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Effects of potash mining on river ecosystems: An experimental study☆

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In spite of being a widespread activity causing the salinization of rivers worldwide, the impact of potash mining on river ecosystems is poorly understood. Here we used a mesocosm approach to test the effects of a salt effluent coming from a potash mine on algal and aquatic invertebrate communities at different concentrations and release modes (i.e. press versus pulse releases). Algal biomass was higher in salt treatments than in control (i.e. river water), with an increase in salt-tolerant diatom species. Salt addition had an effect on invertebrate community composition that was mainly related with changes in the abundance of certain taxa. Short (i.e. 48 h long) salt pulses had no significant effect on the algal and invertebrate communities. The biotic indices showed a weak response to treatment, with only the treatment with the highest salt concentration causing a consistent (i.e. according to all indices) reduction in the ecological quality of the streams and only by the end of the study. Overall, the treatment's effects were time-dependent, being more clear by the end of the study. Our results suggest that potash mining has the potential to significantly alter biological communities of surrounding rivers and streams, and that specific biotic indices to detect salt pollution should be developed.

1. Introduction

Resource extraction is increasing worldwide to meet human demands for energy and goods (Krausmann et al., 2009; Reichl et al., 2016). In 2014 the world production of mineral raw materials was of 17,434 million metric tons, providing a total revenue of thousands of billions of US dollars (Reichl et al., 2016). One valuable mineral is potash, with a world production of 39.55 million metric tons in 2014, increasing at a rate of 19.52% from 2010 (Reichl et al., 2016). Many mining operations generate wastes (e.g. potash mines can generate tailings dominated by NaCl) that are one of the world's largest chronic waste concerns (Bian et al., 2012). It is estimated that hundreds of thousands of tons of mine tailings are produced per day (Jakubick et al., 2003). For example, in the US mining activities generate 10 times as much solid waste as municipal solid waste per capita (Hudson-Edwards et al., 2011). Very often these wastes are stored in impoundments around the mines, from where they can reach surface waters by seepage through embankments or through the base of the tailings pile (Hudson-Edwards et al., 2011).

The ecological impact of coal and metal mines on surface waters has received considerable attention from the scientific community (Dudka and Adriano, 1997; Palmer et al., 2010). Several studies have showed that biological communities can be seriously impaired by mine wastes (e.g. Clements et al., 2000; Pond et al., 2014). However, the potential effects of potash mining on river ecosystems are less understood (Bäthe and Coring, 2011; Braukmann and Böhme, 2011; Cañedo Argüelles et al., 2012; Coring and Bäthe, 2011; Schulz, 2016; Ziemann et al., 2001). The potassium in potash is one of the most important components of commercial soluble fertilizers, since it is an essential plant nutrient. Annual potash production capacity was projected to increase globally from 52 million tons in 2015 to 61 million tons in 2019, and world consumption for all uses of potash

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was projected to increase gradually from 35.5 to 39.5 million tons for the same period (Ober, 2016). As other mining activities, potash extraction generates large quantities of waste. For example, in the potash mines of central Catalonia, 3 tons of waste are generated for each ton of potash that is extracted (Gorostiza Langa, 2014). These wastes are mainly composed of NaCl and they are often stored in open locations near the mines, resulting in artificial mountains (i.e. mine tailings). Although management measures such as brine collectors have been implemented (Martín-Alonso, 1994), the salts from the tailings are still dissolved by rain and humidity (Cañedo Argüelles et al., 2012; Otero and Soler, 2002) and they often leak from the collecting and retention infrastructures (Gorostiza Langa, 2014). Thus, large quantities of these salts end up in streams and rivers around the potash mining areas.

Since river organisms are adapted to freshwater, the increase in the salt concentration caused by potash mining wastes has the potential to significantly alter the river ecosystem (Cañedo-Argüelles et al., 2013). Studies on the River Werra (Germany), which was heavily impacted by potash mining, showed a recovery in biological communities after salt pollution was lowered from maximum chloride concentrations of 27 g L^{-1} in 1992 to 2.5 g L^{-1} in 2000 due to the implementation of management practices (Bäthe and Coring, 2011; Coring and Bäthe, 2011). These studies suggested that biological quality could be further improved if maximum chloride concentration was lowered to 1.5 g L^{-1}. However, the information on the ecological impacts of freshwater salinization is still too scarce to robustly guide management decisions and there are important questions that remain unanswered. Moreover, in field studies it is difficult to isolate the effect of salt pollution on aquatic organisms from the effect of other variables (e.g. habitat characteristics, nutrient concentrations). In this regard, mesocosm studies allow conducting experiments under controlled conditions and, at the same time, capturing some of the complexity of natural ecosystems (Odum, 1984). Previous mesocosm studies on the potential effects of potash mining pollution on river ecosystems suggested that high salt concentrations (i.e. higher than 3 g L^{-1}) could lead to significant changes in the aquatic macroinvertebrate communities (Cañedo-Argüelles et al., 2015, 2012), and that short salt pulses had little effect on diatom and invertebrate communities (Cañedo Argüelles et al., 2014). In all these mesocosm studies the changes in community composition were related with changes in abundance of the different taxa, but not with changes in species richness. However, these studies had two important limitations: they were conducted over short periods of time (i.e. maximum duration = 16 days) and they used NaCl as a proxy to potash mine wastes.

Here we used a mesocosm approach to study the potential impact of potash mining on river ecosystems and to provide management recommendations. Although different strategies exist to avoid disposing salts into rivers and streams (Martín-Alonso, 1994), in some cases salt disposal might be unavoidable. Thus, one of the most pressing management concerns is to know what is the best disposal strategy: to dispose salts continuously at low concentrations (i.e. press release) or to dispose salts at higher concentrations during short periods of time (i.e. pulse releases). Also, there is a need to know what is the maximum salt concentration that should be allowed in rivers and streams to prevent damaging the ecosystem and degrading its ecological status. Here, we tested the effects of a salt effluent coming from a potash mining tailing heap in the River Werra basin (Germany) on algal and aquatic invertebrate communities at different concentrations (similar to those currently registered in the River Werra, impacted by potash mining) and different salt disposal schemes (i.e. pulse versus press releases). We focused on algae and invertebrates, which are good indicators of water quality (Potapova and Charles, 2007; Rosenberg and Resh, 1993). Our initial hypothesis was that high salt concentrations (i.e. above 3 g L^{-1}) would have significant effects on algal and invertebrate communities, whereas moderate and low salt concentrations (i.e. below 3 g L^{-1}) would not. We also expected taxa richness to be unaffected by the salt treatment, leading to a weak response of biotic indices (which heavily rely on richness as an indicator). Finally, we expected short salt pulses (i.e. 48 h long) to have no significant effect on the algal and invertebrate communities due to the capacity of organisms to tolerate these short phases of stress and to recover between pulses (Cañedo Argüelles et al., 2014).

2. Methods

2.1. Experimental setup

The experiment was performed in a set of 12 artificial streams (Supplementary material) of 3 m of length, fed by river water coming from the upper part of the Ter River (Catalonia, Spain). The river water included small concentrations of N and P and dissolved organic matter mainly due to human activities (e.g. use of fertilizers). The Ter headwaters are located in the Pyrenees. Its basin has alkaline-earth bicarbonate waters. Bicarbonate represents 63% of the total anions and calcium 60% of the total cations. Other ions reach relevant concentrations. Sulphate constitutes on average 24% and chloride only 13% of the total anions (Sabater et al., 1992). Water conductivity along the catchment ranges from 250 μS/cm to 950 μS/cm (CERM database). The mesocosm was located in an open field in the Museu del Ter (Manlleu, Catalonia, Spain), close to a river channel. The river water was pumped into four 1500 L mixing tanks, each of them feeding three artificial streams (i.e. pvc pipes) and flowing into 500 L tanks from where it was re-circulated. One of the tanks was left as control (i.e. containing only river water), whereas in the other three a salt treatment was applied. The salt treatments consisted in dissolving a salt-saturated stock solution (coming from a tailing heap in the River Werra basin) into river water (total salt concentration = 0.248 g/L) at different concentrations (i.e. Mod, High and Mod-p treatments). The stock solution contained 172 g/l Chloride, 15.5 g/l Potassium and 24.9 g/l Magnesium. In the moderate treatment (Mod) the salinity was 2.27 ± 0.36 g/L, and in the high treatment (High) it was 3.78 ± 0.26 g/L. These salt concentrations are typically observed in different sections of the River Werra during different periods of the year (Bäthe and Coring, 2011; Coring and Bäthe, 2011). Finally, in the moderate + pulses treatment (Mod-p) the salinity concentration was 1.61 ± 0.08 g/L and it was increased to 2.23 ± 0.03 g/L during 3 pulses of 48 h of duration applied 19, 28 and 35 days after the beginning of the experiment. The pulses were created by slowly adding the salt-saturated solution to the mixing tank while continuously controlling salinity with a conductimeter. After 48 h river water was added to the mixing tanks to dissolve salts and re-establish pre-pulse conditions. Since we knew the volume of water in the tanks and the salt concentration in the river water and the mixing tanks, we could calculate which volume of river water was needed to re-establish initial (i.e. pre-pulse) salt concentrations.

We collected invertebrates and cobbles from the Ter River at Les Masies de Voltregà, which is a river section located near the artificial streams that has a good water quality and mean conductivity of 350 μS/cm (CERM database). A total of 144 river cobbles were collected and transferred to the artificial streams (placing 12 per stream). Additionally, 12 macroinvertebrate samples were collected by kick-net sampling the riverbed for 1 min using a 250 μm mesh size. Each sample was emptied at the top of each stream. Thus, initial algal communities consisted in those that were attached to the river cobbles, whereas invertebrate communities were