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Review

Salinisation of rivers: An urgent ecological issue

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ABSTRACT

Secondary salinisation of rivers and streams is a global and growing threat that might be amplified by climate change. It can have many different causes, like irrigation, mining activity or the use of salts as de-icing agents for roads. Freshwater organisms only tolerate certain ranges of water salinity. Therefore secondary salinisation has an impact at the individual, population, community and ecosystem levels, which ultimately leads to a reduction in aquatic biodiversity and compromises the goods and services that rivers and streams provide. Management of secondary salinization should be directed towards integrated catchment strategies (e.g. benefiting from the dilution capacity of the rivers) and identifying threshold salt concentrations to preserve the ecosystem integrity. Future research on the interaction of salinity with other stressors and the impact of salinization on trophic interactions and ecosystem properties is needed and the implications of this issue for human society need to be seriously considered.

1. Introduction

Salinity refers to the total concentration of dissolved inorganic ions in water or soil (Williams and Sherwood, 1994) and is therefore a component of all natural waters. Dissolved ions can be also expressed as the ionic activity of a solution in terms of its capacity to transmit electrical current (electrical conductivity (EC), measured in Siemens per meter). Therefore, EC is routinely used to measure salinity, and the relation between them is a function of water temperature. Surface waters can be classified according to their salt content as follows (Venice system, 1959): freshwater < 0.5 g L^{-1}; oligohaline 0.5–4.0 g L^{-1}; mesohaline water 5–18 g L^{-1}; polyhaline water 18–30 g L^{-1}; euhaline water 30–40 g L^{-1}; hyperhaline water > 40 g L^{-1}. In inland waters, salinity may vary from 10's of mg L^{-1} to 100's of g L^{-1} and is a major factor limiting the distribution of biota (Williams, 1987). In fact, aquatic biota have been commonly grouped according to their salinity preferences; i.e. freshwater fauna, brackish-water fauna and marine fauna (Remane and Schlieper, 1971). While in principle salinity can refer to any inorganic ions, in practise it is mostly the result of the following major ions: Na^+, Ca^{2+}, Mg^{2+}, K^+, Cl^-, $SO_4{}^{2-}$, $CO_3{}^{2-}$ and $HCO_3{}^-$ (Williams, 1987).

In the absence of anthropogenic influences, salinity and the proportions of the above ions originate from three sources. (1) Weathering of the catchment, which is a function of both geology of the catchment and precipitation. (2) Sea spray, although this is only an important source of salts in coastal locations. (3) Small amounts of salts dissolved in rainwater as a consequence of evaporation of seawater. This third source can be a significant source of salt in the terrestrial landscapes distant from the sea (Herczeg et al., 2001). Regardless of its source, dissolved ions can be concentrated by evaporation and transpiration, and this is particularly important in semi-arid, arid and regions with seasonally hot dry climates. The ability of plants to extract small amounts of soil moisture and the depth that their roots can capture soil moisture play an important role determining the resultant salinity of soil, groundwater and runoff to rivers (and thus dilution of salts). Furthermore salts can be stored in soils, sub-soils and groundwater as a result of previous periods of aridity and then subsequently be released. Especially in regions with flat topography, stored salts can move very slowly and

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remain in the landscape for an extended period. Such storage and release of salts can occur at various time-scales from seasons, decadal scale climate variations (McNeil and Cox, 2007) to 100,000's years (Herczeg et al., 2001). So natural salinity of rivers is a complex and dynamic function of the climate (recent and past), the geology of its catchment, the distance from the sea, topography and vegetation.

Anthropogenic increases in salinity are referred to as secondary salinisation. This contrasts with primary salinisation, which involves the accumulation of salts originating from natural sources at a rate unaffected by human activity as outlined above. Secondary salinisation is a global and growing threat that poses a risk of causing severe biodiversity losses and compromising the ecosystem goods and services that rivers, wetlands and lakes of the world provide. Therefore, salinisation has been rated as one of the most important stressors for freshwater ecosystems in the Millennium Ecosystem Assessment (2005). Moreover, in the United States salinisation is among the top 15 causes of impairment of streams being equally important as pesticide input (US Environmental Protection Agency, 2012). In Australia a survey of river managers rated salinity in the top three most important environmental contaminants (Lovett et al., 2007). Although there are some review papers dealing with Australian rivers (Hart et al., 1991; James et al., 2003; Dunlop et al., 2005), no comprehensive review integrating different regions of the world has been done yet. This paper reviews the major causes of secondary salinisation, its impacts on the biota of rivers, management of rivers subject to secondary salinisation and identifies future research needs in these areas under a global perspective.

2. What causes river secondary salinisation?

River salinisation can have many different causes (Table 1). Irrigation and rising of groundwater tables has been reported as one of the main causes of secondary salinisation, especially in the arid and semi-arid regions of the world where crop production consumes large quantities of water. Since crops absorb only a fraction of the salt of the irrigation water, salt concentrates and soil water becomes more saline (Lerotholi et al., 2004). These salts may be leached out through run-off and end up in the river. In addition irrigation, which is mainly developed in flat geo-morphological bottom areas (the natural salt sinks of the arid landscape), causes the mobilisation of large fossil salt storages dating from the past marine or otherwise saline geological history of the soil (Smedema and Shiati, 2002). Therefore irrigation has been reported to be responsible for the salinisation of many streams, e.g. Amu Darya and Syr Darya Rivers in Central Asia (Létolle and Chesterikoff, 1999; Crosa et al., 2006), Breede River in South Africa (Scherman et al., 2003), the Ebro River in Spain (Isidoro et al., 2006) and the Great Menderes in Turkey (Koç, 2008). In Australia, due to the climatic and geomorphological characteristics of the landscape, a great portion of groundwater is saline (Blinn et al., 2004). This results in river salinity via two major mechanisms a) rising ground water, due to the clearing of natural deep-rooted native vegetation from catchments, passively flowing into rivers (Peck, 1978; Williams, 2001), and b) active discharge of excess saline water into rivers (Kefford, 1998).

Mining activity is another major source of salts entering the rivers. Large quantities of potash salts are extracted each year for the manufacture of agricultural fertilisers. During the manufacturing process of crude salt (containing not only potash but also NaCl and other salts) huge amounts of solid residues are stockpiled. The salts are dissolved during precipitation events and may enter the surface waters. For example, in the Werra basin of Germany, around 2 million tonnes of salts currently enter the river each year (Richter et al., 2010). In that basin peak chloride concentrations of more than 30 g L^{-1} ($\approx$50 g NaCl g L^{-1}) were registered during the period of maximum mining activity (Coring and Bäthe, 2011) and for a long time the salinities of the lower Werra were higher than those of the North Sea (Bäthe, 1997). The same problems have been reported in the Llobregat River near Barcelona, Spain, where the residue stockpiles have considerably altered the landscape and caused important water management problems (Martín-Alonso, 1994; Prat and Rieradevall, 2006; Cañedo-Argüelles et al., 2012). Mountaintop mining, a technique which involves removing 500 or more feet of a mountain to gain access to coal seams, is also responsible for large scale stream salinisation (Pond et al., 2008). The exposure of coal seams to weathering and percolation during coal mining provides many opportunities for the leaching of sulphate from coal wastes into surface waters (Fritz et al., 2010; Bernhardt and Palmer, 2011). By contrast to salt mining, where the main ions are Na^+ and Cl^-, SO_4^{2-} is the dominant ion in the coal mining effluent. In the Central Appalachians, identified as the most biologically diverse freshwater systems in North America, more than 10% of their total area is disturbed by surface mining (Palmer et al., 2010). As a result of mining activities, impacted streams can have 30- to 40-fold increases in SO_4^{2-} concentrations (Hartman et al., 2005; Pond et al., 2008) and 10-fold increases in conductivity (Johnson et al., 2010).

In the cold regions of the world stream salinisation has been often reported as the result of the use of salts as de-icing agents for roads (Williams et al., 2000; Löfgren, 2001; Ruth, 2003). During 1961–66 the amount of salts used to de-ice North American roads increased from 909,000 to 1,347,000 tonnes per winter (Hanes et al., 1970). During the 1980s the amount of salts applied to roads increased to 10 millions of tonnes per year only in the United States (Salt Institute, 1992). Nowadays around 14 million of tonnes are applied annually in North America (Environment Canada, 2001). Most of the salts used on roads are transported to adjacent streams during rainfall events and snow melting periods (Williams et al., 2000). Consequently, chloride concentrations have been measured at over 18 g L^{-1} ($\approx$30 g NaCl g L^{-1}) in road runoff water (Environment Canada, 2001). Concentrations downstream from major roads have been recorded to be up to 31 times higher than comparative upstream concentrations (Demers and Sage, 1990) and

Table 1
Some examples for rivers subjected to secondary salinisation.

Site	Region	Source of salinisation	Selected references
Amu Darya and Syr Darya rivers	Central Asia	Irrigation for agriculture	Létolle and Chesterikoff, 1999; Crosa et al., 2006; Smedema and Shiati, 2002
Appalachians	USA	Coal mining	Pond et al., 2008; Fritz et al., 2010; Palmer et al., 2010
Murray-Darling basin	Australia	Clearing of natural vegetation	Williams, 2001; Kefford et al., 2006a; Muschal, 2006
Meurthe River	France	Soda production factories	Piscart et al., 2005a,b
Werra, Weser and Wipper rivers	Germany	Salt mining	Ziemann et al., 2001; Bäthe and Coring, 2011
Umbilo, Kat and Vaal rivers	South Africa	Sewage and industrial effluents	Williams et al., 2003; Lerotholi et al., 2004; Dikio, 2010
White mountains	USA	Deicer for roadways	Kaushal et al., 2005

some rural streams have registered chloride concentrations exceeding 0.1 g L^{-1} ($\approx$0.16 g NaCl g L^{-1}) which are similar to those found in the salt front of the Hudson River estuary (Kaushal et al., 2005).

These are the most frequently reported causes for river salinisation, but the salts entering the world rivers can have many different origins including discharge from industrial activities (Piscart et al., 2005b; Dikio, 2010), gulp injection or salt dilution technique (Wood and Dykes, 2002), sewage treatment plant effluents (Silva et al., 2000; Williams et al., 2003; Lerotholi et al., 2004) or reduced river discharge due to damming (Mirza, 1998). Thus, it can be stated that river secondary salinisation is a worldwide phenomenon that can have many different causes. The aim of this review is to synthesize and interpret most of the available information concerning river salinization, providing a global picture of its implications for river ecosystems.

3. What is the impact of secondary salinisation on river ecosystems?

3.1. Impact at different levels of ecosystem organisation

3.1.1. Organism/individual

Freshwater organisms need to maintain an internal osmotic pressure relative to the media in which they live. This means that their internal 'salinity' is greater than the external salinity and they must expend energy to maintain ions in their bodies and exclude water. If the salinity of the external water becomes higher than the internal salinity, they will have to either cope with a higher internal salinity (osmocomform) or spend energy to expel ions and keep water. True osmocomforming species (those whose body fluids change directly with a change in the concentrations of dissolved ions in the water) do not exist in freshwater, as their tissue would be too thin for normal metabolic processes, but some freshwater species osmoregulate (i.e. they actively regulate the osmotic pressure) at low salinity and then osmocomform at higher salinities. Some species show different osmoregulatory traits depending on their life-cycle stage, e.g. *Macrobrachium rosenbergii* inhabits brackish-water as a larval stage and develops mechanisms for withstanding the osmoregulatory stresses of fresh water as adult (Funge-Smith et al., 1995). Most freshwater species are osmoregulators, and this has a metabolic cost that may affect the organism's long term viability or resilience (Hart et al., 1991), and thus might play a role in natural selection (Piscart et al., 2006). The concentration of hemolymph solutes in freshwater animals is generally lower than 16 g L^{-1} of NaCl (Withers, 1992) and they rarely survive salinities above 25 g L^{-1} (Pinder et al., 2005). When the salt concentration of the medium becomes too high the osmoregulatory mechanisms will collapse resulting in cellular damage and possibly death.

One might expect that freshwater organisms would thus perform optimally at intermediate salinity where their energy expenditure on osmoregulation would be least. Certainly some freshwater invertebrates (Kefford and Nugegoda, 2005; Kefford et al., 2006b) and fish (Konstantinov and Martynova, 1993; Bœuf and Payan, 2001) species have optimal growth at intermediate salinity but in other species growth is unaffected by salinity until some threshold is reached after which growth decreases (Bœuf and Payan, 2001; Hassell et al., 2006). Indeed, osmoregulation is probably more complex as it involves not just the regulation of total osmotic pressure but also the regulation of individual inorganic ions, intracellular vs. extracellular regulation and conforming, acid–base balance and a number of organic ions (Burton, 1991; Henry and Wheatly, 1992; Madsen et al., 1996; Patrick and Bradley, 2000). The ecological implications of these complexes in the physiology of osmoregulation have not been studied. For example it is not known whether it is individual ions, some function of several ions or the total ions that are setting the upper (and lower) salinity limits of species.

Salinisation can affect the organisms in several different ways from increasing stress to causing outright mortality, determining the viability of populations. The hatching rates of zooplankton diapausing (resting) eggs and the germination of aquatic plant seeds can be reduced by exposure to saline conditions (Nielsen et al., 2003; Bailey et al., 2004). In aquatic plants of *Potamogeton* sp. key morphological aspects related to functionality were significantly affected by salinisation (van den Brink and van der Velde, 1993). The growth of the clam *Corbicula fluminea* and *Ceriodaphnia* fecundity were significantly reduced by the effect of mining effluents (Kennedy et al., 2003). Silva and Davies (1999) registered an increase in invertebrates' oxygen consumption at intermediate salinities attributed to physiological stress, while beyond certain salinity (8.2 ppt) oxygen consumption sharply decreased due to the organisms' collapse. In the trichoptera *Hydropsyche exocellata* the magnitude of fluctuating asymmetry (i.e. random and small deviation from perfect bilateral symmetry in morphological traits) (Bonada et al., 2005) and catalase and enzymatic activities (Barata et al., 2005) were positively related to conductivity. The mosquito *Aedes aegypti* is known to decrease growth at high salinities, and this could be due to decreased feeding rates to avoid ingestion of ions at greater rates than can be eliminated by the excretory system (Clark et al., 2004). Conductivity can also affect key aspects of insect populations such as oviposition (Carver et al., 2009), pupation and emergence (Hassell et al., 2006). The baseline corticosterone levels of the Jefferson Salamander (*Ambystoma jeffersonianum*) significantly increased as a result of increased conductivity, indicating a stress induced response that could affect growth (Chambers, 2011). The frog *Rana clamitans* showed significant malformations when affected by road salt (Karraker, 2007). In fishes increased salinity has been reported to reduce food intake and conversion and growth and respiration rates (Bœuf and Payan, 2001). One common response of all aquatic animals to increased salinity is avoidance. Some species may remain in diapause during which the eggs or cysts can tolerate higher salinities, other species with some mobility can move to shallower depths that may allow survival and the more mobile organisms might migrate to areas with lower conductivity (James et al., 2003; Dunlop et al., 2005; Karraker, 2007). At an evolutionary level, it has been suggested that the organisms exposed to long-term salinisation might be more salinity tolerant (Kay et al., 2001), although a recent study comparing invertebrate communities of different regions around the world did not support that hypothesis (Kefford et al., 2012a).

3.1.2. Population/community/ecosystem

At a community level some species can be more competitive than others under high salinity conditions (Busse et al., 1999; Sarma et al., 2002; Kefford and Nugegoda, 2005). The salinisation of rivers may favour the physiologically tolerant taxa, which use the energetic cost of osmoregulation as an avenue to escape from the adverse influences of predation and competition (McEvoy and Goonan, 2003; Coring and Bäthe, 2011; Millán et al., 2011). Therefore, salinisation may enhance the colonisation of alien and exotic species while preventing the establishment of persistent populations of sensitive freshwater species (Braukmann and Böhme, 2011). After salinisation of the river system, several crustacean species colonised the River Weser (Germany) proceeding from its estuary (Bäthe, 1997) and exotic species were significantly associated to the highest salinity reaches of the Meurthe River (France) (Piscart et al., 2005b). This is particularly true with non-continental non-native species, which have to cope with high salinity during the translocation stage and are hence much more salt-tolerant than their native counterparts (Piscart et al., 2011). Salinisation can also promote the transmission of parasites within the river. For example the parasite *Polymorphus minutus* increased the tolerance of *Gammarus roeseli* to salinity stress thus increasing the parasite fitness (Piscart et al., 2007). The salinity-driven changes in the community composition can have effects on the trophic cascade, although only one case is documented. Dickman and Gochnauer (1978) found that salt stress caused a reduction in algal diversity due to a reduction in phytophagous grazers. Under normal conditions, these grazers preserve a higher algal diversity by preventing *Cocconeis placentula* (the dominant alga) from overgrowing and out-competing the other major algal species.

River salinisation can also have an impact on important ecosystem processes. Increasing NaCl concentrations led to the immediate release of NH_4^+ and Fe^{2+} from sediments of the River Murray floodplain (Australia) due to cations competition (Baldwin et al., 2006). The same study reported a significant change of the archael (methanogen) population with increasing NaCl loading corresponding with a significant decrease in methane production. Increasing SO_4^{2-} concentrations due to mining activity are known to stimulate microbial sulphate reduction leading to elevated HS^- concentrations (Bernhardt and Palmer, 2011). Beyond its direct phytotoxicity, HS^- can have major biogeochemical impacts by interfering with the Fe–P bounds (thereby releasing P) and by inhibiting nitrification (Bernhardt and Palmer, 2011), both of which processes contribute to stream eutrophication. The high concentrations of salinity can also lead to the sedimentation of suspended particles on stream substrates. This can affect periphyton growth (Hart et al., 1991) and may also result in increased light infiltration into the water column enhancing algal blooms (Dunlop et al., 2005). Breakdown of allochthonous organic matter can be reduced by salinity (Fritz et al., 2010; Schäfer et al., 2012). This could result in a lower carrying capacity of the ecosystem (e.g. less biomass in the system) and affect the provision of food to humans (e.g. fish). Salinity related changes in the habitat can also have implications for the ecosystem structure. After reducing the salt load the re-colonization of the salinised zone of the River Werra (Germany) by aquatic macrophytes led to a decline in phytoplankton (against a background of nearly unchanged nutrient contents) and structural changes in the macrozoobenthos community (Coring and Bäthe, 2011). Elevated salinity in riparian zones can diminish the presence of riparian vegetation, increasing the amount of light which enters the stream and changing the ecosystem from heterotrophic to autotrophic (Boulton and Brock, 1999; Millán et al., 2011). Moreover, riparian vegetation buffers the nutrient load of the stream from overland flows and its loss can alter the cycling of nutrients and organic contaminants (Dunlop et al., 2005). The ecosystem level response to secondary salinisation is therefore a complex function of the interactions between the chemical and biological compartments at different levels of organization, and understanding such interactions will require a multidisciplinary approach.

3.2. Response of different community parameters

Secondary salinisation produces structural changes in freshwater communities in terms of density, species richness and functional aspects. Although the response may differ among organisms, secondary salinisation usually leads to increased densities of r-strategist taxa and a reduction in species richness, resulting in lower diversity and evenness (Bunn and Davies, 1992; García-Criado et al., 1999; Petty et al., 2010; Schulz, 2010) and in a reduction of trait diversity (see below). River salinisation has been reported to adversely affect macrophyte cover (van den Brink and van der Velde, 1993; Nielsen et al., 2003) and insect abundance (Carver et al., 2009), while it can promote growth of bacteria (Dickman and Gochnauer, 1978) and phytobenthos (Coring and Bäthe, 2011) and the proliferation of euryhaline taxa (Bunn and Davies, 1992). High salt concentrations have been reported to reduce diatom (Busse et al., 1999), macroinvertebrate (Kefford et al., 2011; Cañedo-Argüelles et al., 2012), amphibian (Odum, 1988) and fish (Ferreri et al., 2004) species density, and to increase invertebrate drift (Wood and Dykes, 2002). As a consequence of reduced species density and richness and increased abundances of the most tolerant taxa, diversity is generally low in salinised rivers (Schulz, 2000). Over long time periods, genetic diversity may also be reduced and affect the ecosystem resilience (Dunlop et al., 2005). Nonetheless the relation between increased salinity and reduced diversity may not be linear. Some studies did not detect any strong response of diversity to salinity gradients below an EC of ≈ 10 mS cm^{-1} (≈ 5.6 ppt at 25 °C) (Horrigan et al., 2005). However, several other studies have found reduced species richness or diversity (Piscart et al., 2005b; Kefford et al., 2006a, Kefford et al., 2011) or changes in community (Kefford et al., 2010; Cañedo-Argüelles et al., 2012; Schäfer et al., 2012) well below this level. Piscart et al. (2005a) registered higher diversities at intermediate (0.42–1.24 mS cm^{-1}) than at low (0.24–0.34 mS cm^{-1}) or high (2.24–4.36 mS cm^{-1}) salinities when analysing net-spinning caddisfly assemblages along a secondary salinisation gradient. They suggested that this could be related to the intermediate disturbance hypothesis (Townsend et al., 1997), i.e. diversity would be highest at an intermediate level of salinity due to the co-occurrence of freshwater and halotolerant species. Kefford et al. (2011) observed that total macroinvertebrate species richness peaked at slightly elevated salinities (0.30–0.49 mS cm^{-1}) in south–east Australia, while it was reduced at both lower and higher salinity. Three hypotheses are given as to why such patterns in richness and salinity would be observed, including: some species having a physiological optimum at a slightly elevated salinity (discussed above); slightly elevated salinity might support both salt sensitive and tolerant species; and the possibility of confounding variables with salinity.

All of the studies regarding on how secondary salinisation may affect trait diversity, have been focused on aquatic invertebrates. Salinised rivers might be characterised by a trait composition usually adopted in variable environments (e.g. multivoltine cycles) or constraining habitats (e.g. asexual reproduction) (Piscart et al., 2006). Increased salinity is associated with a loss of grazer and shredder species in favour of predators and filter and deposit feeders (Marshall and Bailey, 2004; Piscart et al., 2006; Kefford et al., 2012b), a reduction in species with limited dispersal abilities (Schäfer et al., 2011; Kefford et al., 2012b), an increase in air-breathing species (Schäfer et al., 2011; Kefford et al., 2012b) and a modification of the reproduction mode (e.g. internal development of eggs or laying eggs out of the water to protect the young) (Piscart et al., 2006).

3.3. Response of different types of organisms

Besides the fact that the excess of single ions such as K^+ and Mg^{++} may cause specific toxic effects (Ziemann and Schulz, 2011), the tolerance to salinity can vary greatly among organisms. In general direct adverse localised effects to freshwater communities are expected to occur if salinity is increased to 1–3 mS cm^{-1} (Hart et al., 1991; Chapman et al., 2000; James et al., 2003; Böhme, 2011) although regional scale loss of species richness and change in community composition might occur at lower level (Environmental Protection Agency, 2010; Kefford et al., 2010, 2011; Merriam et al., 2011). The tolerance of some groups of organisms is much more documented than others, e.g. most of the information regarding the salinity tolerance of bacteria comes from studies in estuaries, which findings are of limited relevance for inland parts of rivers. Regarding algae, it is known that salinity can reduce the number of planktonic algae (Batterton and Baalen, 1971; Kipriyanova et al., 2007; Coring and Bäthe, 2011) and photosynthetic efficiency of epilithic algae (Silva and Davies, 1999). Zimmermann-Timm (2007) reported that diatoms react to changes in Cl^- as low as 100 mg L^{-1} ($\approx$0.14 mS cm^{-1}). Ziemann et al. (2001) registered a shift in the composition of the diatom assemblages of river Wipper after salt pollution and established that a maximum chloride concentration of 400 mg L^{-1} ($\approx$0.6 mS cm^{-1}) should not be exceeded to ensure the dominance of freshwater diatom species. Nonetheless detailed information on the salinity tolerance of algae is still lacking. A large proportion of macrophytes are sensitive to salinity concentrations between 1.5 and 3 mS cm^{-1} (Hart et al., 1991; James et al., 2003; Dunlop et al., 2005; Kipriyanova et al., 2007) although several freshwater species (e.g. *Ranunculus circinatus*) have been reported to be unaffected by salinities higher than that (van den Brink and van der Velde, 1993; Warwick and Bailey, 1997). Salinisation can affect the photosynthetic rate of aquatic plants too. In example Canadian waterweed (*Elodea canadensis*) reduces its net photosynthesis production at such low levels of salt as 100 mg Cl^- L^{-1} (Zimmermann-Timm, 2007). The information regarding zooplankton tolerances to salinity is scarce. Hall and Burns (2002) determined the 96 h LC50 of *Daphnia carinata* at 1400 mg Cl L^{-1} ($\approx$2 mS cm^{-1}), and most of the Australian rotifer and cladoceran wetland species have not been recorded in salinities higher than 25 mS cm^{-1} (Blinn et al., 2004). In the Chany Lake (western Siberia) zooplankton richness strongly decreased in the range of 0.3–3.0 g L^{-1} (Kipriyanova et al., 2007). Given their wide use as indicators of water quality and ecosystem health, there is much information concerning the salinity tolerances of stream invertebrates. Ephemeroptera, Plecoptera and Pulmonate snails are the most sensitive taxa. These taxa show 48-h and 72-h LC50 around 5–20 mS cm^{-1} (Williams et al., 2003; Hassell et al., 2006; Echols et al., 2010; Kefford et al., 2012a) and they have been rarely registered in salinities higher than 3 mS cm^{-1}. Ephemeroptera, Plecoptera and Trichoptera species richness (EPT) decreases over the entire salinity range (García-Criado et al., 1999; Kennedy et al., 2003; Hartman et al., 2005; Pond et al., 2008; Pond, 2010; Kefford et al., 2011). On the other side, Crustacea, Coleoptera and certain Diptera (e.g. Ceratopogonidae) and Odonata (e.g. Coenagrionidae) are among the most tolerant (Berenzina, 2002; Kefford et al., 2004b, 2006a; Dunlop et al., 2008). Shifts from salinity-sensitive taxa to communities with more tolerant taxa have been registered to occur between 0.8 and 1.0 mS cm^{-1} (Dunlop et al., 2005; Horrigan et al., 2005) and a significant reduction in species richness has been observed above 1.5 mS cm^{-1} (Kefford et al., 2011). With even smaller increases in salinity total species richness could increase and EPT richness decrease (Kefford et al., 2011), leading to changes in community (Kefford et al., 2010). Nonetheless, it should be noticed that the response of the invertebrate species richness might not follow a threshold model, but peak at slightly elevated salinity (0.3–0.5 mS cm^{-1}) (Kefford et al., 2011). Amphibians are particularly sensitive to salinisation as they are generally poor osmoregulators (Dunlop et al., 2005). However, given their mobility, adult frogs may escape localized rising salinity by dispersing to a more favourable environment and laying their eggs there (Viertel, 1999) but would be at greater risk where salinisation is more uniform. Several studies have confirmed adverse effects of salinities ranging between 2 and 3 mS cm^{-1} over the embryonic and larval stages of different frog species (Chinathamby et al., 2006; Sanzo and Hecnar, 2006; Karraker, 2007; Smith et al., 2007). Fish have broader salinity gradients, with juveniles showing optimal development below 6 mS cm^{-1} and adults below 13 mS cm^{-1} (James et al., 2003). Acute salinity tolerances of Australian freshwater fish have been registered to range between 2.7 and 82 mS cm^{-1} (Kefford et al., 2004c). However, the very early life stages of fish, such as sperm and eggs before hardening, have been shown to be more salt sensitive (Chotipuntu, 2003; Whiterod and Walker, 2006) and the ecological consequences of this will depend on seasonal pattern in breading and salinity variation. Furthermore lower salinity concentrations have been reported to cause adverse effects over the river fish fauna. Not exceeding the threshold salinity as established in laboratory toxicity tests of 95% of the fish species tested in the Murray Darling Basin (Australia) required salinities below 3400 mg L^{-1} of Cl^- ($\approx$5 mS cm^{-1}) (Muschal, 2006). In Germany, a threshold of 80 mg L^{-1} of potash concentration has been stated for fish toxicity in the Weser River (Bäthe and Coring, 2011) and diversity losses up to the 10th percentile were expected for fish at 750 mg L^{-1} of Cl^- ($\approx$1 mS cm^{-1}) in the Saale, Bode and Elbe Rivers (Böhme, 2011). In the Great Menderes Basin (Turkey) maximum conductivities of 6.9 mS cm^{-1}, resulted in the extinction of carp (*Cyprinus carpio*), which was previously the most abundant fish species, and the wels catfish (*Silurus glanis*) (Koç, 2008). We hypothesise that the outlined effects may also translate to effects on higher trophic levels such as aquatic birds, mammals and reptiles, but there is a paucity of studies on this issue. However, given that secondary salinisation leads to larger populations of just a few species, shorter and less complex food chains can be expected (Zimmermann-Timm, 2007).

3.4. Interaction of salinity with other environmental factors

The tolerance to salinity of freshwater organisms can vary depending on other environmental factors. Low temperatures have been reported to increase the salinity tolerance of the cladoceran *Daphnia carinata* (Hall and Burns, 2002), zebra mussel (*Dreissena polymorpha*) and quagga mussel (*Dreissena rostriformis bugensis*) (Spidle et al., 1994), the mayfly *Isonychia bicolor* (Kennedy et al., 2004) and salmonid fish (Bœuf and Payan, 2001). Water hardness can also alleviate Na toxicity. Kennedy et al. (2005) registered a strong inverse relationship between Na_2SO_4 toxicity and hardness ($CaCO_3$). Some alterations in pH can occur with an increase in salinity. The dominance of either hydrogen (H^+) or hydroxide (OH^-) ions will determine the buffer capacity of the river against increased salt content (Dunlop et al., 2005). However, Zalizniak et al. (2009) observed that acidic pH had no effect on *Physa acuta* salinity tolerance but an extreme alkaline pH (11) increased its salinity sensitivity. An increase in salinity can also have a strong effect on ion uptake and toxicity. For example, the increase in Cl^- results in the formation of cadmium chloride species that are of much lower toxicity than the free Cd^{2+} ions (De Wolf et al., 2004). It has been reported that fungi can tolerate Cd^{2+} and bacteria can tolerate Hg^{2+} in media with chlorine levels comparable to those occurring in oceans ($\approx$19 g Cl L^{-1}) (Babich and Stotzky, 1983). Also metal uptake by the submersed plant species *Elodea canadensis* and *Potamogeton natans* was reduced at higher salinities (Fritioff et al., 2005). The most common detected response has been a decrease in toxicity of chemicals with increasing but still non-stressful salinity (Hall and Anderson, 1995), although the toxicity of some organic compounds (e.g. pyrethroid insecticides) has been reported to increase with increasing salinity (Dyer et al., 1989; Hall and Anderson, 1995). Salinity in most cases is likely to reduce metal bioavailability due to complexation of major ions and hence reduction of the bio-available free metal. Moreover, salinity influences the distribution of especially polar pesticides between water and sediment or suspended particles via the salting out effect and thus may influence pesticide effects (Saab et al., 2011). However, a field study on the effects of pesticides found no interaction between salinity and pesticide effects (Schäfer et al., 2012). Most studies of salinity with other chemicals have concerned the effect of non-physiologically stressful salinity (to the species being studied) on its sensitivity to some other chemical toxicant. Practically nothing is known about the effect of physiologically stressful salinities combined with chemical toxicants, although presumably the effect of both would be greater than additive. Since salinity is often increased as a result of polluted effluents, e.g. coal mining effluents contain potentially harmful chemicals such as Selenium (Maggard, 2004), research should focus on the interaction of salinity with specific chemicals occurring in the respective point sources (e.g. mining, industrial activities).

The toxicity of salts and the relation between salinity and chemical compounds is dependent on the water ionic composition. Several studies demonstrated that different saline water types have different toxicity on aquatic invertebrates (Mount et al., 1997; Kefford et al., 2004c; Zalizniak et al., 2006, 2009; Ziemann and Schulz, 2011). Although the toxicity of many proportions of ions remains to be reported, the following generalisation can be made. Pure NaCl is more toxic than sea water despite the latter being about 85% NaCl (Kefford et al., 2004c). Saline waters with low Ca being more toxic (at least chronically) than saline waters with high Ca concentrations (Zalizniak et al., 2006; Zalizniak et al., 2009). Moreover, increasing the number of cations tends to reduce toxicity (Mount et al., 1997). Furthermore the effect of different ionic composition may differ between acute and chronic exposure, with Zalizniak et al. (2006, 2009) observing no difference between several ionic compositions for acute exposure but significant differences with chronic exposures.

4. How is secondary salinisation managed?

4.1. Legislation

Although the adverse effects of salinisation have been widely recognised, legislation is generally flexible when it comes to establish limits for salt concentrations in rivers. In some cases this flexibility is a consequence of the economic, political and social power of the polluting industries. West Virginia (USA) is a clear example. West Virginia's low-sulphur coal accounts for more than the 50% of the electricity used in the United States and the coal industry owns 75% of the land in West Virginia's major coal producing counties (Fox, 1999). This has led to one of the most liberal environmental and labour regulations in the United States and the designation of West Virginia as an environmental sacrifice zone (Fox, 1999; Palmer et al., 2010). During the last decade some regulatory advances have been achieved and mitigation is now required for all coal mining activities authorised under Section 404 (Fritz et al., 2010). Even though current mitigation strategies have been proved unsuccessful when trying to restore the ecosystem functions (Palmer et al., 2010).

In Australia and New Zealand, the Bilateral Water Quality Guidelines (ANZECC/ARMCANZ, 2000) provide regional salinity concentrations which range from 0.02 to 5 mS cm^{-1} depending on the type of river (upland or lowland) and the region. These values serve as guidance for river management, but definite thresholds for salinity are not specified. Regional management schemes also exist in Australia, e.g. licenced discharges of saline water from mines and power stations into the Hunter River catchment above Singleton are being managed so that the EC does not exceed 0.9 mS cm^{-1} (Muschal, 2006). In terms of drinking water the 0.8 mS cm^{-1} is typically set as the upper limit in Australia (MDBMC, 1999) with this value derived from the World Health Organisation guidelines, the latest version of which state that: "The palatability of water with a total dissolved solids (TDS) level of less than about 600 mg L^{-1} is generally considered to be good; drinking-water becomes significantly and increasingly unpalatable at TDS levels greater than about 1000 mg L^{-1}" (WHO, 2011).

In Europe no legally prescribed environmental quality standards (EQS) exist for salt. Eutrophication and acidification have been the focus of the European Water Framework Directive (WFD) (European Commission, 2000), aimed to preserve the good ecological status of all European water bodies. Although the salinisation of European Rivers might compromise the possibility of achievement of the directive's aims in several river basins, e.g. Ebro (Isidoro et al., 2006) and Danube (Mádl-Szőnyi et al., 2008), little attention has been paid to this issue during the development of water quality indicators and the establishment of EQS values. This is because salinity has not been perceived as a major problem in most of Europe. While this may be so in many European regions, there are important exceptions including: most of southern Europe with a Mediterranean climate (Barata et al., 2005; Prat and Rieradevall, 2006), northern and alpine regions where road deicing is extensive (Ruth, 2003) and regions with salt mining (Martín-Alonso, 1994; Piscart et al., 2005b; Bäthe and Coring, 2011; Coring and Bäthe, 2011). Consequently, sound methodological standards, mandatory guidelines and best practises for prediction and evaluation of adverse impacts of river salinisation are lacking in Europe (Böhme, 2011). An impact assessment methodology is usually developed at a regional level by the responsible authorities in the regions where saline discharges are recognised as an important ecological issue. In Germany the reference concentration referring to "good" condition sensu WFD is considered to be 200 mg Cl^{-} L^{-1} ($\approx$333 g NaCl g L^{-1}), but areas with active mining or long mining traditions represent an exception (Richter et al., 2010). Since permissions to dispose of saline waters are generally a legacy of less rigorous environmental standards, today's threshold values are decided for individual cases and may be much below 200 mg Cl^{-} L^{-1}. In Spain water quality classes have been established according to river conductivity depending on the river typology, with the conductivity threshold for the good/moderate boundary ranging from 0.15 (for high mountain calcareous rivers) to 2.2 mS cm^{-1} (for rivers in La Mancha region) (ORDEN ARM/2656/2008, 2008). In France, the System of Evaluation of the Water Quality of Running Waters (Système d'évaluation de la qualité de l'eau des cours d'eau, SEQ) considers the electrical conductivity as a parameter for drinkable water with a higher limit of 2.5 mS cm^{-1} referring to a "good" condition. Furthermore, ion composition in the electrical conductivity is also considered with a limit of 200 mg L^{-1} for NaCl, 160 mg L^{-1} for Ca^{2+}, and 50 mg L^{-1} for Mg^{+}.

In the USA the United States Environmental Protection Agency (EPA), which is responsible for drinking water regulation in the United States, includes total dissolved solids (TDS) as a secondary standard, meaning that it is a voluntary guideline in the United States (Environmental Protection Agency, 1988). The actual water quality standard for the palatability of drinking water is 500 mg TDS L^{-1} (Safe Drinking Water Act). Most state-specific water quality standards contain some consideration of effects of high ion content, although following the respective guidance does not always require meeting all water quality criteria (Goodfellow et al., 2000) and no national water-quality criteria for the protection of aquatic life has been defined for Na, SO_4^{2-} or total dissolved solids (TDS) concentrations (United States Environmental Protection Agency, 1999). In South Africa, in spite of the recognised impact of salinisation and the availability of an extensive database on local salinity tolerances (Palmer et al., 2004), freshwater guidelines do not currently treat salts as toxicants.

4.2. Management of river salinisation

Salinisation has important impacts on the goods and services that rivers provide to humans and therefore it may have high economic costs. For example, the salinisation of the Ganges water in Bangladesh has resulted in losses of millions of dollars related to crop and industrial machinery damage and posed a risk to human health (Mirza, 1998). As outlined, concentrations of chloride higher than 250 mg L^{-1} ($\approx$420 mg NaCl g L^{-1}) have been recognised as not potable for human consumption in USA (Environmental Protection Agency, 1988; Environment Canada, 2001) and a conductivity of 2.5 mS cm^{-1} has been established as the limit for water for human consumption (which includes the water used in the food industry) in Spain (RD 140/2003, 2003) and in France (SEQ Eau V2). Salt pollution has been reported to cause problems for the drinking water supply of some cities (Braukmann and Böhme, 2011), therefore requiring additional processing that increases supply costs. In dryland regions, river salinisation can have major impacts (Smedema and Shiati, 2002). Rivers are often the only permanent and reliable source of water and salinities of 1.5 mS cm^{-1} and higher make the water unsuitable for irrigation of most crops (Mirza, 1998; Muschal, 2006).

Many strategies have been adopted to manage river salinisation, some are aimed to prevent further salinisation and decrease salt discharges, while others are aimed to decrease river salinity and minimise impacts on the ecosystem and its derived goods and services (Williams, 2001). The first management decision that has to be taken is to what extent the salinity disposal can be prevented and what is the trade-off (e.g. improving the river ecological quality versus a reduction in the mining activity that could be associated with a loss of economic activity). If the disposal cannot be prevented there are some techniques that can be used (e.g. integrated wastewater control systems using plate dolomite for ion exchange) to decrease the salinity of the water being disposed, but they need to be effective, technically and legally feasible and do not lead to disproportionate ecological (waste, energy) or economic costs (Richter et al., 2010). The most cost-effective techniques are related to maximising the non-river disposal opportunities (e.g. evaporation ponds) and the dilution capacity of the river (i.e. disposal of salts during high flow, when the river's dilution capacity is highest). Another option is to transfer water from non salinised rivers in order to increase the dilution capacity of the river (O'Keeffe and De Moor, 1988), but this can have important ecological, political and social consequences that need to be considered.

A useful tool to guide management decisions is probabilistic risk assessment. Risk-based techniques are based on species sensitivity distributions (Posthuma et al., 2002), which are derived from laboratory toxicity tests (96 h LC50 values), sub-lethal effects or maximum field distributions of the species (Kefford et al., 2004a; Horrigan et al., 2005). When these data are contrasted to the real salinity values of the catchments, ecosystem protection values (trigger values) can be calculated to ensure the protection of a predefined proportion of taxa (Dunlop et al., 2005; Environmental Protection Agency, 2010). Nonetheless these techniques require a large amount of data regarding toxicity or field distribution of the species, and some recent studies suggest that it would be more useful to prevent changes to ecological communities rather than protect species from exceeding their physiological sensitivity (Kefford et al., 2010). It is important to notice that once impacts on the ecosystem have occurred it may be difficult or even impossible to restore it to its initial condition (Landis et al., 2000; Scheffer et al., 2001; Lovett et al., 2007; Duarte et al., 2009). Therefore identifying thresholds of salt stress that produce ecological impairment is required (Petty et al., 2010). If freshwater biodiversity should be preserved and ecosystem goods and services secured, impacts need to be anticipated and integrated catchment strategies need to be adopted. A good example of a successful management strategy is the Hunter River Salinity Trading Scheme upstream of Singleton. The scheme established the total allowable salt discharge from mines and power stations as a function of the ambient salinity in the river (members of the scheme coordinate their discharges to guarantee river conductivity < 0.9 mS cm^{-1}) and a salt credit trading was adopted giving each licence holder the flexibility to increase or decrease their allowable discharge while limiting the combined amount of salt discharged across the valley (Department of Environment and Conservation, 2003).

4.3. What are the expected future trends?

Climate change is likely to increase river salinity in some regions. An increase in water temperature and thus evaporation is expected (Hengeveld, 1990; Arnell and Reynard, 1996; Sereda et al., 2011), and a decrease in the amount of precipitation is forecasted in several regions including Central America, Northern and Southern Africa, most of Australia, Middle-East countries, Southern Europe, and Southern of the USA (Kundzewicz et al., 2008). This will result in a decrease in the catchment runoff (Nielsen and Brock, 2009), which has been predicted for several major river basins like Amazon, Congo, Danube, Nile, Orinoco, Parana or Yangtze (Arora and Boer, 2001; Nijssen et al., 2001). Reduced river discharges imply a lower dilution capacity that could be translated to higher salinity concentrations (Crowther and Hynes, 1977), especially in dry and Mediterranean climates where prolonged droughts are common. Thus, in the context of global warming the areas threatened by secondary salinisation will most likely expand. Nonetheless climate effects on stream salinity are difficult to predict because they involve predictions for precipitation and temperature patterns, and the dynamic interactions between ground and surface water (McNeil and Cox, 2007). The situation is likely to be aggravated by increasing water demand for human consumption, agriculture and industry, which will also reduce the dilution of salt effluents (Schindler, 1997; Schmandt, 2010). In fact, the increasing human population already exceeds the capacity of surface and ground water to sustain human activity in some regions (Grimm and Fisher, 1992; Vörösmarty et al., 2000; Wong et al., 2007). Kaushal et al. (2005) predicted that baseline chloride concentrations in many rural streams in the USA would exceed 250 mg L^{-1} ($\approx$417 mg NaCl L^{-1}), thereby becoming toxic to sensitive freshwater organisms and not potable for human consumption. Although not due to climate change, the Australian Dryland Salinity Assessment (NLWRA, 2000) predicted that 3.1 million ha of land will be affected by salt by the year 2050 and up to 20,000 km of streams could be significantly salt-affected over the next 20 years. In the Murray River (Australia) it has been predicted that riverine salinity will exceed drinking water standards for nearly 150 days a year (MDBMC, 1999). Moreover increasing energy demands are likely to increase mining activity, e.g. coal consumption for electricity is expected to increase 42% from 2008 to 2030 (US Department of Energy, 2008). Therefore, the future predictions clearly indicate that river salinisation will globally increase (i.e. more streams will be impacted and the salt stress will increase in already degraded streams).

5. Future research

At an individual level the true energetic cost of osmoregulation remains under debate, and relevance of osmoregulatory studies to ecological effects in nature is unclear. In particular physiological studies have been undertaken to understand how organisms deal with salinity changes at the sub-individual level and have used environmentally unrealistic exposure scenarios (e.g. exposure to pure NaCl). Consequently, we are currently not able to predict which salinity will be optimal (i.e. not cause adverse effects) for freshwater organisms in nature.

There is also a need to better understand how effects of salinity on individual organisms are affected by the ionic composition of the salinity and by other abiotic stressors. Ionic composition is particularly problematic as there are eight major ions (Na^+, Ca^{2+}, Mg^{2+}, K^+, Cl^-, $SO_4{}^{2-}$, $CO_3{}^{2-}$ and $HCO_3{}^-$). Here, we hypothesise based on previous findings (Mount et al., 1997; Kefford et al., 2004c; Zalizniak et al., 2006, 2009; Ziemann and Schulz, 2011) that the relative proportion of the ions is more predictive of effects than the total concentration in terms of salinity. For example, Sylvestre et al. (2001) reported for Bolivian lakes that the abundance of several diatom species was likely related to the concentration of minor ions. Given that ionic proportions of total measured salinity can vary in space and time an understanding of their effects is critical for managing their management. This holds as well for the interaction of effects of additional stressors with physiologically stressful salinity concentrations.

At a community level more information is needed regarding the relationship between salinity and species richness. Although many studies have claimed to consider this relationship, most have actually considered species density – number of species per unit area or sample – while species richness refers to the number of species in a defined space or habitat of interest (Gotelli and Colwell, 2001). A study that did consider changes in stream macroinvertebrate species richness with salinity in south–east Australia (Kefford et al., 2011) found that there was a complex relationship, with different response in total species and EPT species richness to salinity. Further studies are need in other regions and with other groups of organisms in order to confirm the results obtained from these first studies and to allow for robust conclusions. However, we hypothesise that the species richness of salt sensitive groups (e.g. EPT) will decrease with increasing salinity, while other groups will have maximum species richness at slightly elevated salinities.

Also at the community level there is a need to identify traits of organisms that are associated with resistance or susceptibility to salinisation in order to predict community changes by salt pollution, and there is very little information regarding the impact of river salinisation on trophic interactions. This information is extremely important since it could allow for the determination of the ecosystem response. In brackish-water lagoons it was found that salinity had a cascading effect that induced an hysterical response of the ecosystem, which shifted from a clear water state (macrophyte dominance) to a turbid water state (phytoplankton dominance) (Jeppesen et al., 2007). Such state shifts have important management implications, since it may be impossible to restore the initial conditions of the system and could occur as a result of secondary salinisation.

Finally, the impact of river salinisation on ecosystem goods and services has not been quantified yet. There is a pressing need to consider how salinity is associated with ecosystem functions and services with only one published study on this topic (Schäfer et al., 2012). We need to understand and communicate the economic, environmental and social costs of river salinisation in order to guide management and restoration efforts. Impacts need to be anticipated and mitigated, and future scenarios of climate change and increasing water demand have to be integrated into the ecological impact assessment.

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