative to leaf bags [39]. Our results suggest their use can be expanded to assess soil functioning. Their slow decomposition would make them especially suitable for routine use with one year incubations, less so for shorter time frames. A related question is how to define thresholds for functional impairment based on breakdown rates. Gessner and Chauvet [35] proposed to use the ratio of breakdown rates between test and reference sites, and gave specific values to indicate moderate and strong impairment in streams. Nevertheless, in the present experiment these ratios resulted extremely variable depending on the material used, a result paralleling those reported by Arroita et al. [39] in large streams. Therefore, although wooden sticks offer a cheap, quick and reliable method to measure breakdown in different ecosystem types, clearly more research is needed to adequately define the metrics and the thresholds for functional impairment.

4.2. Impacts of irrigation agriculture

Agricultural practices have deep effects on the characteristics and functioning of soil and nearby water ecosystems [55]. As hypothesized, intensive irrigation agriculture affected the functioning of terrestrial and aquatic ecosystems, accelerating decomposition rates of organic matter irrespective of the type of material. Besides, the fact that irrigation accelerated more the breakdown of leaf litter in coarse than in fine mesh bags indicates that it affected both microbial and invertebrate activities. Our limited dataset precludes attributing enhanced breakdown rates to specific environmental variables, but increased water and nutrient availability seems to be the main factor promoting biological activity in this semi-arid area.

Regarding the terrestrial environment, our results confirm the well-known effect of agriculture on enhancing the cycling of materials in soils. Organic matter breakdown depends on soil moisture, temperature and nutrient contents [56,57], and tends to be promoted in agricultural soils, up to the point of reducing their contents on organic matter [58–64]. In agricultural areas, primary production greatly increases as a consequence of irrigation, thus amounting to a large sequestration of CO_2 in biomass. Nevertheless, the effects of agriculture on the global carbon budget depend largely on the fate of agricultural products. There are no data on agricultural primary production in the study area, but it clearly increased when irrigation was implemented, as the number of crops per year doubled, and low-biomass crops such as wheat were substituted by high biomass ones such as corn. Nevertheless, for most of the irrigated crops such as corn or tomato, only grain or fruits are harvested and most of the biomass is left on the ground. Therefore, the fact that we found lower contents of organic matter in the irrigated soils indicates that breakdown also increases greatly, to the extent of matching primary production.

The most important factor accelerating breakdown of organic matter seems to be water availability [65], especially in semi-arid regions, as moisture promotes microbial activity, thus enhancing decomposition [66,67]. In the Lerma gully, the implementation of irrigation led to a progressive increase in evapotranspiration [52], which has been shown to result in accelerated breakdown [68]. Furthermore, artificial remoistening of dried soil by irrigation disrupts soil structures, increasing substrate availability for bacteria, further enhancing microbial activity [69]. Concerning temperature, irrigation agriculture has been shown to contribute to the reduction of the diurnal temperature range [70], what could slow down decomposition [71]. However, our results suggest that factors enhancing breakdown (moisture) were much stronger than factors decelerating it (reduced temperature oscillations).

Other potential factors behind enhanced breakdown are soil texture, ploughing, and fertilization. Soil texture affects decomposition [69] through their effects on vertical migration of organisms and aeration [72], both of them decisive factors in breakdown [73]. The fact that breakdown rate was located close to the firs axis in our PCA, and not close to the nutrient contents, suggests soil texture to be a determinant factor of the observed differences. Nevertheless, a potential confounding factor is the fact that non-irrigated sites in the Lerma basin are mainly located on tertiary clays, whereas most croplands are located in more sandy quaternary glacis. Their composition makes quaternary glacis more suitable for irrigation, explains part of the differences in soil texture, and might play a role in breakdown rates. A point of concern is the higher percentage of dead bacterial cells in croplands in the present experiment, which might reflect soil toxicity, probably as a consequence of pesticides and heavy metals, which appear in fairly high concentrations in the Lerma basin [51]. The combination of heavy metals and organic pollutants causes synergistic cytotoxic effects on microorganisms [74].

Although we lack any data previous to the implementation of irrigation agriculture in the Lerma basin, the observed spatial patterns suggest that it enhanced the breakdown of organic matter. On one hand, irrigation converted the Lerma Creek from intermittent to perennial [52], and it is known that stream intermittency results in slow breakdown [75–77]. On the other hand, the spatial gradient of intensification produced an acceleration of breakdown rates. Nutrient concentrations in water enhance breakdown rates [78–80] up to a point beyond which they can fall again [36]. Additionally, agricultural practices can have multiple effects on river ecosystems, from altered hydrology to siltation or pesticide toxicity, all of which can have important consequences for ecosystem functioning [81,82]. Therefore, it is not rare to find large variability on breakdown rates in rivers subject to multiple stressors [38]. Nevertheless, our results show that in the Lerma Creek the spatial differences can be explained by nutrient concentration, thus suggesting other factors to play a minor role.

Whatever the exact controlling mechanisms, our results show irrigation agriculture to enhance the breakdown of organic matter both in soils and in adjacent creeks. Given the prevalence and the increasing intensity of agricultural uses, any relative change in global rates of primary production and breakdown might have important consequences in carbon fluxes and climate [83], as both soils [84] and inland waters [85] play an important role in global carbon budgets.

Acknowledgements

This work has been supported by the Spanish Ministry of Economy and Competitiveness through the projects CSD2009-00065, CGL2009-13410-C02-01/02 and BFU2010-22053. We also acknowledge financial support in terms of predoctoral grants from the Basque Government (M. Arroita).

References

[1] R.S. DeFries, J.A. Foley, G.P. Asner, Land-use choices: balancing human needs and ecosystem function, Front. Ecol. Environ. 2 (2004) 249–257.
[2] P.M. Vitousek, Beyond global warming: ecology and global change, Ecology 75 (1994) 1861–1876.
[3] J.A. Foley, R. DeFries, G.P. Asner, C. Barford, G. Bonan, S.R. Carpenter, F.S. Chapin, M.T. Coe, G.C. Daily, H.K. Gibbs, J.H. Helkowski, T. Holloway, E.A. Howard, C.J. Kucharik, C. Monfreda, J.A. Patz, I.C. Prentice, N. Ramankutty, P.K. Snyder, Global consequences of land use, Science 309 (2005) 570–574.
[4] W.B. Meyer, B.L.I.I. Turner, Human population growth and global land-use/cover change, Annu. Rev. Ecol. Syst. 23 (1992) 39–61.
[5] N. Ramankutty, J.A. Foley, Estimating historical changes in global land cover: croplands from 1700 to 1992, Global Biogeochem. Cycles 13 (1999) 997–1027.
[6] G.P. Asner, A.J. Elmore, L.P. Olander, R.E. Martin, A.T. Harris, Grazing systems, ecosystem responses, and global change, Annu. Rev. Environ. Resour. 29 (2004) 261–299.
[7] R.L. Naylor, Energy and resource constraints on intensive agricultural production, Annu. Rev. Energy Environ. 21 (1996) 99–123.
[8] P.H. Gleick, Water use, Annu. Rev. Environ. Resour. 28 (2003) 275–314.
[9] FAO, The Statistic Division of the FAO, Food and Agriculture Organization of the United Nations, Rome, Italy, 2009.
[10] D. Tilman, J. Fargione, B. Wolff, C. D'Antonio, A. Dobson, R. Howarth, D. Schindler, W.H. Schlesinger, D. Simberloff, D. Swackhamer, Forecasting agriculturally driven global environmental change, Science 292 (2001) 281–284.
[11] B.R. Scanlon, I. Jolly, M. Sophocleous, L. Zhang, Global impacts of conversions from natural to agricultural ecosystems on water resources: quantity versus quality, Water Resour. Res. 43 (2007) W03437.
[12] A.L. Gibelin, M. Déqué, Anthropogenic climate change over the Mediterranean region simulated by a global variable resolution model, Clim. Dynam. 20 (2003) 327–339.
[13] E. Sánchez, C. Gallardo, M.A. Gaertner, A. Arribas, M. Castro, Future climate extreme events in the Mediterranean simulated by a regional climate model: a first approach, Global Planet. Change 44 (2004) 163–180.
[14] J. Räisänen, U. Hansson, A. Ullerstig, R. Döscher, L.P. Graham, C. Jones, H.E.M. Meier, P. Samuelsson, U. Willén, European climate in the late twenty-first century: regional simulations with two driving global models and two forcing scenarios, Clim. Dynam. 22 (2004) 13–31.
[15] J. Rockström, W. Steffen, K. Noone, Å. Persson, F.S. Chapin III, E.F. Lambin, T.M. Lenton, M. Scheffer, C. Folke, H.J. Schellnhuber, B. Nykvist, C.A. de Wit, T. Hughes, S. van der Leeuw, H. Rodhe, S. Sörlin, P.K. Snyder, R. Costanza, U. Svedin, M. Falkenmark, L. Karlberg, R.W. Corell, V.J. Fabry, J. Hansen, B. Walker, D. Liverman, K. Richardson, P. Crutzen, J.A. Foley, A safe operating space for humanity, Nature 461 (2009) 472–475.
[16] J.K. Böhlke, Groundwater recharge and agricultural contamination, Hydrogeol. J. 10 (2002) 153–179.
[17] I. García-Garizábal, J. Causapé, Influence of irrigation water management on the quantity and quality of irrigation return flows, J. Hydrol. 385 (2010) 36–43.
[18] Z.L. He, X.E. Yang, P.J. Stoffella, Trace elements in agroecosystems and impacts on the environment, J. Trace Elem. Med. Biol. 19 (2005) 125–140.
[19] P.N.M. Schipper, L.T.C. Bonten, A.C.C. Plette, S.W. Moolenaar, Measures to diminish leaching of heavy metals to surface waters from agricultural soils, Desalination 226 (2008) 89–96.
[20] S.C. Sheppard, C.A. Grant, M.I. Sheppard, R. de Jong, J. Long, Risk indicator for agricultural inputs of trace elements to Canadian soils, J. Environ. Qual. 38 (2009) 919–932.
[21] R.F. Follett, J.A. Delgado, Nitrogen fate and transport in agricultural systems, J. Soil Water Conserv. 57 (2002) 402–408.
[22] P.A. Matson, W.J. Parton, A.G. Power, M.J. Swift, Agricultural intensification and ecosystem properties, Science 277 (1997) 504–509.
[23] D. Scavia, S.B. Bricker, Coastal eutrophication assessment in the United States, Biogeochemistry 79 (2006) 187–208.
[24] A.R. Townsend, R.W. Howarth, F.A. Bazzaz, M.S. Booth, C.C. Cleveland, S.K. Collinge, A.P. Dobson, P.R. Epstein, E.A. Holland, D.R. Keeney, M.A. Mallin, C.A. Rogers, P. Wayne, A.H. Wolfe, Human health effects of a changing global nitrogen cycle, Front. Ecol. Environ. 1 (2003) 240–246.
[25] IPCC, Climate Change 2007: Synthesis Report. Contributions of Working Groups I, II and III to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change, IPCC, Geneva, Switzerland, 2007.
[26] J.D. Allan, D.L. Erickson, J. Fay, The influence of catchment land use on stream integrity across multiple spatial scales, Freshwater Biol. 37 (1997) 149–161.
[27] D. Genito, W.J. Gburek, A.N. Sharpley, Response of stream macroinvertebrates to agricultural land cover in a small watershed, J. Freshwater Ecol. 17 (2002) 109–119.
[28] J. Velasco, A. Millán, J. Hernández, C. Gutiérrez, P. Abellán, D. Sánchez, M. Ruiz, Response of biotic communities to salinity changes in a Mediterranean hypersaline stream, Saline Syst. 2 (2006).
[29] C. Gutiérrez-Cánovas, J. Velasco, A. Millán, Effects of dilution stress on the functioning of a saline Mediterranean stream, Hydrobiologia 619 (2009) 119–132.
[30] J.J. Schröder, D. Scholefield, F. Cabral, G. Hofman, The effects of nutrient losses from agriculture on ground and surface water quality: the position of science in developing indicators for regulation, Environ. Sci. Policy 7 (2004) 15–23.
[31] A. Bondeau, P.C. Smith, S. Zaehle, S. Schaphoff, W. Lucht, W. Cramer, D. Gerten, H. Lotze-Campen, C. Müller, M. Reichstein, B. Smith, Modelling the role of agriculture for the 20th century global terrestrial carbon balance, Global Change Biol. 13 (2007) 679–706.
[32] M.A. Palmer, C.M. Febria, The heartbeat of ecosystems, Science 336 (2012) 1393–1394.
[33] C.E. Prescott, Litter decomposition: what controls it and how can we alter it to sequester more carbon in forest soils? Biogeochemistry 101 (2010) 133–149.
[34] M. Makkonen, M.P. Berg, I.T. Handa, S. Hättenschwiler, J. van Ruijven, P.M. van Bodegom, R. Aerts, Highly consistent effects of plant litter identity and functional traits on decomposition across a latitudinal gradient, Ecol. Lett. 15 (2012) 1033–1041.
[35] M.O. Gessner, E. Chauvet, A case for using litter breakdown to assess functional stream integrity, Ecol. Appl. 12 (2002) 498–510.
[36] G. Woodward, M.O. Gessner, P.S. Giller, V. Gulis, S. Hladyz, A. Lecerf, B. Malmqvist, B.G. McKie, S.D. Tiegs, H. Cariss, M. Dobson, A. Elosegi, V. Ferreira, M.A.S. Graça, T. Fleituch, J.O. Lacoursière, M. Nistorescu, J. Pozo, G. Risnoveanu, M. Schindler, A. Vadineanu, L.B.M. Vought, E. Chauvet, Continental-scale effects of nutrient pollution on stream ecosystem functioning across Europe, Science 336 (2012) 1438–1440.
[37] R.G. Young, C.D. Matthaei, C.R. Townsend, Organic matter breakdown and ecosystem metabolism: functional indicators for assessing river ecosystem health, J. North Am. Benthol. Soc. 27 (2008) 605–625.
[38] I. Aristi, J.R. Díez, A. Larrañaga, A. Navarro-Ortega, D. Barceló, A. Elosegi, Assessing the effects of global change on the functioning of Mediterranean rivers using poplar wood breakdown, Sci. Total Environ. 440 (2012) 272–279.
[39] M. Arroita, I. Aristi, L. Flores, A. Larrañaga, J. Díez, J. Mora, A.M. Romaní, A. Elosegi, The use of wooden sticks to assess stream ecosystem functioning: comparison with leaf breakdown rates, Sci. Total Environ. 440 (2012) 115–122.
[40] M. Hieber, M.O. Gessner, Contribution of stream detrivores, fungi, and bacteria to leaf breakdown based on biomass estimates, Ecology 83 (2002) 1026–1038.
[41] R. Aerts, H. De Caluwe, Nutritional and plant-mediated controls on leaf litter decomposition of *Carex* species, Ecology 78 (1997) 244–260.
[42] J.L. Tank, E.J. Rosi-Marshall, N.A. Griffiths, S.A. Entrekin, M.L. Stephen, A review of allochthonous organic matter dynamics and metabolism in streams, J. North Am. Benthol. Soc. 29 (2010) 118–146.
[43] A. Martínez, A. Larrañaga, A. Basaguren, J. Pérez, C. Mendoza-Lera, J. Pozo, Stream regulation by small dams affects benthic macroinvertebrate communities: from structural changes to functional implications, Hydrobiologia. doi:10.1007/s10750-013-1459-z.
[44] C. Pascoal, M. Pinho, F. Cássio, P. Gomes, Assessing structural and functional ecosystem condition using leaf breakdown: studies on a polluted river, Freshwater Biol. 48 (2003) 2033–2044.
[45] J.C. Niemeyer, M.A. Nogueira, G.M. Carvalho, S.J. Cohin-De-Pinho, U.S. Outeiro, G.G. Rodrigues, E.M. da Silva, J.P. Sousa, Functional and structural parameters to assess the ecological status of a metal contaminated area in the tropics, Ecotoxicol. Environ. Safety 86 (2012) 188–197.
[46] S. Hladyz, M.O. Gessner, P.S. Giller, J. Pozo, G. Woodward, Resource quality and stoichiometric constraints on stream ecosystem functioning, Freshwater Biol. 54 (2009) 957–970.
[47] S.E. Attignon, D. Weibel, T. Lachat, B. Sinsin, P. Nagel, R. Peveling, Leaf litter breakdown in natural and plantation forests of the Lama Forest Reserve in Benin, Appl. Soil Ecol. 27 (2004) 109–124.
[48] A. Lecerf, M. Dobson, C.K. Dang, E. Chauvet, Riparian plant species loss alters trophic dynamics in detritus-based stream ecosystems, Oecologia 146 (2005) 432–442.

[49] D.A. Heemsbergen, M.P. Berg, M. Loreau, J.R. van Hal, J.H. Faber, H.A. Verhoef, Biodiversity effects on soil processes explained by interspecific functional dissimilarity, Science 306 (2004) 1019–1020.
[50] M.O. Gessner, C.M. Swan, C.K. Dang, B.G. McKie, R.D. Bardgett, D.H. Wall, S. Hättenschwiler, Diversity meets decomposition, Trends Ecol. Evol. 25 (2010) 372–380.
[51] R. Abrahao, J. Sarasa, J. Causapé, I. García-Garizábal, J.L. Ovelleiro, Influence of irrigation on the occurrence of organic and inorganic pollutants in soil, water and sediments of a Spanish agrarian basin (Lerma), Span. J. Agric. Res. 9 (2011) 124–134.
[52] R. Abrahao, J. Causapé, I. García-Garizábal, D. Merchán, Implementing irrigation: water balances and irrigation quality in the Lerma basin (Spain), Agric. Water Manage. 102 (2011) 97–104.
[53] R. Abrahao, J. Causapé, I. García-Garizábal, D. Merchán, Implementing irrigation: salt and nitrate exported from the Lerma basin (Spain), Agric. Water Manage. 102 (2011) 105–112.
[54] R.C. Petersen, K.W. Cummins, Leaf processing in a woodland stream, Freshwater Biol. 4 (1974) 343–368.
[55] G. Zalidis, S. Stamatiadis, V. Takavakoglou, K. Eskridge, N. Misopolinos, Impacts of agricultural practices on soil and water quality in the Mediterranean region and proposed assessment methodology, Agric. Ecosyst. Environ. 88 (2002) 137–146.
[56] V. Meentemeyer, Meteorologic control of litter decomposition with an emphasis on tropical environments, in: M.V. Reddy (Ed.), Soil Organisms and Litter Decomposition in the Tropics, Westview Press, Boulder, CO, USA, 1995, pp. 153–182.
[57] J.T.A. Verhoeven, E. Toth, Decomposition of Carex and Sphagnum litters in fens: effect of litter quality and inhibition by living tissue homogenates, Soil Biol. Biochem. 27 (1995) 271–275.
[58] R.J. Haynes, M.H. Beare, Aggregation and organic carbon storage in mesothermal, humid soils, in: B.A. Stewart, M.R. Carter (Eds.), Advances in Soil Science. Structure and Organic Matter Storage in Agricultural Soils, CRC Lewis, Boca Raton, 1996, pp. 213–262.
[59] K. Paustian, H.P. Collins, E.A. Paul, Management controls on soil carbon, in: E.A. Paul, K. Paustian, E.T. Elliott, C.V. Cole (Eds.), Soil Organic Matter in Temperate Agroecosystems, CRC Lewis, Boca Raton, 1997, pp. 15–49.
[60] I.A. Jaiyeoba, Changes in soil properties due to continuous cultivation in Nigerian semiarid Savannah, Soil Till. Res. 70 (2003) 91–98.
[61] D.P. Heenan, K.Y. Chan, P.G. Knight, Long-term impact of rotation, tillage and stubble management on the loss of soil organic carbon and nitrogen from a Chromic Luvisol, Soil Till. Res. 76 (2004) 59–68.
[62] D.D. Malo, T.E. Schumacher, J.J. Doolittle, Long-term cultivation impacts on selected soil properties in the northern Great Plains, Soil Till. Res. 81 (2005) 277–291.
[63] H.J. Koch, N. Stockfisch, Loss of soil organic matter upon ploughing under a loess soil after several years of conservation tillage, Soil Till. Res. 86 (2006) 73–83.
[64] P. Szilassi, G. Jordan, A. van Rompaey, G. Csillag, Impacts of historical land use changes on erosion and agricultural soil properties in the Kali Basin at Lake Balaton, Hungary, Catena 68 (2006) 96–108.
[65] P. Lavelle, E. Blanchart, A. Martin, S. Martin, A. Spain, F. Toutain, I. Barois, R. Schaefer, A hierarchical model for decomposition in terrestrial ecosystems: application to soils of the humid tropics, Biotropica 25 (1993) 130–150.
[66] E.E. Bontti, J.P. Decant, S.M. Munson, M.A. Gathany, A. Przeszlowska, M.L. Haddix, S. Owens, I.C. Burke, W.J. Parton, M.E. Harmon, Litter decomposition in grasslands of Central North America (US Great Plains), Global Change Biol. 15 (2009) 1356–1363.
[67] O. Butenschoen, S. Scheu, N. Eisenhauer, Interactive effects of warming, soil humidity and plant diversity on litter decomposition and microbial activity, Soil Biol. Biochem. 43 (2011) 1902–1907.
[68] R. Aerts, Climate, leaf litter chemistry and leaf litter decomposition in terrestrial ecosystems: a triangular relationship, Oikos 79 (1997) 439–449.
[69] J.A. Van Veen, P.J. Kuikman, Soil structural aspects of decomposition of organic matter by micro-organisms, Biogeochemistry 11 (1990) 213–233.
[70] E. Kalnay, M. Cai, Impact of urbanization and land-use change on climate, Nature 423 (2003) 528–531.
[71] C.K. Dang, M. Schindler, E. Chauvet, M.O. Gessner, Temperature oscillation coupled with fungal community shifts can modulate warming effects on litter decomposition, Ecology 90 (2009) 122–131.
[72] A. Alaoui, J. Lipiec, H.H. Gerke, A review of the changes in the soil pore system due to soil deformation: a hydrodynamic perspective, Soil Till. Res. 115–116 (2011) 1–15.
[73] P. Schjønning, I.K. Thomsen, J.P. Møberg, H. de Jonge, K. Kristensen, B.T. Christensen, Turnover of organic matter in differently textured soils: I. Physical characteristics of structurally disturbed and intact soils, Geoderma 89 (1999) 177–198.
[74] Q. Lin, Z. Wang, S. Ma, Y. Chen, Evaluation of dissipation mechanisms by *Lolium perenne* L. and *Raphanus sativus* for pentachlorophenol (PCP) in copper co-contaminated soil, Sci. Total Environ. 368 (2006) 814–822.
[75] A.J. Boulton, Parallels and contrasts in the effects of drought on stream macroinvertebrate assemblages, Freshwater Biol. 48 (2003) 1173–1185.
[76] A. Bruder, E. Chauvet, M.O. Gessner, Litter diversity, fungal decomposers and litter decomposition under simulated stream intermittency, Funct. Ecol. 25 (2011) 1269–1277.
[77] T. Datry, R. Corti, C. Claret, M. Philippe, Flow intermittence controls leaf litter breakdown in a French temporary alluvial river: the drying memory, Aquat. Sci. 73 (2011) 471–483.
[78] J.L. Greenwood, A.D. Rosemond, J.B. Wallace, W.F. Cross, H.S. Weyers, Nutrients stimulate leaf breakdown rates and detritivore biomass: bottom-up effects via heterotrophic pathways, Oecologia 151 (2007) 637–649.
[79] J.P. Benstead, A.D. Rosemond, W.F. Cross, J.B. Wallace, S.L. Eggert, K. Suberkropp, V. Gulis, J.L. Greenwood, C.J. Tant, Nutrient enrichment alters storage and fluxes of detritus in a headwater stream ecosystem, Ecology 90 (2009) 2556–2566.
[80] J. Pérez, E. Descals, J. Pozo, Aquatic hyphomycete communities associated with decomposing alder leaf litter in reference headwater streams of the Basque Country (northern Spain), Microb. Ecol. 64 (2012) 279–290.
[81] D.K. Niyogi, W.M. Lewis Jr., D.M. McKnight, Litter breakdown in mountain streams affected by mine drainage: biotic mediation of abiotic controls, Ecol. Appl. 11 (2001) 506–516.
[82] A. Elosegi, S. Sabater, Effects of hydromorphological impacts on river ecosystem functioning: a review and suggestions for assessing ecological impacts, Hydrobiologia, doi:10.1007/s10750-012-1226-6.
[83] E.A. Davidson, I.A. Janssens, Temperature sensitivity of soil carbon decomposition and feedbacks to climate change, Nature 440 (2006) 165–173.
[84] R. Lal, Soil carbon sequestration impacts on global climate change and food security, Science 304 (2004) 1623–1627.
[85] T.J. Battin, S. Luyssaert, L.A. Kaplan, A.K. Aufdenkampe, A. Richter, L.J. Tranvik, The boundless carbon cycle, Nat. Geosci. 2 (2009) 598–600.