
Hamish John Appelby (IWMI)

CHAPTER 6. SALTS

Javier Mateo-Sagasta and Joost Albers

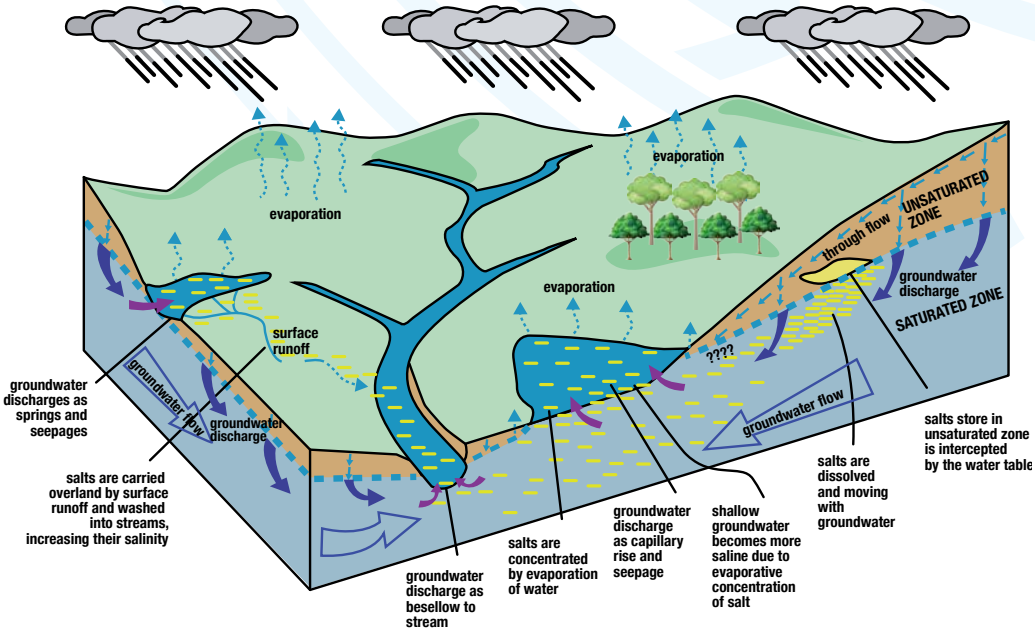
Several decades ago there were few constraints to the disposal of drainage water from agriculture. Now, human-induced salinization of freshwater bodies is a challenge of growing concern with major potential economic impacts, particularly in arid and semiarid areas. The 1260 km^3 of return flows that agriculture is estimated to generate globally every year (FAO AQUASTAT) could result in the mobilization and transport of billions of tonnes of salts to freshwater bodies. The agriculture-induced intrusion of saline groundwater or seawater to freshwaters adds to the problem, especially in coastal areas.

This chapter will briefly review the main processes responsible for salt mobilization with a focus on human-induced salinization of freshwater and with particular attention to salts mobilized by irrigation. The chapter aims to provide a concise review of the extent of salts mobilized by agriculture and consequent effects on human and ecosystem health.

6.1 Agriculture-induced salt loads to water

Salinity is a measure of the quantity of dissolved salts in water, also known as total dissolved solids or total dissolved salts (TDS). Freshwater bodies can receive salt through different pathways, for example through direct surface runoff from saline lands, subsurface drainage of saline waters to fresh water bodies, or the interception of saline stores due to the elevation of the ground water table that also may recharge surface waters (see Figure 6.1). Salts may degrade water quality in fresh water bodies such as wetlands, streams, lakes, reservoirs and estuaries as a result of salt mobilisation and concentration.

FIGURE 6.1 | Salinization processes of soil and water bodies and the hydrological cycle



Source: New South Wales Government, 2013.

Table 6.1 provides an overview of categories of salinity and accompanied concentration levels as proposed by Freeze and Cherry (1979). Dissolved salts typically include ions such as sodium (Na^+), chloride (Cl^-), potassium (K^+), magnesium (Mg^+), sulphate (SO_4^{2-}), calcium (Ca^{2+}) and bicarbonate (HCO_3^-). These salts accumulate in the soil profile over time in areas where evaporation levels are higher than precipitation levels and, eventually, may be washed out to water bodies or percolate to groundwater (Mateo-Sagasta and Burke, 2010).

Table 6.1 | Classes of salinity and salt concentration levels

Class name	Class limits (TDS range, in mg/l)		
Fresh water		<	1 000
Brackish water	1 000	-	10 000
Saline water	10 000	-	100 000
Brine		>	100 000

Source: after Freeza and Cherry, 1979.

Salinization of freshwater has many different causes that could be categorized as natural or human-induced. Natural salinity refers to the 'primary' salinity that was present prior to the development of land for agriculture. Human-induced salinity refers to the 'secondary' salinity often caused by changes in land use.

6.1.1 Natural salinization of freshwater

Natural salinization of freshwater occurs when salts enter bodies of water through natural processes:

- natural inflow of salt groundwater into freshwater aquifers;
- weathering of salt containing rocks within the catchment due to precipitation;
- single submergence event of soils under seawater in coastal areas; and
- atmospheric precipitation, both coastal and inland, of rainwater that includes dissolved salts coming from evaporated seawater.

The rates at which these natural processes occur depend on factors such as climate and geology (Ghassemi, Jakeman and Nix, 1995; Herczeg, Dogramaci and Leaney, 2001; Post and Abarca, 2010; Williams, 2001; Williams, 1987).

6.1.2 Human-induced salinization of freshwater

Human-induced freshwater salinization, or 'secondary salinization' (Williams, 2001; Cañedo-Argüelles, 2013), is often attributed to land-use change, poor land management and agricultural activities (including irrigation and drainage). However, salinization can also occur as a result of discharges of municipal or industrial wastewater, salt mining, de-icing of roads and leaking canals and reservoirs (Anning and Flynn, 2014).

Land-use change may involve the clearance of native vegetation to use the land for crops and pastures, which have increased by 460% and 560% respectively in the last 300 years at the expenses of forests and grasslands (Klein Goldewijk, 2001). This change in land use has decreased evapotranspiration and increased aquifer recharge (by two orders of magnitude) and streamflow (by one order of magnitude) but also degraded water quality by mobilization of salts and salinization caused by shallow water tables (Scanlon *et al.*, 2007). In addition, when native vegetation has long root systems, these can take up shallow water and thus prevent a rise in groundwater. When native vegetation is cleared and replaced by shallower-rooted crops and pastures, net evaporation declines, which

results in a rising groundwater table. When the underlying groundwater is saline and rises to the surface, water bodies are recharged with saline water (Mateo-Sagasta, 2010) A well-known example of this is the wetland adjacent to the Murray river in Australia, where salinity increased after the clearance of native vegetation (Walker, Bullen and Williams, 1993).

Human-induced water salinization can be also specifically related to agricultural irrigation and drainage in multiple ways. For example:

i) excessive irrigation can raise water tables from saline aquifers and this can increase seepage of saline groundwater into water courses and increase their salinization;

ii) salts accumulated in soils (particularly in arid and semiarid areas) can be mobilized by irrigation with the application of leaching fractions for soil-clearing. Soil leaching entails allowing an excess portion of the irrigation water to carry salts away through drainage schemes. Drainage water is typically 4-10 times more saline than irrigation water but, when reclaiming already salinized soils, drainage water will be much more saline (e.g. 50 times more than irrigation water). This effluent risks salinization of receiving water bodies (van Hoorn and van Alphen, 2006);

iii) overexploitation of groundwater for agriculture in coastal areas, which results in sea water intrusion into freshwater aquifers;

iv) excessive fertilizer application may increase the concentration of salts in drainage water in irrigated areas and also in run off and percolation in rain fed areas.

In irrigation systems, salt, once mobilized, can be transported and discharged to surface drainage and river systems as a result of groundwater seepage, surface runoff, engineered subsurface drainage and irrigation channel outfalls (Duncan *et al.*, 2008). Furthermore, salt discharge will change over time as a result of both climatic and management influences.

Table 6.2 shows examples of salt loads from irrigated lands in different global locations. Such data are not well documented and are available only for some countries, mostly in arid and semiarid regions, where soil and water salinization are typically of greater concern than in humid areas. As illustrated by the table, salt mobilisation varies widely between regions and irrigation areas, even when these have similar climatic conditions. The load of salt exported per hectare of agricultural land depends on the drainage volumes (which in turn depend on the irrigation management practices and water use efficiency) and the concentration of salt in drainage water (which depends

Table 6.2 | Salt loads from irrigation return flows in different arid and semiarid areas

Location	Salts in irrigation return flows (TDS)	Year	Source
Different irrigation districts, Ebro river basin, Spain	2-16 Mg/ha/year	1990-2004	Causape *et al.*, 2006
La Violada irrigation district, Ebro river basin, Spain	19.3 Mg/ha/year	80'-90'	Barros, Isidoro and Aragüés, 2012
La Violada irrigation district, Ebro river basin, Spain	9.9 Mg/ha/year	2006-2008	Barros, Isidoro and Aragüés, 2012
Lerma watershed, Ebro river basin, Spain	1.3-5.8 Mg/ha/year	2004-2008	Abrahao *et al.*, 2011
Harat Plan, Yazd Province, Iran	1.0-35.4 Mg/ha/year	2009	Jafari *et al.*, 2012
Colorado River Basin, USA	26-41 Mg/ha/year	1975-1984	Duncan *et al.*, 2008
Shepparton irrigation region, Murray-Darling river basin, Australia	0.04-0.66 Mg/ha/year	2003	Duncan *et al.*, 2005
Kerang irrigation region, Murray-Darling river basin, Australia	3.7-10.1 Mg/ha/year	1997-2003	Duncan *et al.*, 2005
Murrumbidgee/ Coleambally irrigation areas, Murray-Darling river basin, Australia	0.1-2.3 Mg/ha/year	1995-1997	Duncan *et al.*, 2005

on factors such as the soil salinity or saline groundwater seepage) (Causape *et al.*, 2006; Duncan *et al.*, 2008). In areas where evapotranspiration is higher than precipitation, salts tend to accumulate naturally in the soil profile and, with irrigation or after heavy rains, salts are mobilized and loads from farms to downstream waters tend to be high (Abrahao *et al.*, 2011). In terms of irrigation practices, efficient irrigation methods, such as drippers and sprinklers, reduce return flows and, therefore, overall loads.

For example, in La Violada irrigation district, in the Ebro river basin (Spain), investments in adequate management of irrigation water reduced by half the salt exported from the irrigation district between the 1980s and 2006-2008 (Barros, Isidoro and Aragüés, 2012). In the Shepparton irrigation region, Murray-Darling River Basin, Australia, the salt loads were kept low thanks to the low volumes of drainage water and the low concentration of salt in drainage. The low concentration of salt in drainage was mainly due to the low salinity of irrigation water (0.06 dS/m) and to the low contribution of groundwater seepage to irrigation return flows, which is due to low connectivity between surface and groundwater systems and relatively good groundwater quality (Duncan *et al.*, 2005). In other regions such as the Colorado River Basin, in the United States of America, high volumes of highly saline drainage water are discharged. This is associated with the inefficient use of water at farm level and substantial losses during water conveyance, and sometimes with the displacement of saline groundwater through deep percolation of irrigation return flows (Duncan *et al.*, 2008).

The contributions to water salinization by aquaculture and livestock (excluding the production of animal feed) are minor compared to irrigated crops, with only localized effects where livestock and aquaculture are more intense.

6.2 Salinization of soils, groundwater and surface waters

6.2.1 Soils

Irrigation causes salinization of soils in many parts of the world (Figure 6.2) and where soils are salinized, water salinization is an accompanying problem. Worldwide, an estimate 24 percent of the area under irrigation is affected by salinization and water logging in the broadest sense. This equates to 65 million ha, of which 34 million ha faces severe salinization (Mateo-Sagasta and Burke, 2010). Asia and the Americas experience the greatest reported area salinized due to irrigation (Table 6.3). At a country level, Pakistan (7 Mha), China (6.7 Mha), United States of America (4.9 Mha), India (3.3 Mha), Jordan (2.3 Mha), Uzbekistan (2.1 Mha), Iran (2.1 Mha), Iraq (1.8 Mha), Turkey (1.5 Mha), and Turkmenistan (1.4 Mha) lead the absolute rankings (FAO-AQUASTAT).

6.2.2 Freshwater

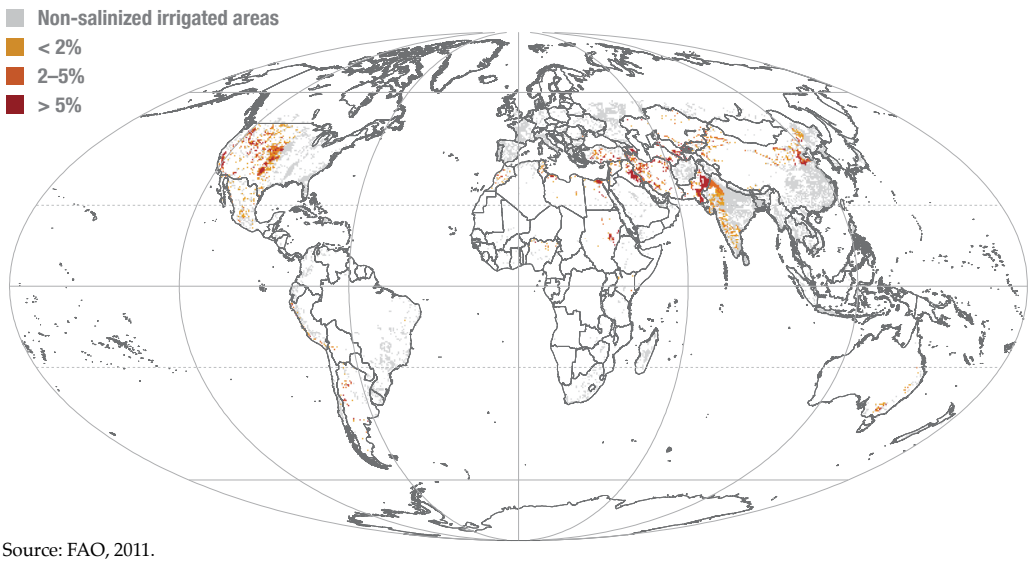
Freshwater salinization is a major environmental problem affecting surface and ground water. Water scarcity is rising, over-abstraction of groundwater occurs in many places, salinization of freshwater bodies is increasing, and aquifers are intruded by seawater in several different coastal areas (FAO, 2011). Salinization of freshwater systems mainly

Table 6.3 | Area salinized by irrigation per region

Region	Million ha
South Asia	10.30
East Asia	6.70
Western Asia	6.12
Northern America	5.34
Central Asia	3.21
Southern America	0.95
Sub-Saharan Africa	0.68
Northern Africa	0.68
Australia and New Zealand	0.20
Total	34.19

Source: AQUASTAT, different years.

FIGURE 6.2 Land salinization due to irrigation. Legend shows the percentage of land salinized by irrigation



Source: FAO, 2011.

affects groundwater (IGRAC, 2009) but also rivers (Cañedo-Argüelles *et al.*, 2013), wetlands (Herbert *et al.*, 2015) and reservoirs (Meybeck, 2004).

Many examples exist of irrigation causing increased salinity of rivers: the Breede River in South Africa (Scherman, Muller and Palmer, 2003), the Amu Darya river in Central Asia[7] (Crosa *et al.*, 2006) and the Murray-Darling River system in Australia. In addition, there are coastal aquifers that have already been permanently salinized, for instance, in Gaza, Gurajat, some coastal areas in Mexico or West Java (Mateo-Sagasta and Burke, 2010). In the Great Menderes river in Turkey, increased salinity has resulted in the extinction of carp (*Cyprinus carpio*) andcatfish (*Silurus glanis*) (Koç, 2008). The broader picture is harder to discern. Despite the existence of many well-documented cases, there is not enough information with the correct geographical and temporal resolution to construct a systematic and quantitative global assessment of surface water salinization. UNEP (2016) tried to address the limitations in the availability of global data with a water quality modelling effort for Latin America, Africa and Asia, using various assumptions and proxies to overcome gaps in the data. This exercise suggests that severe (> 2000mg TDS/l) and moderate (450-2000mg TDS/l) salinity pollution affects around one-tenth of all river stretches in these three regions. The assessment attributed most of the salt loading to irrigation return flows in Africa and Asia, while in Latin America most loadings were attributed to the manufacturing industry.

6.2.3 Groundwater

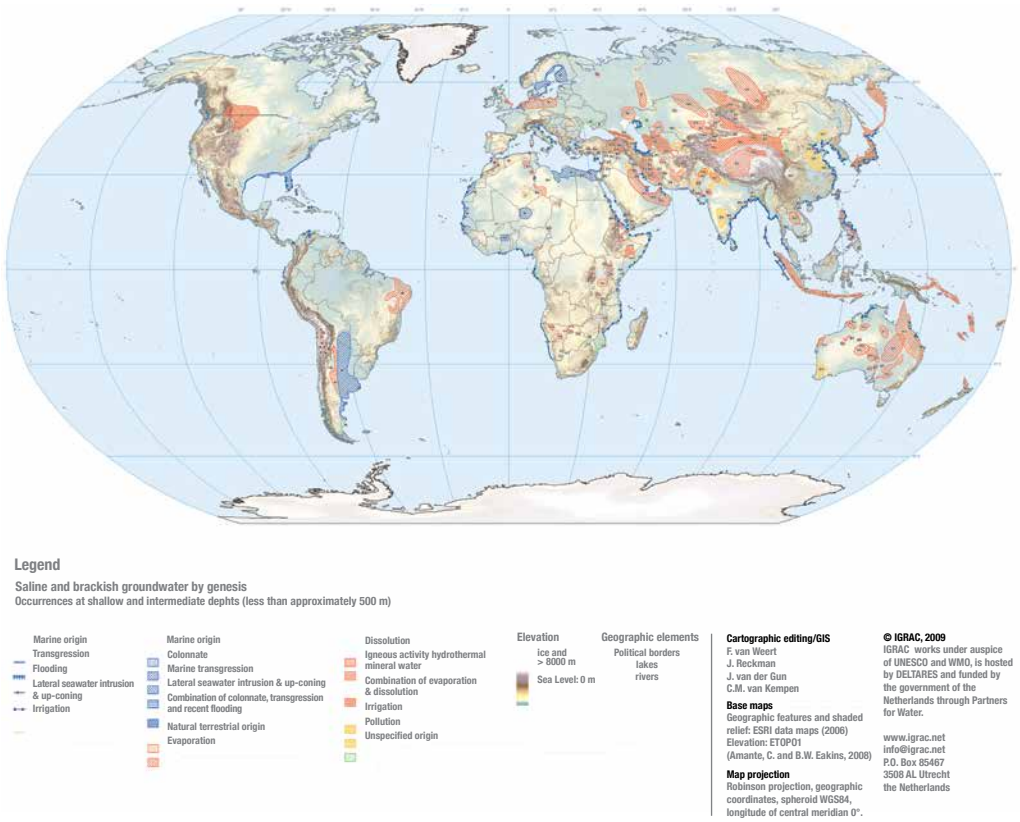
Data are too patchy to allow a quantitative global assessment of groundwater resource status (Foster *et al.*, 2013). Nevertheless, the available data suggest that saline water from irrigation is probably one of the most widespread causes of groundwater quality

BOX 6.1 Enhanced salinity in the Aral Sea

In 1960, to promote agriculture, the Soviet government decided to establish dams and extensive irrigation programs along the Syr Daria and Amy Daria rivers, which drain into the Aral Sea. These two rivers delivered four-fifths of the water to the Aral Sea, while one-fifth came from rainfall. As a consequence of the dams and irrigation, the level of the Aral Sea dropped by 20 m and the volume shrank from 1060 km^3 to 210 km^3 between 1960 and 1998. Salinity rose from 10 g/l to 100 g/l in the southern part of the Aral Sea, varying over time with precise location (Thompson, 2008). The salinization process has been accelerated by positive feedback arising from stratification of salts and temperature in the Aral Sea.

[7] See also Box 6.1.

FIGURE 6.3 | Saline groundwater occurrence and genesis



Source: IGRAC, 2009.

deterioration (Morris, 2003; IGRAC, 2009). In 2009, approximately 1.1 billion people lived in regions that have saline groundwater at shallow and intermediate depths. In these areas, groundwater salinity is mainly caused by irrigation and seawater intrusion and, to a lesser extent, by dissolution and igneous processes (IGRAC, 2009). In Figure 6.3, yellow spots indicate substantial saline groundwater caused by irrigation in North America (USA), the Middle East (Azerbaijan, Iraq, Syria and Turkey), Asia (China, India, Pakistan, Tajikistan, and Uzbekistan) and south-west Australia.

6.3 Impact on the environment, human health and economy

Agriculture-induced salinization of waters can affect environmental health (including biodiversity and ecosystem functions), economic activities (and especially crop production) and human health. Each of these impacts is discussed below.

6.3.1 Environmental health

Highly saline freshwaters alter the geochemical cycles of other major elements, e.g. carbon, nitrogen, phosphorus, sulphur, silica and iron. Salinized water can potentially increase release of nitrogen, phosphorus and silica, which could: enhance eutrophication instream and downstream; disrupt natural processes like denitrification; reduce storage and increase mineralization of carbon; and increase generation of sulphur compounds, which are toxic to plants and animals. However, the extent to which biogeochemical cycles are altered depends on soil and water chemistry, the magnitude and timing of salinization, hydrology, availability of substrate, and reaction of the biota community to higher salinity concentrations. For more information on the effects of salinity on biogeochemical cycles see Herbert *et al.* (2015).

While in some cases, salinized waterbodies (e.g. wetlands) maintain very high levels of biodiversity, in general when salinity rises, biodiversity of all forms—including microorganisms, algae, plants and animals—declines (Pinder *et al.*, 2004; Pinder *et al.*, 2005; Lorenz, 2014). Salinization can affect freshwater biota at three levels: changes within species, changes in community composition, and eventually biodiversity loss and migration. High concentrations of sodium and chloride ions in freshwater causes accumulation of toxic salts in the cells of plant species, which disturbs the uptake of water and important ions and eventually leads to death (Kozlowski, 1997). Other consequences of freshwater salinization are changes in behaviour, food uptake, growth, germination, seedling survival and reproduction (Herbert *et al.*, 2015).

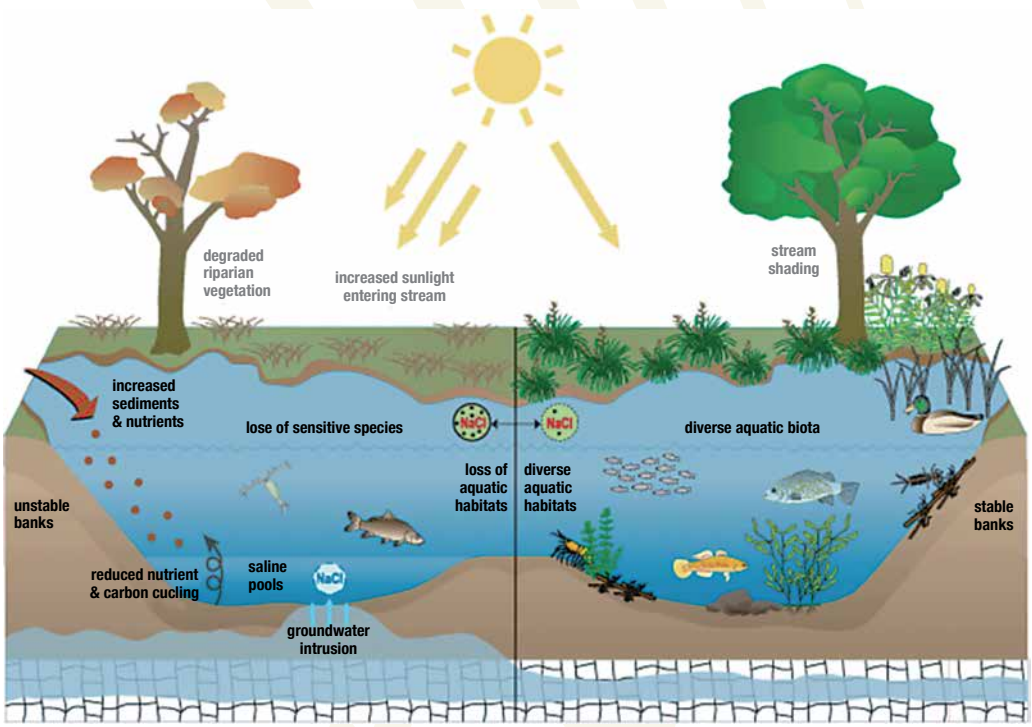
As salinity increases, saline-sensitive species are replaced by more salt tolerant species (Pinder *et al.*, 2004; Nielsen *et al.*, 2003). Increased salinity also provides the opportunity for salt-tolerant invasive species to take hold (Thouvenot, Haury and Thiébaut, 2012). For example, in the Aral Sea freshwater species disappeared as a consequence of increasing salinity from 1960 onwards. By 1990, five fish species were still left, of which only one was indigenous (Kolar and Lodge, 2000). Some freshwater species can be very sensitive to increases in water salinity, even when the water can be acceptable for drinking purposes and irrigation. For example, electrical conductivity (a proxy for salinity) of 2 mS/cm) can displace many freshwater insect species (Cañedo-Argüelles *et al.*, 2016).

One of the first signs of salinization is the disappearance of riparian vegetation and macrophytes, because salts accumulate in the root system and hinder the plants' uptake of water and nutrients (Williams, 2001; Dunlop, McGregor and Horrigan, 2005). Motile species may attempt to avoid increased salinity by migrating to areas with less saline water (Cañedo-Argüelles *et al.*, 2013). For example, fish may move to shallow water where conductivity (salinity) is lower (Dunlop, McGregor and Horrigan, 2005).

Ultimately, salinity negatively affects ecosystem function as a result of positive feedback loops induced by altered geochemical cycles, species healthiness, community composition, or biodiversity loss and migration. Figure 6.4 depicts an overview of salinity impacts on freshwater ecosystems. In the long-run, genetic diversity might be reduced, thereby reducing ecosystem resilience to external shocks and disturbance (Dunlop, McGregor and Horrigan, 2005).

As mentioned above, plant life in riparian zones may be diminished by saline waters and therefore provide less canopy to protect the water from sunlight. More light entering rivers causes a shift from heterotrophic to autotrophic communities (Millán *et al.*, 2011). Another consequence of less abundant riparian zones is higher nutrient flows into river systems, since plants in the riparian zone normally capture nutrients in runoff and groundwater (Dunlop, McGregor and Horrigan, 2005).

FIGURE 6.4 | Overview of salt impacts on aquatic ecosystems



Source: Dunlop, McGregor and Horrigan, 2005.

6.3.2 Economic activities

Salinization entails economic consequences, since ecosystem services such as provisioning of food and regulation of water quality are impaired. If fish populations decrease or change as a result of water salinization, for example, incomes and food security of fishers may suffer.

High salt concentrations prevent the uptake of water by plants causing reductions in crop yields. Salts accumulate in the root zone to such an extent that, as a result of increased osmotic pressure, the crop is no longer able to extract sufficient water from the salty soil solution. If water uptake is appreciably reduced, the plants rate of growth slows, with symptoms that resemble those of drought. In the early stages, soil salinization reduces plant productivity, but in advanced stages it kills all vegetation and transforms fertile and productive land to barren land. With this in mind, the Food and Agriculture Organization recommends limits to the use of saline water for irrigation (Ayers and Westcot, 1994). Restrictions on use for irrigation start at a concentration of 450 mg/l TDS, a concentration that is not unusual downstream of irrigation areas in semiarid regions.

Good livestock production also requires water of sufficient quality. The effect of water salinity on livestock health and productivity depends on many factors, including the species, breed and age of the animals drinking the water, the water and mineral content of the animals' feed, the temperature of the climate and the water, and which minerals are present in the water (Curran and Robson, 2007). Different species differ in their tolerance of drinking water salinity. While poultry and beef cattle are more sensitive, pigs can tolerate more saline water. When they first encounter saline water, animals may initially be reluctant to drink and may show symptoms of diarrhoea. When water salinity is too high, loss of production and decline in animal health should be expected.

Calculation of the economic impact of salinization of land and freshwater bodies remains under-researched in many parts of the world. A review of previous studies shows a very limited number of highly variable estimates of the costs of salt-induced land degradation (Qadir *et al.*, 2014). This review suggests that the global annual cost of salt-induced land degradation in irrigated areas could be US$ 27.3 billion because of lost crop production. No such global estimate exists for the economic impacts of freshwater salinization, with only a few scattered studies. For example, in the Border Rivers catchment in Australia, Wilson *et al.* (2004) estimated the costs of water salinity associated with infrastructure damage to be almost $700 000 per year.

6.3.3 Human health

Human health may be affected by salinized drinking water. The maximum allowable intake of sodium is 2 g per day, equivalent to 5 g salt per day (WHO, 2012). For chloride in drinking water the limit is 250 mg per litre (WHO, 2003). The most common health issue related to saline water is high blood pressure (hypertension), which may lead to higher risks of heart diseases and stroke. Other adverse health effects include skin diseases, miscarriages, diarrhoea and acute respiratory infection (World Bank, 2013).
Global exposure to salinized drinking water and the global implications for human health have not been comprehensively assessed. Nevertheless, the effects are well documented locally, such as in the coastal areas of Bangladesh, where sea-water intrusion, poor water management and shrimp farming have caused the salinization of ponds, rivers and tube-wells used for obtaining drinking water. Significant associations were seen between salinity increases in drinking water and the incidence of both pre-eclampsia and gestational hypertension (Khan *et al.*, 2011; Khan *et al.*, 2014).

This chapter has examined how agriculture mobilizes and transports large amounts of salts every year to receiving water bodies with potential severe effects on ecosystems and human health. Impacts are potentially stronger in arid and semiarid areas, where soil salinity is more frequent and where receiving water bodies have less dilution capacity. The agriculture-induced intrusion of saline groundwater or seawater to freshwaters adds to the problem and requires increasing attention as the remediation of salinized aquifers can be a very costly and a long-term endeavour, if possible at all.

6.4. References

Abrahao, R., Causapé, J., García-Garizábal, I. & Merchán, D. 2011. Implementing irrigation: salt and nitrate exported from the Lerma basin (Spain). *Agricultural Water Management*, 102:105-112.

Anning, D.W. & Flynn, M.E. 2014. Dissolved-solids sources, loads, yields, and concentrations in streams of the conterminous United States [online]. *US Geological Survey Scientific Investigations Report 2014-5012*, 101 pp. US Department of the Interior.

Ayers, R.S. & Westcot, D.W. 1994. *Water quality for agriculture*. FAO Irrigation and Drainage Paper, 29 Rev.1. Rome, FAO.

Barros, R., Isidoro, D. & Aragüés, R. 2012. Three study decades on irrigation performance and salt concentrations and loads in the irrigation return flows of La Violada irrigation district (Spain). *Agriculture, Ecosystems & Environment*, 151:44-52.

Causapé, J., Quílez, D. & Aragüés, R. 2006. Irrigation efficiency and quality of irrigation return flows in the Ebro River Basin: an overview. *Environmental Monitoring Assessment*, 117(1-3):451-461.

Cañedo-Argüelles, M., Kefford, B.J., Piscart, C., Prat, N., Schäfer, R.B. & Schulz, C.J. 2013. Salinisation of rivers: an urgent ecological issue. *Environmental pollution*, 173:157-167.

Cañedo-Argüelles, M., Hawkins, C.P., Kefford, B.J., Schäfer, R.B., Dyack, B.J., Brucet, S., Buchwalter, D. *et al.* 2016. Saving freshwater from salts. Ion-specific standards are needed to protect biodiversity. *Science*, 351(6276):914-916.

Crosa, G., Froebrich, J., Nikolayenko, V., Stefani, F., Galli, P., Calamari, D. 2006. Spatial and seasonal variations in the water quality of the Amu Darya River (Central Asia). *Water Research*, 40 (11):2237-2245.

Curran, G. & Robson, S. 2007. Water for livestock: interpreting water quality tests. *Prime Facts 2007*, 533(1):1-5.

Duncan, R.A., Bethune, M.G., Christen, E.W. & Hornbuckle, J. 2005. A review of salt mobilization and management in irrigated areas of the Murray-Darling Basin. *Cooperative Research Centre for Catchment Hydrology*. 05(1):48.

Duncan, R.A., Bethune, M.G., Thayalakumaran, T., Christen, E.W. & McMahon, T.A. 2008. Management of salt mobilisation in the irrigated landscape – a review of selected irrigation regions. *Journal of Hydrology*, 351(1-2):238-252.

Dunlop, J., McGregor, G. & Horrigan, N. 2005. *Potential impacts of salinity and turbidity in riverine ecosystems*. The State of Queensland, 72.

FAO. AQUASTAT (different years per country). [Cited 7 July 2016] http://www.fao.org/nr/water/aquastat/main/indexesp.stm

FAO. 2011. *The state of the world's land and water resources for food and agriculture (SOLAW) – managing systems at risk*. Rome, FAO and London, Earthscan.

Foster, S., Chilton, J., Nijsten, G.J. & Richts, A. 2013. Groundwater: a global focus on the 'local resource'. *Current Opinion in Environmental Sustainability*, 5:685–695.

Freeze, R.A. & Cherry, J.A. 1979. *Groundwater*. Englewood Cliffs, USA, Prentice-Hall.

Ghassemi, F., Jakeman, A. J. & Nix, H.A. 1995. *Salinisation of land and water resources: human causes, extent, management and case studies*. Cab International.

Herbert, E.R., Boon, P., Burgin, A.J., Neubauer, S.C., Franklin, R.B., Ardón, M., Hoppensperger, K.N., Lamers, L.P.M. & Gell, P. 2015. A global perspective on wetland salinization: ecological consequences of a growing threat to freshwater wetlands. *Ecosphere*, 6(10):1-43.

Herczeg, A.L., Dogramaci, S.S. & Leaney, F.W.J. 2001. Origin of dissolved salts in a large, semi-arid groundwater system: Murray Basin, Australia. *Marine and Freshwater Research*, 52(1):41-52.

IGRAC (International Groundwater Resources Assessment Centre). 2009. *Global overview of saline groundwater occurrence and genesis. Report number GP 2009-1*. Utrecht, the Netherlands.

Jafari, H., Raeisi, E., Hoehn, E. & Zare, M. 2012. Hydrochemical characteristics of irrigation return flow in semi-arid regions of Iran. *Hydrological Sciences Journal*, 57(1):173-185.

Khan, A.E., Ireson, A., Kovats, S., Mojumder, S.K., Khusru, A. *et al.* 2011. Drinking water salinity and maternal health in Coastal Bangladesh: implications of climate change. *Environmental Health Perspective*, 119:1328–1332.

Khan, A.E., Scheelbeek, P.F.D., Shilpi, A.B., Chan, Q., Mojumder, S.K., Rahman, A. *et al.* 2014 Salinity in drinking water and the risk of (pre)eclampsia and gestational hypertension in Coastal Bangladesh: a case-control study. *PLoS ONE*, 9(9): e108715

Klein Goldewijk, K. 2001, Estimating global land use change over the past 300 years: the HYDE database. *Global Biogeochemical Cycles*, 15:417 – 433.

Kolar, C.S. & Lodge, D.M. 2000. Freshwater nonindigenous species: interactions with other global changes. In: Mooney, H.A. and Hobbs, R.J. (eds) *Invasive species in a changing world*, pp.3-30.

Kozlowski, T. 1997. Responses of woody plants to flooding and salinity. *Tree Physiology Monograph*, 1:1–29.

Koç, C. 2008. The environmental effects of salinity load in Great Menderes Basin irrigation schemes. *Environmental Monitoring and Assessment*, 146:479-489.

Lorenz, J.J. 2014. A review of the effects of altered hydrology and salinity on vertebrate fauna and their habitats in northeastern Florida Bay. *Wetlands*, 34:189–200.

Mateo-Sagasta, J. & Burke, J. 2010. *State of Land and Water (SOLAW). Background report on water quality and agriculture interactions, a global overview*. Rome, FAO.

Meybeck, M. 2004. The global change of continental aquatic systems: dominant impacts of human activities., *Water Science and Technology*, 49(7):73-83. (also available at http://www.casinapioiv.va/content/dam/accademia/pdf/sv106/sv106-meybeck.pdf).

Millán, A., Velasco, J., Gutiérrez-Cánovas, C., Arribas, P., Picazo, F., Sánchez-Fernández, D. & Abellán, P. 2011. Mediterranean saline streams in southeast Spain: what do we know? *Journal of Arid Environments*, 75(12):1352-1359.

Morris, B L., Lawrence, A.R.L., Chilton, P.J.C., Adams, B., Calow R.C. & Klinck, B.A. 2003. *Groundwater and its susceptibility to degradation: a global assessment of the problem and options for management*. Early Warning and Assessment Report Series, RS. 03-3. Nairobi, United Nations Environment Programme.

New South Wales Government. 2016. *Dryland salinity: the basics*. [Cited 6 July 2016]. http://www.environment.nsw.gov.au/resources/salinity/Book1DrylandSalinity.pdf

Nielsen, D.L., Brock, M.A., Rees, G.N. & Baldwin, D.S. 2003. Effects of increasing salinity on freshwater ecosystems in Australia. *Australian Journal of Botany*, 51(6):655-665.

Pinder, A.M., Halse, S.A., McRae, J.M. & Shiel, R.J. 2004. Aquatic invertebrate assemblages of wetlands and rivers in the wheatbelt region of Western Australia. *Records of the Western Australian Museum Supplement*, 67(7):37.

Pinder, A.M., Halse, S.A., McRae, J.M. & Shiel, R.J. 2005. Occurrence of aquatic invertebrates of the wheatbelt region of Western Australia in relation to salinity. *Hydrobiologia*, 543:1–24.

Post, V. & Abarca, E. 2010. Preface: saltwater and freshwater interactions in coastal aquifers. *Hydrogeology Journal*, 18(1):1-4.

Qadir, M., Quillérou, E., Nangia, V., Murtaza, G., Singh, M., Thomas, R.J., Drechsel, P. & Noble, A.D. 2014. Economics of salt-induced land degradation and restoration. *Natural Resources Forum*, 38:282–295

Scanlon, B.R., Jolly, I., Sophocleous, M. & Zhang, L. 2007. Global impacts of conversions from natural to agricultural ecosystems on water resources: quantity versus quality. *Water Resources Research*, 43, W03437

Scherman, P.A., Muller, W.J. & Palmer, C.G. 2003. Links between ecotoxicology, biomonitoring and water chemistry in the integration of water quality into environmental flow assessments. *River Research and Applications*, 19(5-6):483-493.

Thompson. 2008. *The Aral Sea crisis*. [Cited 12 July 2016]. http://www.columbia.edu/~tmt2120/introduction.htm

Thouvenot, L., Haury, J. & Thiébaut, G. 2012. Responses of two invasive macrophyte species to salt. *Hydrobiologia*, 686(1):213-223.

UNEP (United Nations Environment Programme). 2016. *A snapshot of the world's water quality: towards a global assessment*. Nairobi.

van Hoorn, J.W. & van Alphen, J.G. 2006. Salinity control. In H.P. Ritzema, ed. *Drainage principles and applications*, p. 533-600. Wageningen, The Netherlands, International Institute for Land Reclamation and Improvement.

Walker, J., Bullen, F. & Williams, B. 1993. Ecohydrological changes in the Murray-Darling Basin. I. The number of trees cleared over two centuries. *Journal of Applied Ecology*, 30:265–273

WHO (World Health Organization). 2003. *Chloride in drinking-water: background document for development of WHO guidelines for drinking-water quality*. [Cited 14 July 2016]. http://www.who.int/water_sanitation_health/dwq/chloride.pdf

WHO. 2012. *Guideline: sodium intake for adults and children*. Geneva.

Williams, W.D. 2001. Anthropogenic salinisation of inland waters. In J.M. Melack, R. Jellison & D.B. Herbst (eds) *Saline lakes* pp. 329-337. Springer, the Netherlands.

Wilson S.M. 2004. *Dryland and urban salinity costs across the Murray-Darling Basin: an overview and guidelines for identifying and valuing the impacts*. Canberra, Murray-Darling Basin Commission.

W**orld Bank.** 2013. *Turn down the heat: climate extremes, regional impacts, and the case for resilience. A report for the World Bank by the Postdam Institute for Climate Impact Research and Climate Analytics*. Washington, DC.