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ORIGINAL ARTICLE

Impact of potash mining in streams: the Llobregat basin (northeast Spain) as a case study

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ABSTRACT

Potash mining is significantly increasing the salt concentration of rivers and streams due to lixiviates coming from the mine tailings. In the present study, we have focused on the middle Llobregat basin (northeast Spain), where an important potash mining activity exists from the beginning of the XX century. Up to 50 million tonnes of saline waste have been disposed in the area, mainly composed of sodium chloride. We assessed the ecological status of streams adjacent to the mines by studying different physicochemical and hydromorphological variables, as well as aquatic macroinvertebrates. We found extraordinary high values of salinity in the studied streams, reaching conductivities up to 132.4 mS/cm. Salt-polluted streams were characterized by a deterioration of the riparian vegetation and the fluvial habitat. Both macroinvertebrate richness and abundance decreased with increasing salinity. In the most polluted stream only two families of macroinvertebrates were found: Ephydridae and Ceratopogonidae. According to the biotic indices IBMWP and IMMi-T, none of the sites met the requirements of the Water Framework Directive (WFD; *i.e.*, good ecological status). Overall, we can conclude that potash-mining activities have the potential to cause severe ecological damage to their surrounding streams. This is mainly related to an inadequate management of the mine tailings, leading to highly saline runoff and percolates entering surface waters. Thus, we urge water managers and policy makers to take action to prevent, detect and remediate salt pollution of rivers and streams in potash mining areas.

Key words: Potash mining; freshwater salinization; Llobregat basin; ecological status; aquatic macroinvertebrates; biotic indices.

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INTRODUCTION

The increase of river salinity by human activities (*i.e.*, secondary salinization) is considered as a major global impact to rivers around the world (Cañedo-Argüelles *et al.*, 2016; Schulz, 2011), and climate change and increased water demand will exacerbate it in the future (Cañedo-Argüelles *et al.*, 2013). Secondary salinization has many different causes, with agriculture and mining being the most important (Cañedo-Argüelles *et al.*, 2013). One of the mining activities directly related to freshwater salinization is potash mining, which has been especially intense in some European regions as the Werra Basin in Germany (Bäthe and Coring, 2011; Braukmann and Böhme, 2011) or the Llobregat Basin in Spain (Gorostiza, 2014). Here we focus on the latter to assess the potential impact of potash mining on the biological quality of rivers and streams.

Potash salts were discovered in the middle part of the Llobregat Basin (when river cross the Catalan Potassic region) in 1912, and their extraction began in the following decade (Gorostiza, 2014). Potash mines are exploited to obtain potassium salts (silvite and carnallite), which are mainly used to produce agricultural fertilizers. During the extraction process important wastes are produced and stockpiled in mine tailings (Ribera *et al.*, 2009; Soler *et al.*, 2011). In the Catalan Potassic region, approximately 3 kg of waste salts are produced for each kg of potash obtained, and they are deposited in large mine tailings (Gorostiza, 2014). Mine tailings are mainly composed of sodium chloride (more than 80%), together with potassium and magnesium chloride, water and traces of other chemical compounds (http://www.lasequia.cat/montsalat/). According to different studies (Otero and Soler, 2002; Rovira *et al.*, 2008, 2006) lixiviates coming from these mine tailings permeate into groundwater and enter nearby rivers, increasing their salt concentrations.

Once the salts enter the river they have the potential to severely damage the ecosystem. Freshwater organisms need to maintain a balance between the concentration of salts in the external medium (*i.e.*, the river water) and their body (Bradley, 2008). This process is called osmoregulation, and it has a high energetic cost (Goolish and Burton, 1989; Komnick, 1982; Sébert *et al.*, 1997). The energy spent in osmoregulation cannot be spent in other metabolic processes, leading to sub-lethal effects on the organisms, *e.g.* delayed growth (Hassell *et al.*, 2006; Kennedy *et al.*, 2003). When salt concentration is too high the organisms are no longer able to keep the osmotic balance within their cells and they die (Cañedo-Argüelles *et al.*, 2013). These effects can scale-up to the ecosystem level; for example, the extinction of shredder aquatic insects (*e.g.*, Trichoptera and Plecoptera) can lead to a significant reduction of leaf decomposition in the river (Cañedo-Argüelles *et al.*, 2014; Schäfer *et al.*, 2012). Among different freshwater organisms, aquatic macroinvertebrates (especially insects) can be highly sensitive to salt pollution, and their communities can be significantly altered by stream salinization (Cañedo-Argüelles *et al.*, 2013). In the case of the Llobregat River basin, salt mining has been proved to have the potential to strongly modify macroinvertebrate communities (Cañedo-Argüelles *et al.*, 2012, 2014, 2015; Prat and Rieradevall, 2006) and it is considered as one of the main anthropic pressures on the ecosystem (Munné *et al.*, 2012). In this basin, increased salinity has been reported to affect the composition of the fauna due to its toxicity (Damásio *et al.*, 2008). For example, the stress caused by salt pollution disrupted the metabolic activity of one of the most abundant and tolerant species in the Llobregat River: the Trichoptera *Hydropsyche exocellata* (Damásio *et al.*, 2011). However, current information on the impact of salt mines on rivers and streams within the area is limited to the main course of Llobregat River; detailed information on the impact of salt pollution on the streams that surround the mines is lacking.

According to the Water Framework Directive (WFD), the ecological status of water bodies has to be ultimately assessed by its biological conditions. Several biotic indices based on macroinvertebrates are currently used by water authorities to establish the ecological status of Mediterranean streams (Munné *et al.*, 2012); *e.g.*, the IASPT (Jáimez-Cuéllar *et al.*, 2002) and IBMWP (Alba-Tercedor *et al.,* 2002) and the multimetric indices IMMi-L and IMMi-T, based on qualitative or quantitative data respectively (Munné and Prat, 2009). The IBMWP and the IMMi-T indices have been intercalibrated with the European intercalibration index ICM-Star and they are the official methods used by the Catalan Water District authorities to assess the ecological status of rivers and streams. However, the sensitivity of these biotic indices has been usually tested for organic or mixed pollutions and its ability to evaluate other kind of pressures, such as physical impacts in headwater streams (Ladrera and Prat, 2013) or salinity, is not clear. Regarding salinity, it is uncertain if IBMWP can accurately reflect river community alterations associated to mining activity (García-Criado *et al.*, 1999). For example, Cañedo-Argüelles *et al.* (2012) tested the response of a variety of macroinvertebrate-based biotic indices and found that only IMMi-T responded to salinity and only after 72 hours of exposure to high salinity values (5 mS cm^{-1}). However, this study was carried out under controlled conditions in a stream mesocosm, examining a macroinvertebrate community that was a sub-set of the community found in natural streams. Thus, it remains unclear if macroinvertebrate-indices can reflect the ecological degradation in streams affected by potash mining activities.

The objectives of the present study were: i) to determine the physicochemical and hydromorphological conditions in streams located nearby mine tailings in the Llobregat basin; ii) to explore the impact of salt pollution (related to the potash mines) on the macroinvertebrate communities; and iii) to assess the ability of biotic indices based on macroinvertebrate communities (within the context of the WFD) to detect salt pollution of streams.

METHODS

Study area

The study was carried out in March 2012 in the middle Llobregat basin (70 kilometers NW of the city of Barcelona), where several potash mines and mine tailings exist. Three important mine tailings can be found within our study area (Fig. 1): 'La Botjosa' and 'Vilafruns' are closed and contain between 3 and 4 million tonnes of waste respectively (Soler *et al.*, 2011), whereas 'El Cogulló' (Fig. 2A, 2B) remains active and contains more than 50 million tonnes of waste (Gorostiza, 2014). The study sites P1-P4 were located in tributaries around this large mine tailing and the site P5 in the main course of the Llobregat River, covering a gradient of exposure to saline lixiviates coming from the mine tailings (Fig. 1).

Environmental variables

Different physicochemical and morphological variables were recorded at each site. Water temperature (°C), dissolved oxygen (ppm) and electrical conductivity (mS cm^{-1}) were measured *in situ* using selective electrodes. We used electrical conductivity as a simple way to measure water salinity, since dissolved ions can be expressed as the capacity of a solution to transmit electrical current (Cañedo-Argüelles *et al.*, 2013; Pawlowicz, 2008) and salinity is linearly correlated with conductivity in the Llobregat River (Prat *et al.*, 1983). At the same time, two hydromorphological variables were analysed at each site: i) The quality of riparian habitat was characterized using the four components of the Riparian Forest Quality Index (QBR), developed by Munné *et al.* (2003), which evaluates the riparian vegetation according to four aspects: total riparian vegetation cover, cover structure, cover quality and channel alterations; ii) the fluvial habitat was characterized using the IHF River Habitat Index (Pardo *et al.*, 2002), which measures habitat heterogeneity according to 7 criteria: substrate embeddedness, rapid frequency, substrate composition, velocity/depth conditions, % of shading, heterogeneity components and in-channel vegetation cover.

Macroinvertebrate sampling and biotic indices

Macroinvertebrates were collected using a 250 µm surber net according to the MIQU sampling protocol designed by the FEM research group of the University of Barcelona (Nuñez and Prat, 2010). This method takes 8 surber (30 x 30 cm) samples in dominant substrates (*i.e.*, covering more than 5% of the sampling area) and 4 in marginal ones (*i.e.*, covering less than 5% of the sampling area). Samples were taken proportionally to the presence of 11 different substrates classes, and were stored in two buckets, one for dominant and another for marginal substrates. Macroinvertebrates were preserved in 4% formaldehyde and taken to the laboratory to be identified. The taxonomic identification was performed at the family level, except for Oligochaeta, Hydracharina and Ostracoda. If necessary, sub-sampling was done in the sorting process and at least 300 individuals per sample were counted.

Different biotic indices adapted to Mediterranean Rivers were calculated (IBMWP, IASPT, IMMi-L and IMMi-T) using macroinvertebrate data (Tab. 1). The Iberian Biological Monitoring Working Party (IBMWP) is based on tolerance of macroinvertebrate families to water quality and river alteration (Alba-Tercedor *et al.*, 2002). The presence of each family provides a single score out of 10 (1 being highly tolerant, 10 highly sensitive) with the cumulative scores providing the final IBMWP score. The Iberian Average Score per Taxa (IASPT) results from dividing the IBMWP score between the numbers of families, providing a single score out of 10 that represents the average tolerance of the community (Jáimez-Cuéllar *et al.*, 2002). The IMMi-L and IMMi-T are multimetric indices (Munné and Prat, 2009). The IMMi-L (Iberian

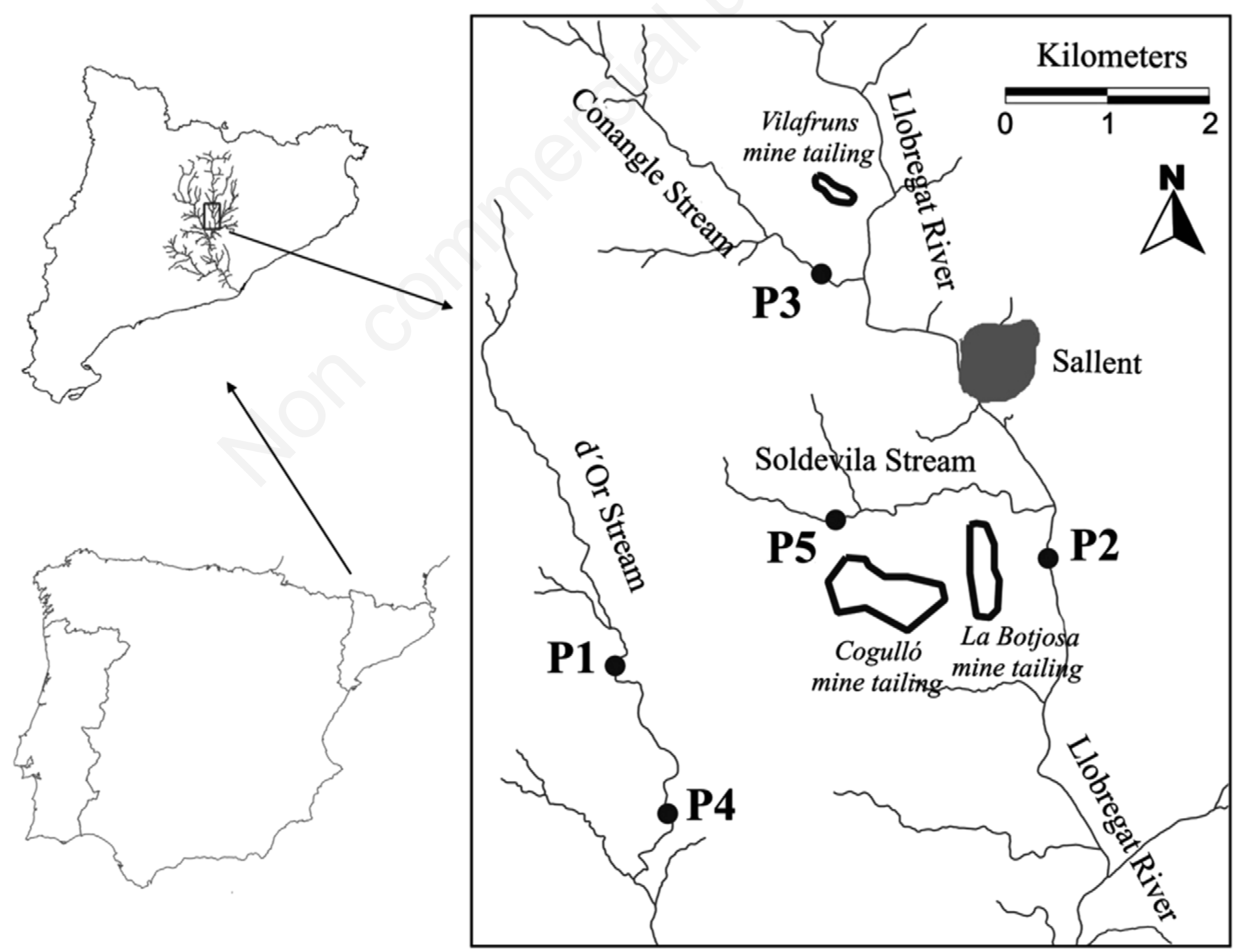

Fig. 1. Sampling sites and mine tailings in the study area.

Mediterranean Multimetric Index, using qualitative data) is based on a combination of the number of families, EPT, IASPT and % of Sel EPTCD, where EPT is the number of families belonging to the Ephemeroptera, Plecoptera and Trichoptera orders and EPTCD is the number of sensitive Ephemeroptera, Plecoptera, Trichoptera, Coleoptera and Diptera families. The IMMi-T (Iberian Mediterranean Multimetric Index, using quantitative data) is based on the number of families, EPT, IASPT and log (selected EPTCD + 1). Following the WFD, the indices were standardized by calculating the EQR value (Ecological Quality Ratio), *i.e.*, dividing the value of each index by the reference value for the correspondence river type. According to the official classification of these rivers by the Spanish Water Authorities (MMARM, 2008). P1, P3, P4 and P5 are catalogued as 'Mediterranean rivers of variable flow', whereas P2, located in the Llobregat River, is considered as a 'Main fluvial axis'. The performance of such indices in relation with the chemical status of streams can be found in Munné *et al.* (2012).

We also included conductivity and IBMWP data from one site (Ref) located in the Llobregat River upstream of salt discharges as a reference site (*i.e.* not affected by mining activity). The data were obtained from the Catalan Water Agency (https://aca-web.gencat.cat/aca/appmanager/aca/aca/) and only used to discuss our results.

Data analysis

Prior to the analyses environmental variables were normalized and macroinvertebrate data were log (x + 1) transformed. Additionally, the variables that were highly correlated (Spearman's correlation coefficient higher than 0.9) were removed from the analysis. In order to establish the main links between environmental variables and macroinvertebrate community, a distance-base linear model (DISTLM analysis; PERMANOVA + for PRIMER; Anderson *et al.*, 2008) was performed. DISTLM analyses the relationship between the macroinvertebrate resemblance matrix and the environmental variables studied as predictors. The macroinvertebrate distance matrices were calculated using the Chord distance method. Forward selection and R^2 selection criterion (Sokal and Rohlf, 1981) were used to obtain the environ-

Fig. 2. Images of the study area. A and B, photograph and Google Earth image of 'El Cogulló' mine tailing; C, rudimentary collection system of saline leachates in the Llobregat River at P2 site; D, Soldevila Stream at P5 sampling site.

mental variables that accounted for more variation in the DISTLM. A dbRDA plot analysis was used to visualize the final model using the CANOCO 4.5 software (Ter Braak & Smilauer, 2002).

RESULTS

Environmental variables

The study sites showed very different conductivities (Tab. 2), increasing from P1 to P5. The site P5 (Soldevila stream), which is located very near to ‘El Cogulló’ mine tailing, registered a very high conductivity (132.4 mS cm^{-1}). In the case of d’Or stream, conductivity was very different between the two studied sites: P1 showed the lowest conductivity value (1.4 mS cm^{-1}), whereas conductivity reached 11.0 mS cm^{-1} in P4. Conductivity was also relatively high in the Conangle Stream (4.5 mS cm^{-1}), near to ‘Vilafruns’ mine tailing, and in the Llobregat River at P2 (2.5 mS cm^{-1}), near to ‘La Botjosa’ mine tailing. Every studied site showed high concentrations of dissolved oxygen, with values ranging from 8.0 to 10.4 mg l^{-1} (Tab. 2). Temperature ranged between 12.0ºC in d’Or Stream at P1 site and 19.3ºC in the Soldevila Stream at P5 (Tab. 2).

According to the IHF hydromorphological index (Tab. 2) the P5 site showed the most deteriorated habitat (value of 27/100). The factors that reduced the IHF were the total absence of vegetation in the river channel, the great inclusion of larger substrate in the fine sediment and the low diversity of speed regimes and depth in the cannel. The rest of streams showed IHF values around 60, with a maximum at P2 (value of 68). Site P5 also showed the lowest value of the QBR index (Tab. 2), which was related to the extremely deteriorate gallery forest that exists in this stretch. Riparian forests in the other studied sites were in much better condition (Tab. 2), although QBR values were always lower than 80.

Macroinvertebrate community and relation to environmental variables

Fig. 3 shows macroinvertebrate community richness (Fig. 3A) and abundance (Fig. 3B) at the different studied sites. Both parameters resulted negatively correlated with electrical conductivity, being lowest at the sites with the highest conductivities (*i.e.*, P4 and P5). Only 2 macroinvertebrate families were recorded in P5: Ceratopogonidae and Ephydridae (2% and 98% of total macroinvertebrate abundance, respectively). The number of families detected at P4 increased to 7, with 4 of them belonging to the Diptera order. In the other studied sites the number of families was 15 or higher (Fig. 3A).

Diptera were dominant in every studied site in terms of relative abundance (Fig. 3B). The richness of EPT taxa decreased along the conductivity gradient. At P3, P4 and P5 only 2, 1 and 0 families of EPT were recorded, respectively; whereas P1 and P2 had 7 EPT families each. However, the abundance of EPT families was low at all sites, including P1 and P2 (4.76 and 7.56 individuals m^{-2}, respectively).

Fig. 4 shows the dbRDA graph resulting from DISTLM

Tab. 1. Biotic indices used in this study.

Biotic indices	Acronym	Reference/source
Unimetric indices		
Based on qualitative data		
Iberian Average Score per Taxon	IASPT	Jáimez-Cuéllar *et al.*, 2002
Iberian Biological Monitoring Working Party	IBMWP	Alba-Tercedor *et al.*, 2002
Multimetric indices		
Based on qualitative data		
Intercalibration Common Multimetric Index 11a	IMMi-L	Munné and Prat, 2009
Based on quantitative data		
Intercalibration Common Multimetric Index 10	IMMi-T	Munné and Prat, 2009

Tab. 2. Results of physico-chemical and morphological variables in the study.

Site	River	Conductivity (mS/cm)	Temperature (°C)	Oxygen (mg/L)	IHF Index	QBR Index
P1	d´Or Stream	1.4	12.0	8.0	59	75
P2	Llobregat River	2.5	12.8	10.2	68	40
P3	Conangle Stream	4.5	14.5	9.3	60	60
P4	d´Or Stream	11.0	15.9	10.4	57	55
P5	Soldevila Stream	132.4	19.3	9.7	27	5

analysis, relating the macroinvertebrate community with the environmental variables. This analysis explained 89.2% of the total variance in the macroinvertebrate data, with 47.7% explained by the first axis and 30.3% by the second one. The only environmental variable significantly selected in the final model was conductivity ($P<0.05$), which explained the 43.8% of the total variance. This variable showed a Spearman's correlation coefficient value of 0.9 with the first axis of the dbRDA. The other two variables included in the final model (although not significantly) were the total QBR index and the first component of this index (which refers to the total riparian vegetation cover).

According to the dbRDA, most families resulted negatively correlated with conductivity. Among these families, Nemouridae, Baetidae, Philopotamidae, Corixidae, Anthomyiidae, Psychodidae and Cambaridae were only recorded at P1 site, where the lowest conductivity was registered. Tubificidae, Hydropsychidae, Polycentropodidae and Tabanidae were exclusively registered at P1 and P2. Only Ephydridae and Ceratopogonidae showed a positive and strong (>0.6) correlation with the first axis (*i.e.*, conductivity).

Biotic indices

According to the limits proposed by Munné and Prat (2009), the different biotic indexes showed a significant ecological deterioration due to salt pollution (Fig. 5). P4 and P5 showed values of EQR IBMWP of 0.14 and 0.03 respectively (poor and bad categories). The rest of streams presented values around 0.4, although none of them reached the good ecological status required by the WFD. The IASPT metric also decreased along the conductivity gradient, from 0.74 at P1 to 0.36 at P5.

The multimetric indices IMMi-T performed similarly to IBMWP, since neither of sites reached the good status and their values progressively decreased as water conductivity increased. P4 and P5 sites were also classified as poor and bad ecological status respectively. The IMMi-L clearly differed from the IBMWP and IMMi-T indices, showing higher values at all sites. For example, according to this index, P1 and P2 had a high and good ecological status respectively.

Discussion

Conductivity was higher in all sites than in the reference stream located upstream the mining activity ('Ref' site in Fig. 6; 0.8 mS cm^{-1}). Our results confirm the existence of mine leachates coming from the salt tailings within the study area (Otero and Soler, 2002; Rovira *et al.*, 2008, 2006), although we registered a larger gradient of conductivity (from 1.4 to 132.4 mS cm^{-1}) than the one reported in those studies. The impact of salt pollution was especially high at P5 (Soldevila Stream), which registered conductivities almost 3 times higher than seawater due to salt leachates coming from the Cogulló mine tailing. This stream receives the overflow of the perimetral spillway that collects the rainwater falling on the mine tailing; when heavy rains occur not all the water is collected by such infrastructure and it is diverted to Soldevila stream. The same mine tailing is very likely responsible for the salt pollution of site P4 (11.0 mS cm^{-1}), given the highly saline water springs (*i.e.*, conductivities higher than 50 mS cm^{-1}) coming from that tailing located at a site between P1 and P4. The high conductivity values registered at P3 (Conangle Stream, 4.5 mS cm^{-1}) were related to the Vilafruns mine

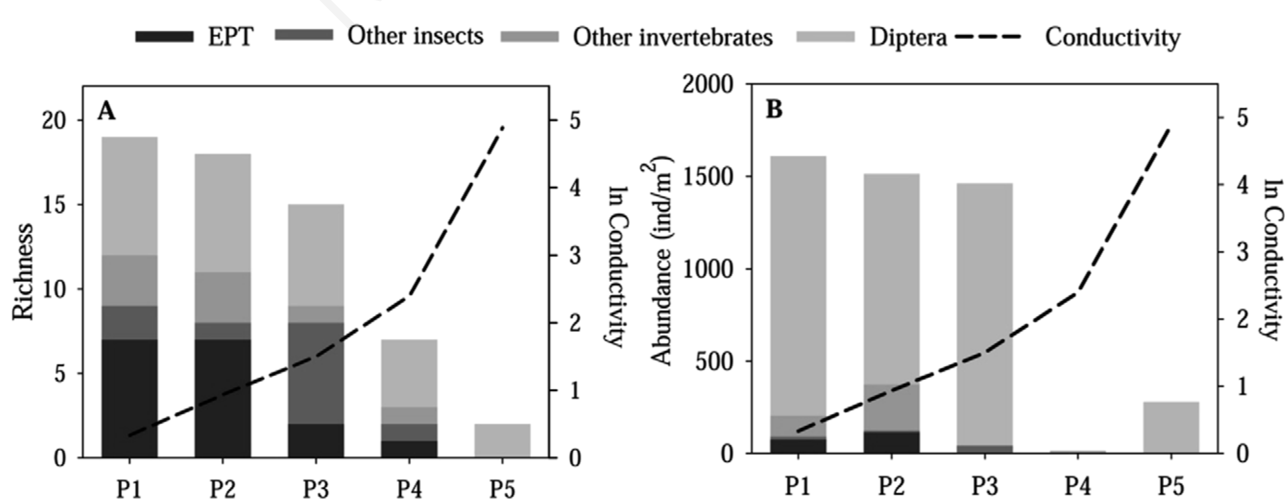

Fig. 3. Macroinvertebrate richness and abundance at each studied site. EPT means Ephemeroptera, Plecoptera and Trichoptera families.

tailing, since we registered conductivities higher than 50 mS cm^{-1} in water springs coming from it. Finally, site P2 (located at the Llobregat River) registered lower conductivities than the rest of sites (2.5 mS cm^{-1}), but still beyond reference conditions (*i.e.*, upstream the mine tailings). The lower conductivities registered at P2 are related to the construction of a salt collector conducting diluted salt wastes directly to the sea (Martín-Alonso, 1994). However, salt-polluted waters from the mine tailing still infiltrate to the subsoil and discharge to the Llobregat River.

The salinization of the Llobregat River downstream the salt mines has already been evidenced by previous studies (Barata *et al.*, 2005; Damásio *et al.*, 2008; Prat and Rieradevall, 2006). However, these studies never reported values beyond 2 mS cm^{-1}, probably because of the capacity of the river to dilute salts. Our results show that the streams that surround the mines received a heavy salt pollution, which had already been reported by local stakeholders (http://www.lasequia.cat/montsalat/) but ignored by water authorities. The conductivities registered in these streams are even higher than those reported in other potash mining areas. For example, in the Werra River (Germany) conductivities in salt polluted areas ranged from 0.46 to 7.24 mS cm^{-1} (Braukmann and Böhme, 2011). Additionally, according to Spanish law (ORDEN ARM/2656/2008, 2008), the conductivity values registered in all the studied streams (except for P1) exceed the limit established for good water quality (1.5 mS cm^{-1}) and even the limits established for human uses of water (2.5 mS cm^{-1} RD140/2003, 2003) and what is considered as suitable for irrigation (1.5 mS cm^{-1}; Mirza, 1998; Muschal, 2006).

Salt pollution led to a clear degradation of the habitat, as reflected by low values of the QBR and IHF indices. Habitat degradation was especially severe in the P5 site, where extreme water conductivity and high floods of saline water coming from the perimetral collectors of the Cogulló mine tailing, led to the disappearance of any kind of vegetation in the stream margin (Fig. 2D). The reduction in the canopy cover can have important consequences for stream metabolism, such as increased light penetration and a modification of the organic matter inputs into the stream (Dunlop *et al.*, 2005; Millán *et al.*, 2011). Riparian vegetation also plays a key role in regulating the amount of sediments that enter the stream (Schoonover *et al.*, 2006). Accordingly, we registered a high amount of fine sediment covering the streambed in P5, reflecting a clear alteration of the fluvial habitat. The rest of sites did not show such clear habitat degradation; however, conductivity was high

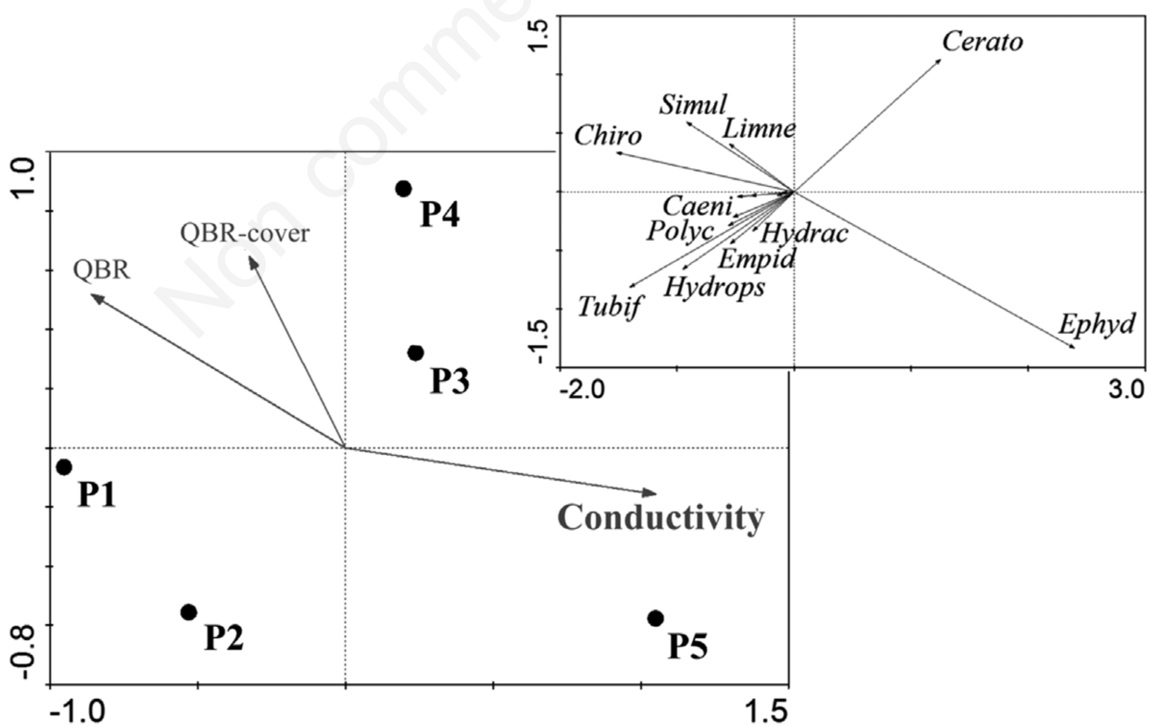

Fig. 4. dbRDA plot result from DISTLM analysis. Only the variables included in the final model are plotted, and significant variables ($P<0.05$) are shown in bold. Macroinvertebrate taxa with Spearman's correlation coefficient higher than 0.6 with the first axis are represented at the high right corner of the figure. QBR-cover, the component of the QBR index that refers to the proportion of riparian area covered by trees and shrubs.

enough to reduce aquatic vegetation. These findings align with previous studies, which reported that most freshwater macrophyte species can be damaged by conductivities ranging from 1.5 to 3 mS cm^{-1} (Dunlop *et al.*, 2005; Hart *et al.*, 1991; James *et al.*, 2003; Kipriyanova *et al.*, 2007).

Salt pollution also had clear effects on the aquatic macroinvertebrate communities, as had been previously reported for streams affected by potash mining in the Werra River basin (Bäthe and Coring, 2011; Braukmann and Böhme, 2011) and for the Llobregat River (Prat *et al*., 1983; Prat and Rieradevall, 2006). Taxa richness strongly responded to salinization, with only 7 taxa being registered in site P4 (conductivity=11.0 mS cm^{-1}) and 2 (Ephydridae and Ceratopogonidae) in P5 (conductivity=132.4 mS cm^{-1}). These last two families were the only to register a positive relationship with conductivity according to the DISTLM analysis. Ceratopogonidae and Ephydridae have already been reported as salt tolerant (Berezina, 2003; Kefford *et al.*, 2006), probably due to their high hyperosmotic regulation ability (Barnby, 1987; Herbst *et al.*, 1988; Nemenz, 1960; Sutcliffe, 1960). The physiological processes that allow these taxa to withstand high salinities have a high energetic cost, but they allow them to dominate in extreme habitats, where competition and predation are low (Southwood, 1988).

Even at the lowest range of the studied conductivity gradient, our sites showed an impoverished biological quality, when compared with reference conditions (Fig. 6). These findings align with those of previous studies, which reported that biological quality of salt polluted streams can be affected by conductivities as low as 1.5 mS cm^{-1} (Dunlop *et al.*, 2005; Hart *et al.*, 1991; Horrigan *et al.*, 2005; Kefford *et al.*, 2011), or even lower (Cormier *et al.*, 2013; Kefford *et al.*, 2011). Anyhow, the damage to the aquatic invertebrate community can be expected to increase with increasing conductivity (Berezina, 2003; Böhme, 2011; Braukmann and Böhme, 2011; Cañedo-Argüelles *et al.*, 2012; Hart *et al.*, 1991; Horrigan *et al.*, 2005; Pinder *et al.*, 2005; Piscart, 2005; Williams *et al.*, 1999). It is important to notice the reduction in EPT taxa at sites P3 and P4, and

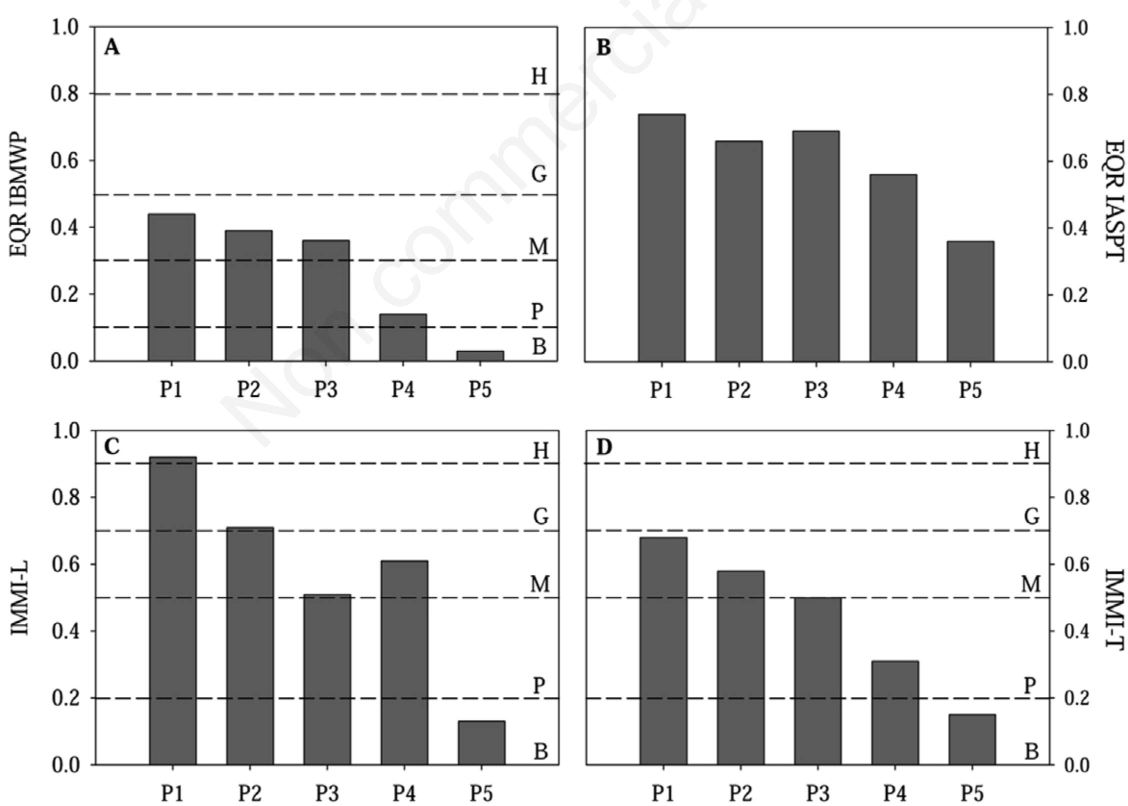

Fig. 5. Scores of each of the analysed biotic indices (IBMWP, IASPT, IMMI-L and IMMI-T) at each study site. IBMWP = Iberian Biological Monitoring Working Party (Alba-Tercedor *et al.*, 2002); IASPT, Iberian Average Score per Taxa (Jáimez-Cuéllar *et al.*, 2002). IMMi-L, Iberian Mediterranean Multimetric Index, using qualitative data (Munné and Prat, 2009); IMMi-T, Iberian Mediterranean Multimetric Index, using quantitative data (Munné and Prat, 2009). Discontinued lines show the quality class boundaries (EQR values) according to Munné and Prat (2009): H, high; G, good; M, moderate; P, Poor; B, bad.

their absence at P5. Previous studies have already proved the salt sensitivity of these taxa (García-Criado *et al.*, 1999; Kefford *et al.*, 2011; Pond, 2012; Pond *et al.*, 2008), which can rarely be found at conductivities above 3 mS cm^{-1} (Cañedo-Argüelles *et al.*, 2013; Cormier *et al.*, 2013). We found a significant reduction in EPT taxa at conductivities above 4.5 mS cm^{-1} (site P3).

The overall decrease in taxa richness and in EPT taxa led to an important reduction in the IBMWP index along the conductivity gradient; although none of the sites met the requirements of the WFD (*i.e.*, good ecological status). Thus, the IBMWP seems to be an efficient tool to detect salt pollution of streams (at least in the studied region). García-Criado *et al.* (1999) already observed a decrease in the IBMWP values along a conductivity gradient in streams affected by coal mining in Northern Spain, although they could not verify the efficiency of the index for distinguish this kind of pollution from other sources. However, it should be noticed that the conductivity of the streams in that study never exceeded 0.5 mS cm^{-1}, which is even lower than the conductivity registered at our reference (*i.e.*, not impacted by mining) stream (0.83 mS cm^{-1}). The extinction of those taxa associated with high ecological status is reflected by the values of the IASPT index, with all the studied sites showing values well-below reference conditions.

The multimetric and quantitative index IMMi-T showed also the strong impairment of the biological communities. On the contrary, the qualitative index IMMi-L did not show a degradation of the studied streams since P1 and P2 fell into the 'high' and 'good' category respectively, meeting requirements established by the WFD. Also P4 showed a high value of this index (0.61), close to the good category, despite the significant community disruption previously commented for this site. The high values of these sites are mainly due to the presence (at very low densities) of some taxa sensitive to organic pollution as Limnephilidae (at P1, P2 and P4), Polycentropodidae (at P1 and P2) and Philopotamidae (at P1), which are taken into account in the selected EPTCD metric included in this index. Thus, IBMWP and IMMi-T indices seem to be well suited for detecting salt pollution in our region, whereas we should advice against using IMMi-L

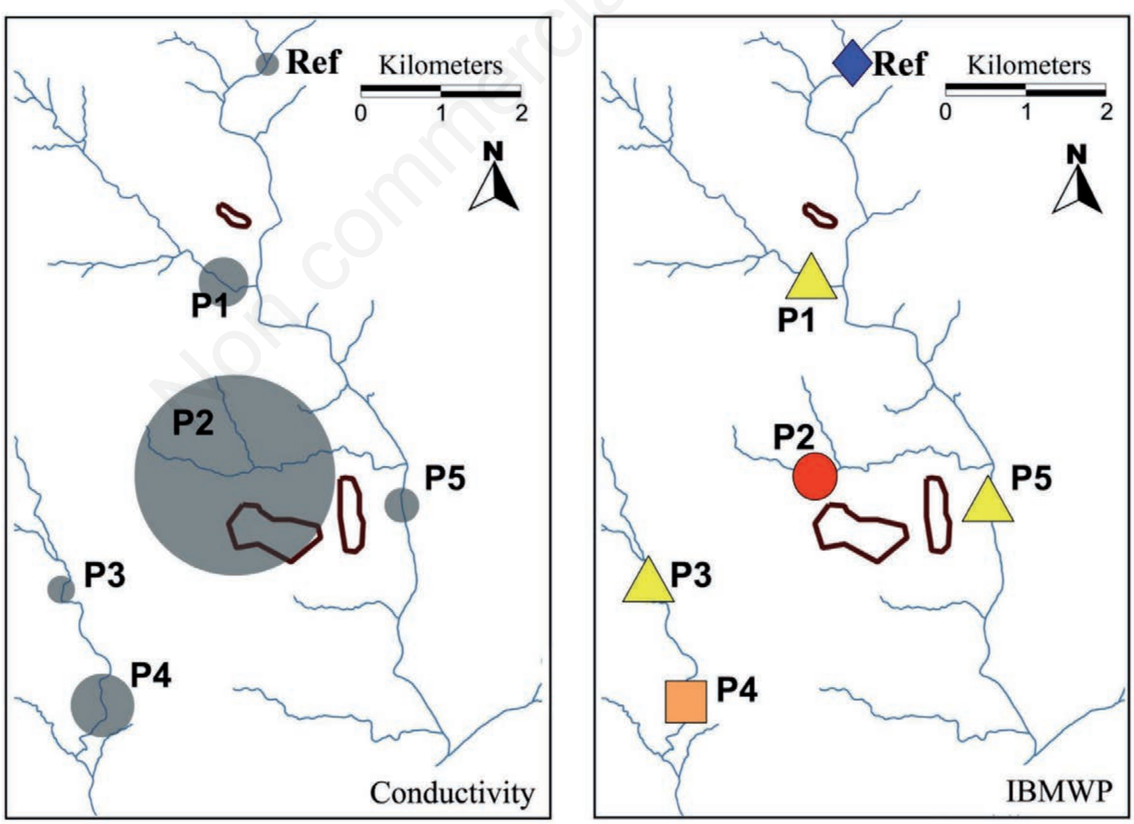

Fig. 6. Maps of the study area with proportional symbols to conductivity values at each site (left) and with ecological status categories according to IBMWP index and to Munné and Prat (2009) (right). Shapes indicate ecological status according to the WFD: diamond shape is "high", triangle is "moderate", square is "poor" and circle is "bad". Ref site has been used as reference and data has been obtained from the Catalan Water Agency.

when evaluating the damage caused by freshwater salinization. Our results highlight the importance of testing different biotic indices when assessing the effect of pollution sources different from organic enrichment.

Conclusions

Overall, we can conclude that potash-mining activities have the potential to cause severe ecological damage to their surrounding streams. This is mainly related to an inadequate management of the mine tailings, leading to highly saline runoff and percolates entering surface waters. Thus, we urge water managers and policy makers to take action to prevent, detect and remediate salt pollution of rivers and streams in mining areas. In the case of the potash mines studied here, the Superior Court of Justice of Catalonia already ruled against the mining industry (Iberpotash), demanding restoration actions for the affected watercourses. Our results suggest that restoration measures should be implemented as soon as possible, without waiting for the mine closure (expected for 2035). Anyhow, given the economic importance of these activities (*e.g.*, creating jobs), management actions are much more likely to be successful if mining companies, scientists and stakeholders get together to find cost-effective solutions for preventing salt pollution (Cañedo-Argüelles *et al.*, 2016).

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