*REVIEW*

# Salt: an emerging water concern for the global mining and minerals industries

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Water scarcity represents a sustainability challenge affecting economies, societies and ecosystems globally, and this challenge is increasingly exacerbated by salt, or total dissolved solids (TDS), loading in fresh waters. For the mining and minerals industries, there is a growing need to recognise TDS as a potential concern, and to strategize and innovate solutions. This paper examines the mechanisms by which these industries may contribute to TDS loading, the possible implications for environmental and socioeconomic well being, as well as those for industry itself, and the control and treatment options that currently exist. Finally, several case studies of management approaches in different parts of the world are presented in order to highlight the importance of site specific parameters, and the global commonality of the issue.

**Keywords:** Total dissolved solids, Mine water treatment, Acid mine drainage, Mining, Minerals

## Introduction

Sustainable use of natural resources is perhaps the most critical challenge faced by the global population today. Within this scope exists a multitude of interconnected and often competing problems, including many related to the dichotomy between environmental conservation and economic and/or social development. The mining and minerals industries hold a unique position in the natural resources challenge, as both a primary producer and consumer. Indeed, these industries provide the raw materials needed to develop and sustain healthy economies and qualities of life; but to do so require tremendous amounts of other resources – not the least of which is water.

On a unit basis, the socioeconomic benefits derived from water usage by the mining and minerals industries are comparatively large. In 2009 in Australia, for example, the coal mining industry added a gross value (The gross value added refers to the total production of goods and services less the costs of all inputs and raw materials necessary for production.) of AUD 298 million per gigalitre of water, while the agricultural industry added AUD$4 million per gigalitre (ABS, 2011). However, related impacts to quality and quantity of water resources often cannot easily be measured in economic terms. During mining and minerals processing, water supplies can be degraded or reduced through productive use, interference with the hydrologic cycle, or alteration of the chemical, physical and biological properties of the surrounding ecosystem (Hartman *et al.*, 2005). Although acid drainage and metals contamination are the most commonly considered water quality impacts of the mining and minerals industries, salt loading is a quickly emerging concern.

Salt or, more accurately, total dissolved solids (TDS) loading to fresh water may have varying degrees of environmental and socioeconomic implications, as discussed below. These implications depend on site specific factors such as background TDS levels, sensitivity of biota, and relative availability of water. In natural waters, TDS is primarily constituted by bicarbonate ($HCO_3^-$), sulphate ($SO_4^{2-}$), chloride ($Cl^-$), calcium ($Ca^{2+}$), magnesium ($Mg^{2+}$), sodium ($Na^+$) and potassium ($K^+$), with various other ions (e.g. metal cations, oxy anions) as minor contributors (ALPSDP, 2009). Because it is a measure of the total ionic content, TDS is often used as a parameter for the comparison of different bodies of water, though this is not necessarily reasonable (ABS, 2011). The presence and concentration of specific constituents depend on both natural and anthropogenic parameters (e.g. regional geochemistry or degree of urbanisation) (Banks *et al.*, 1997), resulting in ionic compositions that have variable ramifications for the environment, and for domestic or industrial water use.

A related water quality parameter is specific conductance (SC), which is a measure of water's ability to conduct an electric current (Stumm and Morgan, 1996). Since increased ion concentrations promote conductance, higher TDS is generally associated with higher SC, although the relationship between these two parameters varies substantially depending on the particular ionic composition. It also happens that SC is much simpler to measure than TDS (i.e. via a basic working and reference electrode system, as opposed to iterative evaporations), so SC is usually preferable for monitoring programs requiring frequent data collection. For these reasons, TDS and SC are often used somewhat interchangeably, but their environmental implications are not necessarily equivalent. For example, it is well

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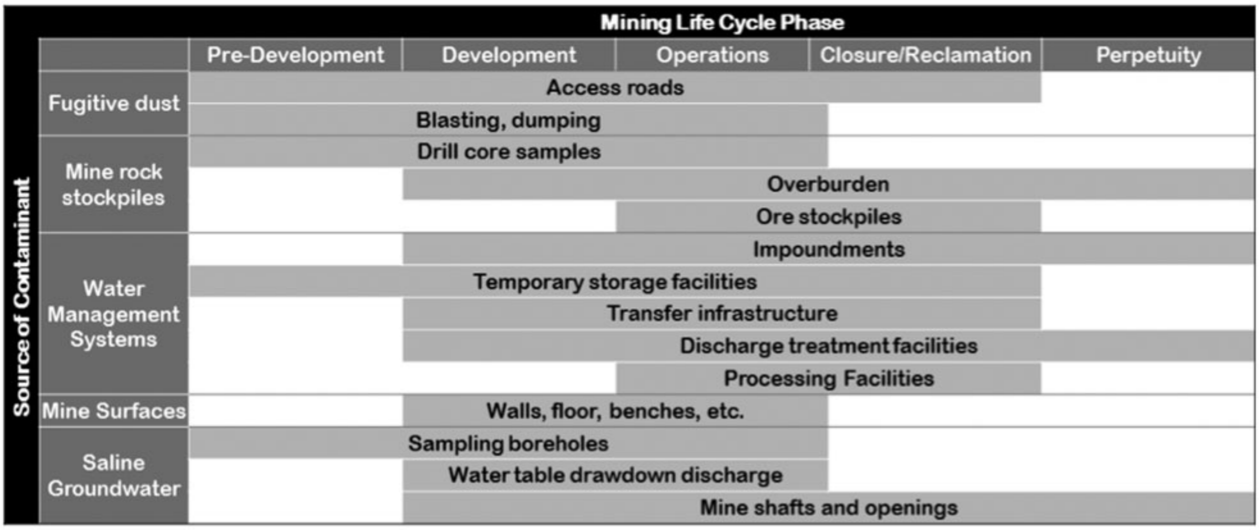



**1 Mining related sources of TDS loading by life cycle phase**

accepted that particular ions have very different characters in terms of impacts on biologic function and overall aquatic ecosystem health (e.g. Mount *et al.*, 1997).

For the mining and minerals industries, TDS is especially problematic for several reasons. First and foremost, the potential for TDS generation exists across all sectors of the industry (e.g. hardrock, coal, industrial minerals) and all phases of the mining life cycle (Fig. 1) (Tetra Tech, 2009; USEPA, 2011). Additionally, a considerable number of site specific factors confound prediction of the effects of TDS loadings (Livingstone, 1963; Timpano, 2011; MCMPR, 2006). And finally, if and when significant impacts may occur, few economically feasible and/or environmentally or socially acceptable mitigation methods are currently available.

This paper reviews the mechanisms by which industry may contribute to TDS loading to fresh waters, the potential implications for environmental and socio-economic well being, and also those for the industry, and the range of control and treatment strategies that might be utilised. Additionally, several TDS management strategies are presented as case studies.

## Mining related sources of TDS loading

During each stage of the mining cycle, from exploration and into perpetuity, mining and mining related activities have the potential to alter the natural TDS content of surrounding waters (Fig. 1), and each operation has a unique set of circumstances.

Natural water chemistry is significantly related to the mineral content of rock and soil materials with which the water interacts. Dissolution of these minerals, and thus TDS loading to water, is promoted by their oxidation upon exposure to water or air (Griffith *et al.*, 2012). Since mining and minerals operations are fundamentally dependent upon materials breakage and transport, increased access to mineral surfaces is inherent. Perhaps the most common example of mining related TDS loading is that of pyrite induced acid generation followed by natural or engineered neutralisation. (Pyritic minerals are those containing iron sulfides (e.g. $FeS_2$). When these or other metal sulphides are oxidised in the presence of water, acid is generated (e.g. by $2FeS_{2(s)}+7O_2+2H_2O\rightarrow 2Fe^{2+}{}_{(aq)}+4H^+{}_{(aq)}+4SO_4{}^{2-}{}_{(aq)}$) and TDS is increased by dissolved mineral content. Subsequent reactions of the acid, such as neutralisation by carbonaceous minerals (e.g. $2CaCO_{3(S)}+4H^+{}_{(aq)}\rightarrow 2Ca^{2+}{}_{(aq)}+2H^+{}_{(aq)}+2CO_{2(aq)}$) or dissolution of other minerals, may further add to TDS loading.) Loading of TDS due to dissolution reactions can impact both surface and subsurface waters.

Mining activities may also promote mixing of high TDS and low TDS waters. Mine dewatering, for instance, is often necessary for both surface and underground operations, particularly in cases where a surface mine is flood prone or where activities are being conducted below the natural water table (Banks *et al.*, 1997). For example, coal mining operations in East Kent in the United Kingdom pumped saline ground water from the mines out onto the surface and in surface waters from 1906 until 1974, which led to contamination of fresh groundwater found in the Chalk aquifer. Originally, the aquifer had a conductivity of approximately 400 μS and TDS content of 450 mg $L^{-1}$; after being contaminated, the conductivity and TDS were raised to 7950 μS and 5240 mg $L^{-1}$, respectively (Headworth *et al.*, 1980). In addition to dewatering, mining activities often withdraw water to provide supplies for a variety of auxiliary operations (e.g. irrigation for reclamation activities, minerals processing, etc.), which may result in TDS loading to that water, as well as receiving waters upon discharge or other return (i.e. infiltration or runoff) to the hydrologic system.

## Implications of TDS loadings

Mining activities' impacts on the surrounding hydrologic system have been documented for centuries (Agricola, 1950). However, with the exception of isolated cases such as the aforementioned Chalk aquifer, the implications of TDS loading have only recently become a broad topic of interest. Since altering TDS content of a fresh water supply might have far reaching impacts, emerging interest in this topic is generally well founded, but whether this issue should present universal cause for concern is not yet clear.

## Environment

It is undeniable that mining and mining related activities interfere with the aquatic geochemistry of an ecosystem. Often, these interferences alter the concentration of TDS and the ratios of constituent species. However, since TDS has no defined composition, the impacts on water quality cannot be broadly defined.

Many studies have attempted to clarify the relationship between water quality and TDS by examining the effect of various concentrations and compositions of TDS on individual organisms. Overall, findings show that toxicity may be dependent on a range factors, including the concentrations of specific ions or combinations of ions (Mount *et al.*, 1997; Goodfellow *et al.*, 2000; Weber-Scannell and Duffy, 2007). For example, chloride, sulphate and potassium have been shown to be toxic to daphnids, but their toxicity decreases when more than one cation is present (Mount *et al.*, 1997). In other cases, high TDS concentrations and specific conductivity, in general, have been reported to affect the osmoregulation of aquatic species, causing cellular level ionic and water balance problems that harm growth and potentially increase mortality (Hart *et al.*, 1991). Moreover, the particular life cycle phase of an organism (e.g. development or reproductive stages) may determine its reaction to TDS loading. For example, when exposed to elevated levels of TDS during fertilisation, salmon embryos have been shown to experience reduced and delayed hatch rate, growth and developmental effects, and increased mortality rate. When the embryos were exposed after fertilisation, though, there were no detrimental effects (Stekoll, 2003).

If TDS loading is shown to harm an individual organism, there are potential implications for an ecosystem's overall function, but clarifying and quantifying these larger scale impacts is quite difficult. A negative correlation between levels of TDS and aquatic diversity or density has been shown for many situations, such as for the naturally saline lakes in Canada (Bierhuizen and Prepas, 1985; Derry *et al.*, 2003). For mining impacted streams that have elevated TDS, several studies have also shown impacts to aquatic biota (e.g. Hartman *et al.*, 2005). Typically, though, these streams are suffering from other pollutants and disturbances as well, and studies have additionally found that the cumulative nature of disturbances to a watershed or ecosystem makes it difficult to isolate the specific impact of TDS loading on water quality (Timpano, 2011).

Given the complexity of site specific factors that may be involved, quantification of the strain that TDS loadings may place on aquatic ecosystems is difficult at best. Therefore, determining when and to what degree TDS loadings should be managed is also quite difficult. Considering the costs and energy inputs associated with many mitigation strategies (see below), the goal should be to accurately assess the cause of water impacts, and then act accordingly. This approach may avoid unnecessary use of some resources, while ensuring protection of waters in legitimate jeopardy. In all cases, simple and low cost strategies to prevent water impacts (e.g. source control) should be considered as a top priority.

## Socioeconomics

For human consumption and use in agriculture and industry, high levels of TDS generally decrease the quality of clean water and increase the costs associated with using it. As a result, each major water use sector faces increasing direct costs associated with TDS loading, with domestic users paying the largest portion of these cost increases. For example, a study conducted in the Middle Vaal River area of South Africa (SA) found that if the average local TDS content increased by 100 mg $L^{-1}$, assuming a baseline of 500 mg $L^{-1}$, the household sector would bear $\sim$85% of the direct costs (Urban Econ, 2000). For the Middle Vaal River area, this cost would be ZAR26·6 million per year (in 1995 values of SA Rands), which would mean ZAR22·7 million per year for the household sector.

The primary reason that domestic users may bear significant costs of increased TDS levels is that drinking water quality requirements are high. For aesthetics (e.g. taste), the World Health Organization recommends 600 mg $L^{-1}$ as the threshold limit for TDS (WHO, 2011), and the United States has a secondary (non-enforceable) drinking water standard of 500 mg $L^{-1}$ (USEPA, 2002). The health effects of excessive TDS in drinking water are inconclusive, though sufficiently high levels could exceed the typical dietary limit for salt, leading to some predictable consequences. For example, a study conducted in Bangladesh found that during the dry season when drinking water contained on average 2600 mg $L^{-1}$ TDS, the rate of hypertension in pregnancy was 7% higher than during the monsoon season when drinking water contained 600 mg $L^{-1}$ TDS (Khan, 2011).

Well beyond the demand for drinking water, agriculture accounts for two-thirds of global freshwater withdrawals each year (UNEP, 2008). In order to meet that demand, extensive studies have been conducted on the use of high TDS water for crop irrigation and livestock watering. When using high TDS water for irrigation purposes, productivity is limited based on the tolerance of different crops for water stress or for an excess of a particular ion, although water stress can be mitigated by increasing the amount of irrigation water (Pasternak, 1987). In terms of use of high TDS water for livestock, site specific parameters strongly determine the extent to which TDS loading may limit productivity. For example, polioencephalomalacia, a neurological disorder (Higgins *et al.*, 2008), is a potential concern for cattle watered with high TDS water. However, the level of TDS that may lead to polioencephalomalacia depends on a variety of factors including whether the cattle are grass fed or confined (Patterson and Johnson, 2003), the type of pastureland used for grazing cattle (Johnson *et al.*, 2008), and the ionic composition of the water (USEPA, 1976). Moreover, many types of livestock are capable of adapting to higher levels of salinity, if given enough time, which allows for a greater degree of flexibility (Higgins *et al.*, 2008; Patterson and Johnson, 2003; Johnson *et al.*, 2008; USEPA, 1976).

Use of water with elevated TDS or SC can also significantly impact industrial processes, including those associated with mining or minerals processing. Damage to infrastructure or equipment may occur due to scaling and corrosion. Scaling occurs when calcium and magnesium salts accumulate on surfaces or in water pipes (Driscoll, 1986; Masters and Ela, 2008), and this problem is exacerbated by an increase in temperature (Masters and Ela, 2008; Funke, 1990). This especially limits the productivity of equipment used for heat exchange, such as cooling systems in deep underground mines (Funke, 1990). Elevated TDS can also increase corrosiveness of water, which is most often linked to the conductivity rather than ionic composition; water that contains levels of TDS greater than 1000 mg $L^{-1}$ has been shown to cause electrolytic corrosion, regardless of composition of TDS (Driscoll, 1986). Another problem with industrial use of high TDS waters is reduced efficiency of wet processing circuits. Many examples could certainly be identified specifically for minerals processing or refining operations (e.g. froth flotation, electrometallurgy), but this problem could affect any wet processing method; and it may be particularly noticeable in systems where water is continuously recycled, and thus allowed to concentrate dissolved species. The added water treatment and/or maintenance required to offset effects of high TDS water in industrial processes may result in substantial increases in overall production costs.

### Consequences for mining and minerals industries

While costs associated with intake and usage of high TDS waters may impact the bottom line for mining and minerals processing operations (Gunther and Mey, 2008), in many cases this may be a far lesser problem than production and discharge of such waters. Indeed there is growing pressure in some areas to limit TDS loadings (e.g. USEPA, 2011). In the Central Appalachian coalfields of the United States this topic has sparked real controversy, as many stakeholders do not agree on the root causes of aquatic impairments in this region, or the role of mining related TDS (Soraghan, 2011). Despite differences in scientific findings or opinions, what seems clear in this case is that TDS loading may become a critical business risk for mine operators, and one which is much more challenging to quantify than other unit operational costs. Although TDS discharges are not widely regulated at present, the potential for regulation and the overall sentiments of the public in a given region may well impact whether or not a mining project can progress. The unknowns, including time and costs, associated with impact assessments, future treatment requirements, and litigation must all be considered. In essence, gaining and maintaining the tangible and social licenses to operate could be more difficult, if not impossible, for projects which stand to significantly increase TDS loading to nearby water resources.

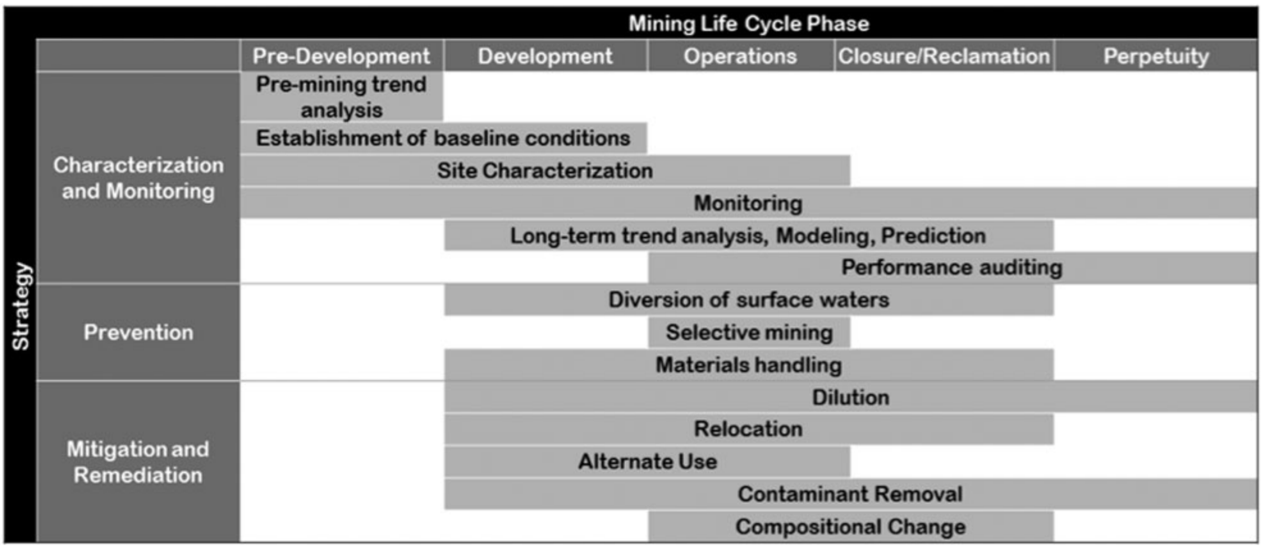
Interestingly, the mining and minerals industries appear well poised to tackle water use and degradation problems. Although concern over TDS loadings are currently nowhere near those over acid drainage or metals contamination, the concept of water risks is deeply ingrained in the way that these industries conduct business. In fact, a 2010 study of global corporate reporting on water risk showed that the mining industry included water related business risks as part of their annual financial reports more often than any other industry (Barton, 2010). The report found that among all industries, companies from the mining sector included the most detailed and water specific policies, standards, plans and management systems. Moreover, the mining industry had the best overall stakeholder engagement, including discussions with stakeholders about new or expanding projects and their potential impacts to water supply and quality. Given the industries' consciousness surrounding water impacts, as well as proven capabilities to problem solve and innovate, defining and managing issues associated with TDS loading should be considered a new, but not insurmountable, challenge.

## Management strategies

The mining and minerals industries must accept a tremendous amount of responsibility for the quantity and quality of fresh water in the regions where they operate. Management strategies, including those for TDS loading (Fig. 2), should therefore be considered across all phases of the mining life cycle.

### Site characterisation and monitoring

Site specific characteristics determine the best combination of methods for TDS management, which may need to evolve over the course of the mining life cycle. These characteristics are primarily determined by the spatial and temporal variability of TDS content in a given water




**2 Total dissolved solids management strategies by life cycle phase**

body, which is defined by the relationship between geologic, meteorological, hydrologic and anthropogenic processes that occur within a particular watershed (USEPA, 2011; Timpano, 2011). Establishment of the premining, baseline condition of a watershed gives a benchmark for comparison to help clarify the exact effects a project has on a watershed's structure and function with respect to TDS. Once the baseline has been established, continuous monitoring and data analysis enables the detection of problems and allows for performance evaluation and technique adjustment. Finally, designing the most cost efficient management strategies requires modelling and predicting TDS dynamics, analysing possible risks and keeping abreast of possible changes in TDS regulation, all of which stem from effective data collection and management (Hirsch, 2006; ALPSDP, 2009).

### Prevention of TDS impacts

Site characterisation provides the necessary information for design and implementation of TDS management strategies aimed at preventing changes in the naturally occurring concentration and composition of TDS. One such strategy, selective mining, may be used to avoid exposing particularly problematic materials to oxygen and water, and therefore prevent TDS production. Alternatively, materials handling and water drainage systems involve a combination of careful spoil placement and water routing channels to keep water–rock interactions at a minimum.

For example, the Red Dog Mine in Alaska successfully manages the TDS content of discharge water using preventative strategies (Tetra Tech, 2009). By identifying reactive, TDS producing mine rocks and placing them in specific covered waste piles with engineered drainage, two things are accomplished: infiltration of atmospheric waters and consequent reaction with problematic waste is minimised, and impacted seepage waters are collected and treated. The mine also collects surface waters upstream of the mine site and directs them through the site using diversion channels and piping systems to avoid contact with spoil material.

### Management of impacted waters

If significant TDS generation cannot be avoided, mitigation strategies may be applied to reduce or eliminate harmful effects. The most prevalent strategy employed is to simply dilute impacted waters, which can be an effective given ample assimilative capacity of the receiving water body. Often, however, reliance on dilution methods can lead to long term impacts on the receiving water body when discharge water volumes are allowed to approach or exceed the receiving body's assimilative capacity. For example, after 67 years of saline mine water discharged to the Chalk aquifer of East Kent in the United Kingdom, the aquifer was effectively sterilised as a supply of potable water, and a pipeline was built to the sea where the discharge was thought to be less harmful (Headworth *et al.*, 1980).

Aside from dilution, another option for managing high TDS waters may be to find an alternative use, including irrigation of nearby agricultural land, dust suppression at the mine site, or make-up water at the mineral processing plant. For example, in 2003, Xstrata Coal's Ulan Coal Mine began releasing excess mine waters through their Bobadeen Irrigation Scheme, watering 242 hectares of pastureland in New South Wales, Australia (Ulan Coal, 2012). Of course, the suitability of the specific water quality for the alternative use must be carefully considered in both the near and long term.

In the most extreme case, TDS impacted waters can be desalinated using thermal or membrane based filtration combined with post-filtration brine treatment. Because these types of treatment facilities are very expensive to build and maintain, they are only considered when other options are not feasible; and often great care is taken to minimise the volumes of water that must be treated. Ironically, concentrated salts derived from the brine treatment can be sold as a byproduct for use as de-icing agents on roads; currently, TDS loading from highway runoff does not appear to be a widespread concern.

## Global context and site specificity

Clearly, management strategies must be designed according to site specific parameters and constraints, as highlighted by the several examples below.

### Poland

Following the end of the communist era in Poland in 1989, the quality of that nation's surface waters was severely degraded, in large part because of the aggressive mining techniques practiced until then. It was estimated that only 5·1% of rivers were suitable for drinking water (Pawlowski, 1990). By 1996, only 4% of rivers were suitable for drinking water, and the Vistula River, which represents an estimated 55% of the water resources in Poland, was so degraded by TDS loading that its waters were unable to serve either agriculture or industry (Ericsson and Hallmans, 1996). The main source of TDS was from coal mine drainage, which was (and largely still is) being pumped directly into either the Vistula or the Odra River.

The coal mines discharging to the Vistula and Odra are located in the Upper Silesian Coal Basin, which contains approximately 80% of coal reserves in Poland (CSO, 2011). Less than half of the mines that were operational at the time of the political transition in 1989 are currently operating; the others were abandoned after being rendered unprofitable due to the transition from a centrally planned economy to a market economy (Wolkersdorfer and Bowell, 2005). Mines that remained active were consequently at risk from the spill over of seepage waters from abandoned mines, and in 2001, a central state agency was created in order to maintain the dewatering process for abandoned mines (Janson *et al.*, 2009). By 2006, two-thirds of pumped mine waters were being discharged to either the Vistula or the Odra, over half of which had greater than 3000 mg $L^{-1}$ TDS (Razowska-Jaworek *et al.*, 2008).

The pollution of the Vistula and Odra have been a recognised problem since before 1989 (Wolkersdorfer and Bowell, 2005), and since the establishment of a new political system, a strong effort has been made to address the high TDS content. One such effort was the development of the Debiensko desalination plant, which in 1995 began treating wastewater for two of the mines discharging to the Vistula River (Wolkersdorfer and Bowell, 2005). At the time of commission, the estimate for the total investment cost was US$60 million.

However, because desalination is five to six times more expensive than legally discharging waters to the river, most mining companies have, understandably, chosen to continue discharging largely untreated water (Janson *et al.*, 2009). This is a result of the Polish environmental law, based on the 'polluter pays' principle, which requires payment of fees and penalties associated with the use of natural resources and the discharge of waste materials (NFEPWM, 2012). As a result, in order to legally discharge mine drainage water, a payment must be made that accounts for each kilogram of chloride and sulphate discharged.

As a member of the European Union and so operating under the European Union Water Framework Directive, Poland has improved and strengthened its water policies and regulations, and there has been evidence of water quality improvements in both the Vistula and Odra (Absalon, 2009). Nonetheless, mine water discharges are expected to continue causing significant TDS loading problems in large portions of Poland's river systems, since economic incentives are not present to encourage more effective management.

## United States

CONSOL Energy, one of the largest coal mining companies in the Central Appalachian region of the United States, is currently building a reverse osmosis (RO) treatment plant capable of treating the wastewater discharges from four of its mines (US v. CONSOL, 2011). Estimated to cost US$200 million, this treatment plant was built both to avoid litigation struggles with the United States Environmental Protection Agency (USEPA) and as a proactive measure in anticipation of changing permitting policies, or perhaps regulations, related to TDS loading (CONSOL, 2012).

In 2009, thousands of aquatic fauna died due to an algae bloom caused by TDS loading in one of the receiving streams for mine discharges from one CONSOL operation (Soraghan, 2011). Though responsibility for the fish kill was shared amongst numerous parties, including natural gas producers in the area, the role of the CONSOL operation in question was arguably the most heavily scrutinised. In 2011, the United States filed a complaint against the company for excess chloride discharges, TDS loading, and enhancement of the golden algae population that caused the fish kill (Soraghan, 2011); and in response, CONSOL agreed to sign a Consent Decree with the USEPA (CONSOL, 2012). The decree did not implicate CONSOL in the fish kill, but mandated the construction of the RO plant, including its specifications, and required payments to the United States and the state of West Virginia for permit violations. Representing the most expensive capital expenditure for environmental control facilities in the region, the RO plant will reduce CONSOL's liability for future TDS loading problems, which may be especially important if TDS becomes a regulated water quality parameter (CONSOL, 2012).

## Australia

The Hunter River Salinity Trading Scheme (HRSTS) was developed in an effort to allow coal mines and power utilities the ability to discharge into the Hunter River, while limiting increased TDS loading impacts to either agricultural developments or downstream water users (DECNSW, 2006). Each operation that wishes to discharge waters to the Hunter River buys discharge credits, and during periods of high flow, the operations take turns releasing waste waters. The volume of water allowed for release is based on the salinity of the receiving water, and in this way, the HRSTS is able to account for cumulative impacts to allow more flexibility for upstream users and better protection for downstream users.

The HRSTS provides the opportunity for a high degree of site and operational specificity. For example, the coal seam groundwater at Xstrata's Bulga Coal Complex is of high enough quality to allow for its use with minimal corrosion or scaling of longwall and development equipment. Along with other on-site water uses such as coal washing, the Bulga is able to minimise discharges through creative water management and reuse of mine water (Bulga Coal, 2012). Throughout the system, each firm may buy as many or as few credits as needed, and in the case of performance improvements, may sell excess credits. As opposed to directly controlling discharges through specific regulations, which provide no incentive to discharge less than the regulatory standard, this scheme provides awards for improving TDS management practices and sets the scene for technological advancement.

## South Africa

The eMalahleni Water Reclamation Plant (EWRP), a desalination facility in the Upper Olifants River Catchment of South Africa, was recently highlighted by the United Nations Framework Convention on Climate Change as part of their Momentum for Change Initiative (UNFCCC, 2011). A public–private partnership between local mining and processing companies and the town of eMalahleni, the EWRP is the product of over 10 years of research initiated by Anglo Coal in 1994. Through balancing the needs of the mining companies with those of the community, the EWRP improves conditions for all stakeholders, including other nearby mining operations that share the treatment facility. By effectively dealing with excess mine waters that would otherwise pose an environmental threat, the EWRP allows these operations to continue to responsibly dewater their mines for extraction of coal reserves. Sale of the clean water to the local community and nearby water users alleviates a portion of the treatment costs, increases the profitability of mining operations, and provides a significant source of drinking water to a particularly water stressed community. Moreover, the salts removed from the treated water are dried and sold as gypsum bricks to local builders.

In order to extract coal in this area, many mining operations deal with large volumes of water, amounts that exceed the mine's water balance capabilities. Anglo Coal, the company responsible for the initiation of EWRP, began mining a site with acidic, high TDS waters in 1994 (Gunther and Mey, 2008). Having initially chosen a water treatment technology appropriate only for treating acidic mine drainage, TDS induced corrosion and scaling damaged $\sim$70% of the water treatment plant's piping within the first several years of operation. For the following 10 years, a variety of active water treatment technologies appropriate for TDS management were tested from pilot to demonstration scale plants, eventually resulting in the approach currently employed at the EWRP. Over many years of continuous improvement, the EWRP has become a benchmark for best management practices in TDS loading. Indeed, this project was chosen for the Momentum for Change initiative because it represents an ideal approach to public–private partnerships, and has provided a template that can be scaled and replicated to as the basis for future projects.

### Discussion of case studies

The four cases summarised above illustrate both the global commonality of mining-related TDS issues, and the diversity of surrounding circumstances and management approaches. The disparities between these cases, and others not included here, are the result of many site specific differences that dictate if, when, why and how TDS and other water quality issues are addressed:

(i) local and regional water needs, and relative scarcity
(ii) broad environmental policies and particular water regulations
(iii) public values, perception and influence (e.g. via social license)
(iv) corporate governance and responsibility
(v) technical parameters and constraints.

Although Australia and the United States share many similarities in terms of public environmental consciousness and corporate responsibility principles, differences in policy and the value of stakeholder cooperation are evident. The HRSTS case in Australia demonstrates how cooperation amongst a variety of stakeholders, including government, can promote creative and sustainable solutions. In South Africa, the EWRP example also highlights mutually beneficial outcomes for various parties, despite perhaps lesser regulatory oversight. In the United States, on the other hand, there is currently far less emphasis on cooperative strategies for holistic management of water resources, at least in the context of mining; and this translates to narrowly focused regulation, and thus narrowly focused compliance efforts. In Poland, long term TDS effects on water quality, the enormous scope of the continuing problems and technical challenges associated with mitigation, and a regulatory system that does not incentivise solutions all combine to present major challenges to further progress.

## Conclusions

Last century, demand for fresh water grew substantially. In conjunction with an expanding global population and the growth of many industrial sectors to support this population, water demands are expected to continue on an upward trend (UNEP, 2008). At the same time, readily usable supplies will shrink as more surface waters become heavily polluted and groundwater aquifers are depleted more rapidly than recharged. It is the combination of these factors that has increased global awareness of the true value of fresh water and the value of the ecosystem services provided by this precious resource.

Loading of TDS represents an emerging, perhaps unseeming, concern for fresh water supplies in many areas of the world. For the mining and minerals industries, this concern is particularly important, not because these industries are the sole contributors, but because their inputs can be somewhat ubiquitous, easily identifiable and significant, at least at local scales. The degree to which any impacts of mining are managed is ultimately determined by the balance between societal demands for mineral and energy resources, environmental protection, and human well being. At present, the mining and minerals industries have an opportunity to proactively investigate the impacts of their TDS loadings, which are most certainly site specific and dynamic, and to develop practical solutions that satisfy competing demands. In doing so, these industries stand to be at the forefront of technological advancements that potentially benefit all sectors of the global economy.

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## References

Absalon, D. 2009. Changes in water quality in the Upper Wisla (Vistula) River Basin, 2009. Proc. 2nd Int. Conf. on Environmental and computer science, Dubai, UAE, December 2009, IEE Computer Society, 29–32.

Agricola, G., Hoover, H. C. and Hoover, L. H. 1950. De re metallica, New York, Dover Publications.

ALPSDP. 2009. Australia Leading Practice Sustainable Development Program (ALPSDP) for the mining industry, evaluating performance: monitoring and auditing, Department of Resources, Energy, and Tourism, Canberra, Australia.

Australian Bureau of Statistics (ABS). 2011. Water Account Australia 2009–2010, cat. no. 4610·0, http://www.abs.gov.au/ausstats/abs@.nsf/mf/4610·0 (accessed 1 January 2012).

Banks, D., Younger, P. L., Arnesen, R., Iversen, E. R. and Banks, S. B. 1997. Mine-water chemistry: the good, the bad and the ugly, *Environ. Geol.*, **32**, (3), 157–174.

Barton, B. 2010. Murky waters? Corporate reporting on water risk: a benchmarking study of 100 companies. A Ceres report.

Bierhuizen, J. F. H. and Prepas, E. E. 1985. Relationship between nutrients, dominant ions, and phytoplankton standing crop in prairie saline lakes, *Can. J. Fisher. Aquat. Sci.*, **42**, (10), 1588–1594.

Bulga Coal. 2012. Bulga Complex Water Management, Xtrata Coal, http://www.bulgacoal.com.au/EN/environment/Pages/WaterManagement.aspx (accessed 1 January 2012).

Central Statistical Office (CSO). 2011. Environment 2011, statistical information and elaborations, Statistical Publishing Establishment.

CONSOL Energy Inc. 2012. Annual 10-K report 2011, Thomson Reuters, 10 February 2012.

Department of Environment and Conservation New South Wales (DECNSW). 2006. Hunter River Salinity Trading Scheme: Working together to protect river quality and sustain economic development, Department of Environment and Conservation NSW, Sydney, Australia.

Derry, A. M., Prepas, E. E. and Hebert, P. D. N. 2003. A comparison of zooplankton communities in saline lakewater with variable anion composition, *Hydrobiologia*, **505**, 199–215.

Driscoll, F. G., 1986. Groundwater and wells, 2nd edn, St Paul, Johnson Division.

Ericsson, B. and Hallmans, B. 1996. Treatment of saline wastewater for zero discharge at the Debiensko coal mines in Poland, *Desalination*, **105**, 115–123.

Funke, J. W. 1990. The water requirements and pollution potential of South African gold and uranium mines, South African Water Research Commission report no. KV 9/90.

Goodfellow, W. L., Ausley, L. W., Burton, D. T., Denton, D. L., Dorn, P. B., Grothe, D. R., Heber, M. A., Norberg-King, T. J. and Rodgers, J. H. 2000. Major ion toxicity in effluents: A review with permitting recommendations, *Environ.Toxicol. Chem.*, **19**, (1), 175–182.

Griffith, M. B., Norton, S. B., Alexander, L. C., Pollard, A. I. and LeDuc, S. D. 2012. The effects of mountaintop mines and valley fills on the physiochemical quality of stream ecosystems in the central Appalachians: a review, *Sci. Total Environ.*, **417–418**, 1–12.

Gunther, P. and Mey, W. 2008. Selection of mine water treatment technologies for the eMalahleni (Witbank) Water Reclamation Project, Anglo Coal, Ingwe Collieries.

Hart, B. T., Bailey, P., Edwards, R., Hortle, K., James, K., McMahon, A., Meredith, C. and Swadling, K. 1991. A review of the salt sensitivity of the Australian fresh-water biota, *Hydrobiologia*, **210**, (1), 105–144.

Hartman, K. J., Kaller, M. D., Howell, J. W. and Sweka, J. A. 2005. How much do valley fills influence headwater streams? *Hydrobiologia*, **532**, (1), 91–102.

Headworth, H. G., Puri, S. and Rampling, B. H. 1980. Contamination of a Chalk aquifer by mine drainage at Tilmanstone, East Kent, U.K., *Q. J. Eng. Geol. Hydrogeol.*, **13**, (2), 105–117.

Higgins, S. F., Agouridis, C. T. and Gumbert, A. A. 2008. Drinking water quality guidelines for cattle, University of Kentucky Cooperative Extension Service ID-170.

Hirsch, R. M. 2006. US Geological Survey perspective on water-quality monitoring and assessment, *J. Environ. Monit.*, **8**, (5), 512–518.

Janson, E., Gyzl, G. and Banks, D. 2009. The Occurrence and Quality of Mine Water in the Upper Silesian Coal Basin, Poland, *Mine Water Environ.*, **28**, (3), 232–244.

Johnson, P. S., Patterson, H. H. and Haigh, R. 2008. Effects of sulfates in water on performance of steers grazing rangeland, *Proc. West. Sec. Am. Soc. Anim. Sci.*, **55**, 261–264.

Khan, A. E. 2011. Drinking water salinity and maternal health in coastal Bangladesh: Implications of climate change, *Environ. Health Perspect.*, **119**, (9), 1328–1332.

Livingstone, D. A. 1963. Chemical composition of rivers and lakes, In: Data of geochemistry, 6th edn; Washington, DC, USGS.

Masters, G. M. and Ela, W. 2008. Introduction to environmental engineering and science, 3rd edn; Upper Saddle River, Pearson Education.

Ministerial Council on Mineral and Petroleum Resources (MCMPR). 2006. Strategic water management in the minerals industry; Canberra, Union Offset Printers.

Mount, D. R., Gulley, D. D., Hockett, J. R., Garrison, T. D. and Evans, J. M. 1997. Statistical models to predict the toxicity of major ions to Ceriodaphnia dubia, Daphnia magna and Pimephales promelas (fathead minnows), *Environ. Toxicol. Chem.*, **16**, (10), 2009–2019.

National Fund for Environmental Protection and Water Management (NFEPWM). 2012. http://www.nfosigw.gov.pl/en/nfepwm/ (accessed 1 January 2012).

Pasternak, D. 1987. Salt tolerance and crop production: A comprehensive approach, *Ann. Rev. Phytopathol.*, **25**, (1), 271–291.

Patterson, T. and Johnson, P. 2003. Effects of water quality on beef cattle, Proc. Range Beef Cow Symp., Mitchell NE, USA, December 2003, USDA Cooperative Extension Service and the animal science departments at University of Wyoming, South Dakota State University, Colorado State University and University of Nebraska, Paper 63.

Pawlowski, L. 1990. Chemical threat to the environment in Poland, *Sci. Total Environ.*, **96**, (1–2), 1–21.

Razowska-Jaworek, L., Pluta, I. and Chmura, A. 2008. Usage of mine waters in the Upper Silesia in Poland. Mine water and the Environment Proceedings 10th IMWA Congress, June 2–5 2008, Carlovy Vary, Czech Republic. Rapantova, N. and Hrkal, Z. (Eds), pages 537–539.

Soraghan, M. 2011. Fish-Kill Mystery, EPA Scientist Points at Shale Drilling, New York Times, 12 October 2011.

Stekoll, M. S. 2003. Salmon as a bioassay model of effects of total dissolved solids. Final report for ASTF grant no. 98-1-012.

Stumm, W. and Morgan, J. J. 1996. Aquatic chemistry: chemical equilibria and rates in natural waters, 3rd edn; New York, Wiley.

Tetra Tech. 2009. Red Dog Mine Extension – Aqqaluk Project, Final supplemental environmental impact statement.

Timpano, A. J. 2011. Levels of dissolved solids associated with aquatic life effects in headwater streams of Virginia's central Appalachian coalfield region. Master's thesis, Virginia Polytechnic Institute and State University, Blacksburg, VA, USA.

US Environmental Protection Agency (EPA). 2002. Secondary drinking water standards, Clean Water Act of 1972, 33 USC § 1251 et seq.

US Environmental Protection Agency (USEPA). 2011. The effects of mountaintop mines and valley fills on aquatic ecosystems of the central appalachian coalfields. Office of Research and Development, National Center for Environmental Assessment, Washington, DC, USA. EPA/600/R-09/138F.

US Environmental Protection Agency (USEPA). 1976. Quality criteria for water; Office of Water and Hazardous Materials, Washington, DC, USA.

Ulan Coal. 2003. Bobadeen irrigation scheme, Xstrata Coal http://www.ulancoal.com.au/EN/Biodiversity/Pages/BobadeenIrrigationScheme.aspx (accessed 1 January 2012).

United Nations Environment Programme (UNEP). 2008. Vital water graphics – an overview of the State of the World's Fresh and Marine Waters, 2nd edn; Nairobi, UNEP.

United Nations Framework Convention on Climate Change (UNFCCC). 2011. Launch of the Momentum for Change Initiative Launch Report; United Nations Climate Change Secretariat, Durban, South Africa.

United States of America, State of West Virginia, West Virginia Department of Environmental Protection v. CONSOL Energy Inc. 2011. Consolidation Coal Company, and Windsor Coal Company, Consent Decree, In the United States District Court for the Northern District of West Virginia Clarksburg Division.

Urban Econ. 2000. The economic cost effects of salinity, Vol. 1, Integrated report, South African Water Research Commission report no. TT 123/00.

Weber-Scannell, P. K. and Duffy, L. K. 2007. Effects of total dissolved solids on aquatic organisms: a review of literature and recommendation for salmonid species, *Am. J. Environ. Sci.*, **3**, (1), 1–6.

Wolkersdorfer, C. and Bowell, R. 2005. Contemporary reviews of mine water studies in Europe, Part 2, *Mine Water Environ.*, **24**, (1), 2–37.

World Health Organization (WHO). 2011. Guidelines for drinking-water Quality, 4th edn; Malta, Gutenberg.