Salton Sea Species Conservation Habitat Project

Selenium Treatment Technologies

Draft

October 2010

Introduction

1.1 PURPOSE AND NEED

The purpose of this report is to update information from the PEIR concerning the available technologies for treating waters containing selenium. Water treatment will be considered as part of a broader range of strategies as part of an overall selenium management plan for the SCH Project.

The Salton Sea Species Conservation Habitat Project (SCH Project), proposed by the California Department of Water Resources (DWR) and California Department of Fish and Game (DFG), will create approximately 2,400 acres of shallow ponds at the edge of the Salton Sea. The ponds will be designed to provide appropriate foraging habitat for piscivorous (fish-eating) bird species that depend on the Salton Sea. Selenium is present in the water sources for the SCH Project ponds, as well as the sediments of proposed pond areas. Exposure to elevated concentrations of selenium has the potential to adversely affect aquatic organisms and those wildlife that forage on aquatic biota, principally fish and birds.

SCH Project managersare developing management strategies to reduce selenium exposure and risks to ecological receptors. This involves identifying potential ecological receptors, characterizing sources and concentrations of selenium in water, understanding pathways to receptors, estimating potential ecorisk, identifying mitigation and source control strategies to reduce exposure, and developing treatment strategies if mitigation and source control strategies are not applicable. In 2005, the Salton Sea Ecosystem Restoration Program reviewed technologies and management techniques to limit selenium exposure (DWR 2005) as part of the Draft Programmatic Environmental Impact Report (PEIR; DWR and DFG 2006).

1.2 APPROACH

This report reviews existing physical (engineering), biological, and chemical technologies for selenium removal from water, indicates if these technologies are proven effective at high-volume applications, estimates their costs, and evaluates their applicability for use at the Salton Sea SCH Project. Methods for managing selenium in soils and sediments are not addressed in this review. The 2005 report relied on previous studies of treatment options for selenium removal and disposal from agricultural drainwater in the San Joaquin Valley, California (Frankenberger et al. 2004 cited in DWR 2005) and other reviews (Frankenberger and Benson 1994, Frankenberger and Engberg 1998 cited in DWR 2005). For this update, we consulted recent reviews of treatment options including Higashi et al. (2005), Alberta Environment (2006), Nitrogen and Selenium Management Program (NSMP 2007, 2008), Gusek et al. (2008), Alcoa (2009), and CH2M HILL (2010).

In addition, a science panel was convened in June 2010 to provide updated information at the smaller scale of the SCH Project. The panel consisted of scientists with expertise in selenium environmental toxicology, geochemistry, treatment, and Salton Sea issues[1]. Panelists were asked to provide information on the current state of the science and engineering in selenium treatment technologies, and to provide

[1] Panel members included Chris Amrhein (University of California [UC] Riverside), Doug Barnum (US Geological Survey [USGS] Salton Sea Science Office), Rick Gersberg (San Diego State University), Chris Holdren (US Bureau of Reclamation), Keith Miles (USGS), Harry Ohlendorf (CH2M HILL), Carol Roberts (US Fish and Wildlife Service [FWS]), Mike Saiki (USGS), Joe Skorupa (FWS), and Norman Terry (UC Berkeley).

recommendations for technologies that merit investigation for possible application to the SCH Project. They evaluated the technologies for effectiveness in treating low concentrations of selenium, feasibility, and cost. This report benefited from the panelists' input but is not intended to be a consensus report.

1.3 SELENIUM IN AQUATIC ENVIRONMENTS

The biogeochemistry of selenium in aquatic systems is complex and is controlled by several factors, particularly pH and redox conditions. Both the biotic and abiotic activity of selenium depends on its physiochemical form or species. Selenium chemistry resembles that of sulfur (Masscheleyn and Patrick, 1993). Selenium, like sulfur, can exist in four different oxidation states: selenide (Se -II), elemental selenium (Se 0), selenite (Se IV or SeO_3^{2-}), and selenate (Se VI or SeO_4^{2-}) (Robberecht and Van Grieken 1982). Alterations in oxidation state of selenium greatly affect solubility and play a major role in mobility, transport, fate, and effects of selenium species in wetland environments (Masscheleyn and Patrick 1993, Lemly 2002).

When dissolved selenium enters an ecosystem, it can be absorbed or ingested by organisms, bind or complex with particulate matter, or remain free in solution (Lemly 2002). Various biological, chemical, and physical processes can move selenium into or out of sediments; therefore, sediments may serve as only a temporary repository for selenium (Masscheleyn and Patrick 1993). Aquatic systems are dynamic, and selenium can be cycled back into biota even after waterborne inputs to the system have stopped (Lemly 2002).

Selenium biotransformation, bioaccumulation, and transfer through both sediment and water column foodwebs constitute major biogeochemical pathways in aquatic ecosystems (Masscheleyn and Patrick 1993, Fan et al. 2002, Louma and Presser 2009). Selenium can be removed from solution and sequestered in sediments through the natural processes of chemical and microbial reduction of the selenate form to the selenite form followed by adsorption onto clay and the organic matter, reaction with iron, chemical coprecipitation, or settling. After selenium enters the sediment, further chemical and microbial reduction may occur, resulting in insoluble organic, mineral, elemental, or adsorbed selenium (Lemly 2002). Microscopic planktonic organisms, such as bacteria, protozoa, phytoplankton, and zooplankton, are a major component of the particulate matter in the water column. The particulate matter, in turn, forms the basis for detrital materials that settle onto the sediment and become the food source for sediment organisms, such as benthic macroinvertebrates. In addition, waterborne selenite can be physically adsorbed onto the sediment particles, ingested, absorbed, and transformed by the sediment organisms. Sediment-bound selenite can be reduced to insoluble elemental selenium by anaerobic microbial activities. Elemental selenium can be reduced further to inorganic and organic selenides and/or reoxidized to selenite and selenate by microorganisms in the sediment and/or in the digestive tracts of sediment macroinvertebrates. Selenides can enter the foodchain via absorption into sediment organisms or be oxidized to selenite and selenate. Selenium of different oxidation states can be further biotransformed by sediment organisms and transferred up the food chain (Fan et al. 2002; Hamilton 2004). Some selenium forms may be volatilized to the atmosphere through microbial activity in the water and sediments or through direct release by aquatic plants (Lemly 2002, Lin and Terry 2003).

Speciation affects transformation from dissolved forms to living organisms (e.g., algae, microbes) and nonliving particulate material at the base of the food webs (Luoma and Presser 2009). Selenate in the water column is taken up only slowly, especially if competition with sulfate (SO_4^{2-}) is involved. Selenite and organo-selenides are much more reactive. When any form of selenium is taken up at the base of the food web by plants and microbes, it is converted to organo-selenide. With extended residence times in a system the result is a build-up of proportionately more organo-selenides and selenite as selenium is recycled through the base of food webs. In general, selenium concentrations in algae, microbes, sediments, or suspended particulates are 100–500 times higher than dissolved concentrations in selenatedominated environments such as streams and rivers: however when selenite or organo-selenide are proportionately more abundant, the ratio can be 100–10,000, such as in wetlands. This unidirectional build-up of potentially reactive forms is a key factor in the ecological risks posed by selenium, especially in environments where water residence times are extended, as seen in wetlands and estuaries, compared to rivers (Luoma and Presser 2009).

Over time, most of the selenium associated with plant and animal tissues is deposited as detritus and eventually incorporated into the sediments. High TOC (total organic carbon) can be an issue for selenium bioaccumulation in ponds because selenite readily sorbs onto suspended particulates and detritus, thus becoming more bioavailable to invertebrates.

1.4 REGULATORY STANDARDS AND TOXICITY THRESHOLDS

Designation of threshold levels of selenium in water and sediments that are considered to pose a potential toxicity risk to aquatic biota has varied (Amrhein and Smith 2010, Ohlendorf and Heinz in press). For surface waters in the Salton Sea Basin, the Colorado River Basin Regional Water Quality Control Board (CRBRWQCB) adopted the U.S. EPA numerical limits for selenium of 5 µg/L for chronic exposure and 20 µg/L for acute (one hour average) exposure (CRBRWQCB 2006). For sediment, the U.S. Dept. of Interior (1998) and Hamilton (2004) classified selenium concentrations between 1 – 4 µg/g (or mg/kg) as "elevated above background" or "level of concern" and concentrations >4 mg/kg as the "toxicity threshold." Lemly (2002) considered the effect of bioaccumulation within a food chain and recommended somewhat lower selenium thresholds of 2 µg/L of inorganic Se in water, 2 µg/g in sediments, 3 µg/g in food-chain organisms, and 4 µg/g in whole fish. For bird eggs, which may exhibit reduced hatching success or teratogenesis from selenium exposure, a conservative and widely reported toxicity reference value is 6 µg/g, but selenium sensitivity can vary widely among species (Ohlendorf and Heinz, in press). Evaluation of the effectiveness of water treatment techniques should therefore strive to achieve water concentrations of less than 5 µg/L and possibly less than 2 µg/L.

1.5 CURRENT CONDITIONS

Selenium has been measured in the water, sediment and biota around the Salton Sea region. The U.S. Bureau of Reclamation (Reclamation) has monitored seasonal water quality in the Salton Sea and its tributaries in 1999 and 2004-2009 (C. Holdren, Reclamation, unpublished data). Along the southern shoreline of the Salton Sea, the USGS conducted a baseline survey of water quality and biota in 29 agricultural drains and ponds operated by the Imperial Irrigation District (IID) for 2005-2009 (Saiki et al. 2010). Extensive sediment sampling was conducted in 2010 at prospective sites of the SCH adjacent to the mouths of Alamo River and New River (Amrhein and Smith 2010).

In addition, a 50-hectare complex of four interconnected shallow saline habitat ponds (SHP) was constructed in 2006 by Reclamation at the southeastern shore of the Salton Sea. Selenium concentrations in water, sediment and biota were monitored by USGS in 2006-2008 (Miles et al. 2009).

1.5.1 Water Quality

Most of the selenium entering the Salton Sea comes originally from the Colorado River as water used to irrigate agriculture in the Imperial Valley. The majority of this selenium becomes concentrated by agricultural usage and is discharged from subsurface tile drains into surface drains that flow into the Salton Sea either directly or via tributaries (New River and Alamo River) (Saiki et al. 2010). Table 1 summarizes recent measurements of selenium concentrations in water in the Salton Sea area.

Table 1 – Selenium Concentrations in Water

Location	Se concentration (µg/L)	Year(s)	Source
Salton Sea	1-2	1999, 2004-2008	Reclamation (unpub. data)
	1.9-3.2	2006-2008	Miles et al. 2009
Whitewater River	1.7-2.4	1999, 2004-2008	Reclamation (unpub. data)
Alamo River	5.1-5.8	1999, 2004-2008	Reclamation (unpub. data)
	5.2-7.0	2006-2008	Miles et al. 2009
New River	3.2-3.5	1999, 2004-2008	Reclamation (unpub. data)
New River Imperial wetlands Brawley wetlands	 2.7-5.4 2.2 - 4	2006-2007	Johnson et al. 2009
Saline Habitat Ponds Pond 1 Pond 2 Pond 3 Pond 4	 1.9-3.0 0.9-2.4 1.2-2.7 3.4-5.7	2006-2008	Miles et al. 2009
Agricultural drains into south Salton Sea	0.8 – 79.1	2005-2009	Saiki et al. 2010

The U.S. Bureau of Reclamation (Reclamation) has monitored seasonal water quality in the Salton Sea and its tributaries in 1999 and 2004-2009 (C. Holdren, Reclamation, unpublished data). Average water concentrations of total selenium vary depending on water body (Table 1). The Salton Sea has the lowest levels (1.33-1.47 micrograms per liter [µg/L]) because the deeper areas function as a sink for selenium (DWR and DFG 2006). Since 1999 selenium concentrations in water remain low (1-2 µg/L) in the Salton Sea, remain steady in the Alamo (5.1-5.8 µg/L) and New rivers (3.2-3.5 µg/L), and have decreased in the Whitewater River (1.7-2.4 µg/L). Other constituents in the three rivers, such as nutrients, have not decreased with TMDLs or changes in agricultural practices.

Selenium concentrations varied widely and were often higher in agricultural drains (Saiki et al. 2010). USGS measured total selenium in 29 drains or ponds operated by IID. Further detail was provided for seven sites, by measuring the different forms or species of selenium: particulate and dissolved selenium, including inorganic and organic fractions. Total selenium in unfiltered samples for all sites averaged 4.18 µg/L (range, 0.790–79.1 µg/L). Total selenium concentrations in water were directly correlated with salinity and inversely correlated with TSS concentrations. Dissolved selenium in unfiltered water samples from the seven intensively monitored drains ranged from 0.700 to 32.8 µg/L, with selenate as the major constituent. Selenium speciation from these seven sites yielded average percentages of selenium constituents as follows: 82% dissolved selenate, 9% dissolved selenite, 8% dissolved organic selenium, and 1% particulate selenium.

Selenium concentrations in the experimental SHP complex were measured in 2006-2008 (Miles et al. 2009). The ponds were flooded in 2006 with waters blended from the Alamo River (selenium range 5.2-7.0 µg/L) and the Salton Sea (selenium range 1.9-3.2 µg/L). Mean salinity varied among the ponds and over time: 4-24 ppt in Pond 1, 9-10 ppt in Pond 2, 30-70 ppt in Pond 3, and 150-175 in hypersaline Pond 4. The blended waters had a selenium concentration of less than 5 µg/L flowing into the ponds. Mean concentrations of selenium in water were 1.9-3.0 µg/L in Pond 1, 0.9-2.4 µg/L in Pond 2, 1.2-2.7 µg/L in Pond 3, and 3.4-5.7 µg/L in Pond 4. Selenium concentrations in water frequently exceeded the more conservative 2.0 µg/L toxicity threshold.

1.5.2 Selenium in Sediment

Selenium concentrations in sediment are greater in the north Salton Sea basin than the south basin (DWR and DFG 2006). Most of the selenium load comes from river discharges in the south and is circulated to the north, where the particulate selenium (adsorbed on sediment or bioaccumulated in algae) falls out of suspension. Backwater areas can be places with more settling out of suspended solids.

In 2010, extensive sediment sampling was conducted at the southern shoreline of the Salton Sea adjacent to the mouths of Alamo River and New River (Amrhein and Smith 2010). The majority of sediment samples (63%) would be considered “low risk” (< 1 mg/kg). The remaining 37% of the samples were in the “level of concern” category with concentrations between 1 and 4 mg/kg, with only two samples exceeding 2.5 mg/kg. No sample exceeded the “toxicity threshold” value of 4 mg/kg.

At the experimental SHP, mean selenium concentrations in sediment were 1.03-2.32 mg/kg in Pond 1, 0.94-1.61 mg/kg in Pond 2, 1.73-3.00 mg/kg in Pond 3, and 1.67-2.35 mg/kg in Pond 4. Selenium concentrations in water decreased over time in water. Concentrations in sediment increased in Ponds 1 and 2 and decreased in Pond 4. Sediment concentrations did not exceed the 4.0 mg/kg toxicity threshold after nearly three years of operation.

1.5.3 Selenium in Salton Sea Biota

Selenium concentrations have been measured in various biota at the Salton Sea area, including algae, vegetation, invertebrates, fish, and bird eggs (DWR and DFG 2006, Johnson et al. 2009, Miles et al. 2009, Saiki et al. 2010).

Recent studies at the experimental SHP measured selenium in invertebrates (biannually fall 2006-fall 2008) and black-necked stilt eggs (2006, 2007, 2008) (Miles et al. 2009). Fish (desert pupfish, mosquitofish and tilapia) and fish-eating birds were present but not tested. Selenium concentrations in black-necked stilt eggs collected from the experimental saline habitat ponds (2-8 mg/kg DW, mean 5 mg/kg DW) were significantly higher than reference sites for two out of the three years, and 47%of the eggs exceeded the 6.0 mg/kg dw selenium toxicity threshold (Miles et al. 2009). Anderson (2008) reported that selenium concentrations in stilt eggs in SHP ponds were elevated, but concentrations were similar to those found in other stilt nesting habitats in the Salton Sea. Selenium concentrations were highest in Pond 1. Stilts were tracked feeding in both ponds and the Salton Sea, however, and therefore the egg concentrations reflect a composite of prey from multiple sources and potentially different selenium levels.

Selenium concentrations in the agricultural drains biota were measured in the IID agricultural drains along the southern border of the Salton Sea (Saiki et al. 2010). Concentrations varied widely among drains and ponds, with one drain (Trifolium 18) exhibiting especially high concentrations in food chain matrices: particulate organic detritus 5.98–58.0 mg/kg dry weight (dw), midge larvae 12.7–50.6 dw, and fish (mosquitofish 13.2–20.2 mg/kg dw, sailfin mollies 12.8–30.4 mg/kg dw). Although selenium was accumulated by all trophic levels, biomagnification (defined as a progressive increase in selenium concentration from one trophic level to the next higher level) in midge larvae and fish occurred only at lower exposure concentrations. The health and wellbeing of poeciliids and pupfish did not appear to be threatened by ambient exposure to selenium in the drains and ponds (Saiki et al. 2010).

In the New River, the constructed Imperial and Brawley Wetlands were designed to reduce nutrients as well as selenium in the New River (Johnson et al. 2009). At Imperial, selenium concentrations in water were 2.7-5.4 µg/L in inflow and 2.0-4.8 µg/L in outflow. At Brawley wetlands, water concentrations were 2.2-4 µg/L in inflow and 1.1-2.0 µg/L in outflow. Concentrations were 1.5 - 8.2 mg/kg dw in invertebrates and 1.9 – 20.0 mg/kg dw in fishes.

1.5.4 Expected Future Conditions

Future scenarios as modeled in the Salton Sea Programmatic EIR (75 years) are not expected to exceed 10 µg/L in the New and Alamo rivers (DFG and DWR 2006).

Selenium Treatment Technologies

2.1 INTRODUCTION

This section provides an overview of treatment technologies for selenium removal and an assessment of their applicability for use in the SCH Project.

A variety of physical, chemical, and biological technologies have been applied to remove selenium from water. Information was obtained from literature and an expert workshop on the current state of the science and engineering in selenium treatment technologies. Each approach was evaluated for its effectiveness in removing selenium from water, in particular treating large volumes of water with low concentrations of selenium (less than 10 μg/L) in order to achieve selenium concentrations less than 5 μg/L in inflow water to the SCH ponds, based on the RWQCB Colorado River Basin Plan standard. Where possible, the assessment considered the potential for conversion of selenium to different, more bioavailable forms (selenite and organo-selenides) that pose greater risk to wildlife. Other considerations include whether it is a proven technology, feasibility of scaling up operations for full-scale testing, and costs.

The type, scale and cost of treatment will depend on the water demands for the SCH ponds. Initial estimates are that SCH pond complex will be approximately 2,400 acres, with a maximum depth of approximately 6 feet and a storage capacity of up to approximately 7,200 acre feet (AF) (2,346 million gallons). Evaporative losses will be approximately 14,400 AF (approximately 6 feet annually). The water filling the ponds will be a blend of river water and Salton Sea water or saline groundwater. The amount of water to be treated will depend on the salinity in the ponds, since only river water would need to be treated. Water required to keep up with evaporation losses in summer (July) would be approximately 32 cfs. An average diversion rate of 50 cfs would accommodate some flow through (outflow) as well as evaporation. This diversion rate is equivalent to approximately 32.3 million gallons per day [mgd] or 22,440 gallons per minute [gpm]).

Cost estimates were obtained principally from a 2010 review (CH2M HILL 2010) that looked at direct and indirect costs for installation, operation and maintenance for treatment per mgd, expressed in 2010 US Dollars (with a cost range of + 100% and – 50%). These values are approximate for cost comparisons among technologies, which are compared for treatment of 1 mgd (Table 2). Estimating costs for the SCH Project for all potential technologies is difficult because the scale of water treatment capacity would be approximately 30 times this level, and the costs would not be expected to increase linearly. These estimates are provided for relative comparison only.

2.2 PHYSICAL TREATMENT

Physical methods for selenium treatment include membrane filtration (reverse osmosis and nanofiltration) and ion exchange methods. In general, treatment systems using these processes are expensive, due to requirements for pre-treatment steps, utility costs, and post-treatment disposal of concentrated reject brines, which has limited their application (Alcoa 2009).

2.2.1 Reverse Osmosis (RO)

Reverse osmosis (RO) provides treatment by forcing a solution at high pressure through a membrane that retains salts (e.g. selenite and selenate) in the reject water. This technology can potentially provide a filtered permeate with a desirable selenium concentration (e.g. <5 µg/L), but it also produces a brine (concentrate) waste that requires disposal (CH2M HILL 2010). The concentrate waste is typically a small percentage of the overall volume of treated water (RO treating 100 gpm with a 90% efficiency will produce 90 gpm "treated water" and 10 gpm concentrate). Due to the high energy requirements of this system, costs associated with RO technologies are often prohibitive.

Effectiveness, Removal Efficiency, and Costs

- High pressure forces a solution through a membrane that retains salts (e.g. selenite and selenate) as the reject water.
- Treatment effectiveness for selenium removal has been reported to remove selenium to less than 5 µg/L. A 200 gpm RO system designed to treat a mining/ agricultural effluent was reported to decrease the inflow selenium concentrations of 12 to 22 µg/L to an outflow of 2 µg/L (Sobolewski 2005 reported in CH2M HILL 2010).
- Golder (2009) reported a RO treatment removal of selenium at a gold mine drainage system from 60 µg/L to < 5 µg/L.
- A pilot study using an RO for treatment of Shell refinery wastewater reported that the inflow concentration of selenium of 760 µg/L was decreased to 1 µg/L (U.S. Bureau of Reclamation 2008 reported in CH2M HILL 2010).
- A pilot study conducted for the Newport Bay Watershed concluded that an RO system would be feasible for removing selenium from water (NSMP 2008).
- Based on data from full scale systems operating to date RO technology is a feasible option to meet the treatment criteria proposed; however, due to the high energy demands of this system, it is likely cost prohibitive.
- The June panel agreed that although RO produces exceptional water quality, it is very expensive and the produced clean water would likely have greater demand for uses other than habitat restoration.
- Capital costs (year 2010) for treating 1 mgd with RO are approximately $40 million with annual O&M approximately $3 million (CH2M HILL 2010).

2.2.2 Nanofiltration

Nanofiltration is another treatment option that uses membrane filters to separate constituents from source waters. It is often referred to as a "loose" RO membrane, having larger pore size membranes and operating at roughly one-third the pressure of an RO system. Compared to RO, the nanofiltration process can potentially yield higher water recoveries and requires lower pressure and less pretreatment. These characteristics could make nanofiltration more cost-efficient than RO when comparing similar treatment applications.

Effectiveness, Removal Efficiency, and Costs

- In a laboratory study, nanofiltration was reported as removing more than 95% of selenium from agricultural drainage water with inflow concentrations ranging to 1,000 μg/L (Kharaka et al. 1996 reported in CH2M HILL 2010). Additionally, based on the preliminary laboratory findings, a pilot study was conducted that treated inflows ranging from 42 to 63 μg/L with outflows ranging from 1.0 to 3.2 μg/L.

- Pretreatment is a typical requirement for nanofiltration (e.g. removal of TSS and other constituents) to prevent fouling and/ or scaling of the membranes. Lifespans of membranes typically range from 2 to 3 years (CH2M HILL 2010).

- Nanofiltration has not been tested at full-scale for selenium removal. More data are needed for large-scale application.

- Like RO, nanofiltration is very expensive with high capital and O&M costs. In addition, the technology may not work for the SCH Project due to its size and scale.

- Capital costs (year 2010) for treating 1 mgd are similar to reverse osmosis: approximately $40 million with annual O&M approximately $3 million (CH2M HILL 2010).

2.2.3 Ion Exchange

Ion Exchange technology can be designed to remove or "exchange" ions targeted for removal and replaced with the other "desirable" ions using resins. Resins that have been reported to specifically remove selenium include both weak base and strong base anionic resins (Patterson 1985 and Twidwell et al. 1999 cited in CH2M HILL 2010).

Effectiveness, Removal Efficiency, and Costs

- There are limited data from full scale operations. Resin selection is a key design feature with consideration of pretreatment needs to increase the capacity and efficacy of resin.

- A laboratory scale test using process solutions from mining effluent reduced selenium from 930 μg/L to 1 μg/L with a silica polyamine resin (Golder 2009 in CH2M HILL 2010).

- In order to maintain resin efficacy, resins need to be regenerated when the ion exchange sites have been saturated. Regeneration for weak and strong base anion exchange resins typically need to be flushed with a sodium hydroxide solution. Additionally, this flushed effluent will contain a concentrated solution of selenium and will need to be treated (CH2M HILL 2010).

- Capital costs (year 2010) for treating 1 mgd are approximately $28 million with annual O&M approximately $4 million (CH2M HILL 2010).

2.2.4 Mechanical Evaporation - Vertical Tube Extraction (VTE)

Mechanical evaporation relies on heating water to produce a high quality distillate as well as a concentrated waste brine stream. This process can be very expensive if a waste energy source is not available. Fortunately, the region at the south end of the Salton Sea is seismically active and has several geothermal energy plants which could be a source of heat and energy.

Vertical Tube Extraction (VTE) is a distillation process that would use waste geothermal heat. The clean water produced would likely go to a higher use (e.g., domestic water supply). If water is blended with another source, however, this technology has potential for the SCH project. Advantages include a potential thermal refuge in winter and the ability to use waste heat to power the distillation process.

Effectiveness, Removal Efficiency, and Costs

- Mechanical evaporators in general are mechanically and thermodynamically complex systems that require regular maintenance. This has been implemented at full scale in industrial settings, but not specifically for selenium treatment (CH2M HILL 2010).

- Provides a high level of treatment with pure water distillate.

- Capital costs for mechanical evaporation technology treating 1 mgd are approximately $70 million with annual O&M approximately $8 million (CH2M HILL 2010).

- Information on VTE effectiveness and removal efficiency at full scale was unavailable. Operating costs could be reduced if VTE could take advantage of local geothermal waste heat.

2.3 CHEMICAL TREATMENT

2.3.1 Zero Valent Iron

Zero valent iron (ZVI; i.e. elemental iron) can be used as a chemical oxidation/ reduction process that reduces the oxidized forms of selenium (selenate and selenite) (CH2M HILL 2010). A number of different media can be utilized as the form of ZVI including powder, granular, or fibrous materials. Ultimately, the oxidized forms of selenium will co-precipitate out of solution as elemental selenium.

Effectiveness, Removal Efficiency, and Costs

- Iron co-precipitation methods used to treat mine runoff found selenium concentrations reduced from 100 µg/L to 12 to 22 µg/L (Sobolewski 2005 reported in CH2M HILL 2010).

- Pilot-scale studies over a 250-day period with an inflow of 5 to 14 µg/L did not consistently achieve selenium levels below 5 µg/L (Golder 2009 reported in CH2M HILL 2010).

- A pilot-scale study for flue gas desulfurization (FGD) wastewater at a coal fired powerplant treated an inflow concentration of 7,270 µg/L to an outflow concentration of 159 µg/L (EPRI 2009 reported in CH2M HILL 2010).

- Using this technology requires a control of the wastewater characteristics (i.e., pH, DO, temperature, competing oxyanions, etc.). Additionally, sufficient hydraulic retention times (up to 5 hours) and low pH (4) are necessary for the targeted reactions to occur (CH2M HILL 2010).

- There are considerable maintenance, replacement and disposal issues with the iron mass, which rusts together, becomes clogged within one year, and can fail hydraulically.

- Capital costs for a column-based system with steel wool are approximately $13 million with annual O&M approximately $3 million (CH2M HILL 2010).

2.3.2 Ferrous Hydroxide

Another chemical method for removing selenium from aqueous solutions involves the use of ferrous hydroxide. Under this patented process, ferrous hydroxide is generated by the addition of ferrous chloride or ferrous sulfate in alkaline water. The ferrous hydroxide solids generated under these conditions react to reduce selenium ions to elemental selenium. The selenium will then co-precipitate with the ferrous hydroxide solids. According to the literature, this reaction is optimized at pH 9.0, which limits its application in most aquatic systems.

Effectiveness, Removal Efficiency, and Costs

- Limited data showing selenium removal to less than 5 µg/L (CH2M HILL 2010).
- Residual sludge handling is required.
- Year 2000 capital costs for a 300 gpm FGD wastewater treatment plant was $15 million. Annual O&M costs ranged from $1.5 to $2 million (CH2M HILL 2010).

2.4 BIOLOGICAL TREATMENT

Treatment by biologically-based processes relies on bacterial activity that reduces selenate to selenite, then to elemental selenium (Alcoa 2009). The method causes the selenium to be converted to insoluble forms that can be captured or entrained by larger particles. Conversion of the selenium to a filterable form is also accompanied by conversion to volatile selenium compounds, typically including hydrogen selenide and methyl selenide. Such compounds can generally be eliminated from the discharge of the system through aeration (Frankenberger et al. 2004 cited in DWR and DFG 2005). The water to be treated by a biological process is normally spiked with a food source for the biomass, especially an assimilable carbon source. Free oxygen must be eliminated from the system so that the biological conversion proceeds in an anaerobic or anoxic state.

2.4.1 Anaerobic Bacterial

Anaerobic bacterial treatment chemically reduces selenium from selenate to selenite to less soluble elemental forms of selenium. This technique often uses engineered biological reactors that can include aboveground tanks or in-ground basins with suspended growth or attached growth systems (CH2M HILL 2010). This system includes a single or multi-stage reactor that encourages anoxic conditions. Suitable reactor types include fixed-bed reactors, fluidized-bed reactors, sludge-blanket reactors, and stirred reactors. The microbes use the oxygen in selenate, selenite or nigrate as an electron acceptor, and an organic carbon source (e.g. ethanol, methanol, molasses, corn syrup) as an energy source and electron donor. The use of these systems provides more control of flow rates and operating conditions.

One example of this technology is the Advanced Biological Metals Removal (“ABMet”) System, developed by Applied Biosciences and now marketed by GE Water and Process Technology (CHM HILL 2010). This bioreactor system uses an attached growth downflow filter, composed of a biofilm, or a layer of microorganisms grown on granular activated carbon beds. The ABMet system has been successfully applied in pilot-scale tests and full-scale implementation to treat mining (Alberta Environment 2006, Gusek et al. 2008) and agricultural effluent to achieve very low selenium levels (CH2M HILL 2010).

The capital costs of these systems are the primary limitation to their application, as well as O&M requirements such as flow equalization equipment, external carbon source, media cleaning and replacement, and sludge disposal.

Effectiveness, Removal Efficiency, and Costs

- A slow sand filter reactor was tested and reported effluents ranging from 15-30 µg/L with a 10-30 hour hydraulic retention time (Alberta Environment 2006).

- Mine water treated with an active bioreactor (flow ranging from 40-300 gpm) reported selenium inflow and outflow concentrations of 15 µg/L and <1 µg/L, respectively (Alberta Environment 2006).

- The ABMet system has been tested for removing selenium from water and is ready for full scale implementation (CH2M HILL 2010). This process consistently achieved final concentrations below 50 µg/L and often <2 µg/L (MSE 2001 reported in NSMP 2007).

- Total installed costs for ABMet system to treat 1 mgd are approximately $30 million with annual O&M approximately $3 million (CH2M HILL 2010). Another estimate ranges from $1.23-1.48 per 1,000 gal for a 10 mgd system (Alberta Environment 2006).

2.4.2 Algal Treatment

Selenium removal technologies using algae can include algal assimilation, algae-enhanced bacterial reduction, and algal volatilization. These are passive systems that require less operational support and little to no added resources (e.g. chemical or energy input) (CH2M HILL 2010).

Algal-Bacterial Selenium Removal (ABSR)

ABSR is an example of algal-enhanced bacterial reduction, a process in which naturally-occurring bacteria are stimulated by the addition of algae as a carbon source for the microbes (NSMP 2007). This system includes two ponds in series: a high rate pond to grow algae, and reduction ponds to provide an anaerobic environment for selenate reduction. The ABSR system requires high residence time and low flow rate, and potentially high land requirements for ponds. This process has been tested for selenium removal in agricultural drainage in the San Joaquin Valley (Panoche Drainage District) (Quinn et al. 2000). However, while the system did remove about 80%of the total selenium, over 30%of the selenium remaining in the system effluent consisted of selenite and organoselenium, which are more bioavailable and thus more harmful to wildlife (Stuart 2001 reported in NSMP 2007). Invertebrates exposed to treated water accumulated more selenium than those in untreated water (Amweg et al. 2003 reported in CH2M HILL 2010).

Controlled Eutrophication Process (CEP)

The Controlled Eutrophication Process (CEP) is an algal assimilation technology developed by Kent SeaTech, San Diego. The approach is to stimulate rapid algal growth in well-mixed ponds to take up nutrients and presumably selenium. The technology was origianally developed in concert with fish farming (tilapia). Algae are removed by gravity settling of algal sludge and fish waste. Carlberg et al. (2003) reported that fish tissue selenium concentrations were below detection, which led them to suggest that algae are not accumulating significant concentrations and that a CEP may not be an effective method for selenium treatment. However, this facility used Whitewater River water, which already has low selenium concentrations. The latest design concepts rely only on algal blooms, using larger algal species. CEP can operate at a wide range of salinity and has the added benefit of nutrient removal. Disadvantages include potential risks to wildlife that utilize ponds and unproven selenium removal efficacy. The process can be labor intensive and thereby costly.

Effectiveness, Removal Efficiency, and Costs

- Golder (2009) reported decreases in selenium using algal treatment in combination an *in situ* carbon treatment to promote bacterial growth. Using algae as an additional carbon source, selenium concentration was decreased from an inflow of 460 µg/L to an outflow of < 10 µg/L (Martin et al. 2009 reported in CH2M HILL 2010).

- Algal-bacterial selenium removal typically requires a relatively long hydraulic retention time. Golder (2009) reports a hydraulic retention time of 20-25 days in treatment ponds designed to reduce selenium via an algal-bacterial process.

- Algal treatments have not been demonstrated to reduce selenium levels to less than 5 µg/L in the effluent (NSMP 2007, CH2M HILL 2010). Furthermore, the effluent contains more bioavailable forms of selenium.

- Algal treatments are relatively low cost technologies (ABSR cost $.0008 per gallon; NSMP 2007 as reported in CH2M HILL 2010) compared to physical treatments. However, the large footprint of ponds can lead to high costs associated with land acquisition.

- A 2007 estimate for a CEP pilot treatment of a 9,000 gpm water supply (13 mgd) would require 150-200 acres of CEP ponds, with an estimated capital cost of $5-6 million and annual operating expenses of $1.6-1.8 million (pers. comm. J. Carlberg, Kent SeaTech to DWR, 2007).

2.4.3 Constructed Wetlands

Passive biological treatment using constructed wetlands removes selenium by reduction and depositing insoluble forms in sediments, accumulation in plant tissues, and volatilization to the atmosphere. These biological processes can be mediated by plants, plant/microbe associations, and/or microbes alone. Selenium can be removed from solution and sequestered in sediments through the natural processes of chemical and microbial reduction of the selenate form to the selenite form, followed by adsorption onto clay and the organic matter, reaction with iron, chemical coprecipitation, or settling. After selenium enters the substrate layer, further chemical and microbial reduction may occur, resulting in insoluble organic, mineral, elemental, or adsorbed selenium (Lemly 2002). Over time, most of the selenium associated with plant and animal tissues is deposited as detritus and eventually incorporated into the sediments. Some have reported that isotope data indicate that elemental selenium found in sediments results from a release from decaying plants, and not microbial reduction (Herbel et al. 2002). Some selenium forms may be volatilized to the atmosphere through microbial activity in the water and sediments or through direct release by aquatic plants (Lemly 2002, Eggert et al. 2008). Enhancing volatilization is particularly attractive because it removes selenium entirely from the system, rather than transforming and sequestering it in sediments or biota (Higashi et al. 2005).

Constructed wetlands can be surface flow wetlands (shallow marshes), subsurface flow wetlands (planted beds of gravel or soil media with water flow through the root zone), or vertical flow wetlands (a variation of subsurface flow where overlying water is used to create anoxic conditions within the substrate bed) (CH2M HILL 2010). Subsurface-flow wetlands typically have a better performance in pollutant removal than surface-flow wetlands, but they have a tendency to clog over time, leading to greater costs for operation and maintenance (Kadlec and Wallace 2009 in CH2M HILL 2010). Constructed wetland systems allow regulation of inflow rates and water depth in order to affect hydraulic and mass loading and hydraulic retention time. Retention time is typically longer than in tank-based biological treatment systems (days-weeks rather than hours). The wetland size depends on the rate of inflow, influent concentrations, and desired effluent criteria. Plant characteristics such as root and stem density can also influence selenium removal. Climate conditions can also affect rates of treatment, with less volatilization in colder winter months than summer (Johnson et al. 2009).

Performance of various constructed wetlands experiments and projects has been reviewed in several reports (DWR 2005, NSMP 2007, CH2M HILL 2010, Lin et al. 2010). Treatment effectiveness varies widely for surface flow wetlands, ranging from 0% to 96% (Kadlec and Wallace 2009 reported in CH2M HILL 2010). A few relevant examples are discussed below.

In the San Joaquin Valley (Tulare Lake Drainage District, Corcoran), several wetland treatment ponds were constructed in 1996 to treat agricultural drainage (Gao et al. 2003 and Lin and Terry 2003, reviewed in Lin et al. 2010). Each cell was planted with various combinations of plants (sturdy bulrush (*Schoenoplectus robustus*), Baltic rush (*Juncus balticus*), smooth cordgrass (*Spartina alterniflora*), rabbitsfoot grass (*Polypogon monspeliensis*), saltgrass (*Distichlis spicata*), cattail (*Typha latifolia*), tule (*Schoenoplectus acutus*), and widgeon grass (*Ruppia maritima*). The cells (0.29 acres each) recieved tile drainage water (8-15 cm deep, 3-15 day retention time). After four years, 90% of selenium removed from the water entering the wetlands was retained in the top 10 cm of soil in each wetland, and 48% of the selenium was retained in the top 3 cm. In plants, roots accumulated more selenium than shoots or litter. Fairly high concentrations of selenium (>10 mg/kg) were found in the treatment wetlands, such as 10 mg/kg in rabbitfoot grass roots, 10–17 mg/kg in fallen litter, and 8–15 mg/kg in the top 3-cm layer of sediment. While organic forms of selenium are more bioavailable and therefore potentially more toxic to aquatic biota, it has been suggested that the common organic form (dimethylselenide) is likely to volatilize and limit exposure to aquatic biota (Lin et al. 2010).

In the San Francisco Bay area (Richmond), wetlands were constructed to treat Chevron's oil refinery effluent (reviewed in Albert Environment 2006). The three 30-acre cells treated 1,000 gpm with inflow selenium concentrations of 10-30 µg/L (primarily selenite). Selenium concentrations were reduced 60-70% after one cell, and achieved < 5 µg/L at the outflow. Most of the dissolved selenium was retained in sediments (up to 3-8 mg/kg). Approximately 10-30% of selenium was volatilized. Selenium concentrations were elevated in invertebrates (12.8 – 31.0 mg/kg dw) and black-necked stilt eggs (20.4 mg/kg dw). To mitigate impacts on bird populations, Chevron modified the treatment wetland by increasing plant density and removing islands from the first cell to discourage bird use, which has resulted in decreased selenium concentrations in bird eggs to less than 10 mg/kg (CH2M HILL 2002).

In the Salton Sea basin, two treatment wetlands were constructed on the New River to reduce nutrient levels in water and opportunistically test selenium removal as well (Johnson et al. 2009). The Imperial wetland was supplied with agricultural drainage water and comprised 17.5 hectares, with 9 hectares of water surface area and 1.2 hectares of planted bulrush, tamarisk, and wild grasses. These wetlands received approximately 11,000 m^3 of water per day and had a residence time of 18 days. The Brawley wetland was supplied with water from the New River and comprised 3.6 hectares, with 2 hectares of water surface area and 0.5 hectares of planted bulrush, tamarisk, and wild grasses. This wetland received approximately 2,700 m3 of water per day and had a residence time of 9 days.

Influent concentrations of selenium ranged from 2.7 - 5.4 µg/L at the Imperial wetlands and 2.2 - 3.9 µg/L at the Brawley wetlands (Johnson et al. 2009). Concentrations decreased 22±14% in the Imperial wetlands and 42±13% in the Brawley wetlands. No significant decreases were observed in the month of December. Mass balance calculations indicated mass removal of selenium of 56% for Imperial and 70% for Brawley. Mass removal takes into account evapotranspiration which would normally increase the aqueous concentration of selenium. Selenium loss due to volatilization was estimated between 17% and 61% for the wetlands. All effluent concentrations of selenium were less than the EPA criterion for protection of aquatic life (5 µg/L), except for one sample. However, concentrations in invertebrates (corixids) at the Imperial (4.1 µg/g dw) and Brawley (3.7 µg/g dw) wetlands were at or above the toxicity threshold for invertebrates (3-4 µg/g dw). In the Imperial wetlands, levels of selenium in mosquitofish, carp and shad all exceeded the 4 µg/g dw threshold for fish tissue after the 6 years of operation.

The City of Oxnard constructed wetland pilot-scale studies were conducted for treatment of RO membrane reject water (selenium concentration 12-22 µg/L) (CH2M HILL 2007 reported in CH2M HILL 2010). Wetland mesocosms (1 cubic meter) were used to test different wetland scenarios of wetland features (including vertical upflow, subsurface flow, and subsurface aquatic vegetation). A system consisting of vertical upflow, sub surface flow, and subsurface aquatic vegetation in series decreased selenium concentrations by 67% from inflow (12 µg/L) to outflow (<4 µg/L).A system of vertical upflow, shallow surface flow, and deep surface flow beds in series decreased selenium concentrations by 92% from inflow (12 µg/L) to outflow (<1 µg/L).A single subsurface flow system decreased selenium from 19 µg/L to <1 µg/L (CH2M HILL 2007 reported in CH2M HILL 2010).

One possible approach for enhancing the effectiveness of selenium removal is the acceleration of the process of selenium volatilization (the relatively non-toxic gas dimethyl selenide is released into the air) by amending the inflowing waters with organic carbon. For example in a two year field study, Banuelos and Lin (2007) showed that amending drainage sediments planted with salado grass with methionine and casein increased the selenium loss rates by volatilization by nearly 20-fold (from less than 25 µg m^2 d^{-1} to nearly 435 µg m^2 d^{-1}).

A major concern for constructed treatment wetlands is potential ecological risk of selenium exposure to wildlife, especially if selenium is converted to more bioavailable and toxic forms (Fan et al. 2002, Lemly and Ohlendorf 2002). Thus, while water concentrations of total selenium decrease, concentrations in biota may increase (Higashi et al. 2005). It may be possible to improve design and engineering of wetlands to sequester selenium in sediments (hydrosoil) in reduced (and thereby less bioavailable/toxic) forms, or increase volatilization. Proper selection and implementation of hydrosoil characteristics, hydraulic retention time, and vegetation can be integrated to promote formation of reduced selenium species (Eggert et al. 2008). Use of treatment wetlands by wildlife may also expose them to selenium in sediments and vegetation. Wildlife usage can be reduced by minimizing open water areas with dense plantings (Eggert et al. 2008). This may prove effective for open water species such as waterfowl, but likely less effective for species such as rails that inhabit marsh vegetation. Another approach is to eliminate habitat features attractive to birds, such as islands (CH2M HILL 2002). Deterrent devices such as visuals, water spray, automatic exploders, pyrotechnics, lights, and alarm/distress calls have variable and limited uses (DWR 2005). Complete enclosure is the most effective approach, but costly to implement and impractical for large areas (DWR 2005). Subsurface flow wetlands would have lesser wildlife exposure.

Effectiveness, Removal Efficiency, and Costs

- Selenium removal from full-scale wetlands can vary widely, from 0% up to 96% of selenium, with outflow concentrations as low as ~1 µg/L (Kadlec and Wallace 2009 reported in CH2M HILL 2010). Uncertainties remain for consistently achieving low selenium levels (<5 µg/L).

- To date there is limited information about large-scale subsurface flow wetlands designed for selenium removal; however, laboratory scale tests have reported selenium removal to < 0.2 µg/L (Azaizeh et al. 2007 reported in CH2M HILL 2010).

- Although wetlands can reduce selenium concentrations in water, selenium concentrations in biota can be elevated (Higashi et al. 2005, Alberta Environment 2006, Johnson et al. 2009, CH2M HILL 2010).

- Treatment wetlands require a large footprint and long hydraulic retention time. Surface flow wetlands designed to treat municipal and industrial waste typically range from 10 to 1,000 acres (CH2M HILL 2010).

- Effectiveness is affected by temperature, with greater removal in warmer summer months (Johnson et al. 2009, CH2M HILL 2010).

- Costs for constructed treatment wetlands are relatively low compared to other treatment technologies. Capital costs for subsurface wetlands for treatment of 1 mgd flowrate are approximately $17 million, with annual O&M cost approximately $150,000 (CH2M HILL 2010). In general, costs for surface wetlands are lower than for surface wetlands.

Table 2 – Selenium Treatment Technologies Overview

Technology	Description	Efficiency	Applicability/ Development Stage	Advantages	Considerations / Disadvantages	Capital and O&M Costs[1]
PHYSICAL						
Reverse Osmosis	Forces water through a membrane against its concentration gradient, separating constituents like selenium out of the water.	Demonstrated Se removal to <5µg/L. Between 70 and 90% recoveries in the reject water.	Full scale operations to treat wastewater. Large amount of data available.	Produces high-quality water. Small space requirements.	High energy cost. Pretreatment of influent often necessary. Treatment and disposal of the brine "reject" water is necessary. Membranes need to be replaced every 3-5 years due to scale and fouling.	1 million gallons per day (mgd) system has an estimated capital cost $40 million and $3 million annual O&M.
Nanofiltration	Membranes are used to separate different fluids or ions.	Up to 95% removal (laboratory-scale). Pilot scale reduced Se to <5µg/L.	Pilot scale. Not well tested at full-scale for selenium removal.	Operates at lower pressure than RO. Relatively small space requirements.	Not much performance data for selenium removal at large scale. Requires pre-treatment to decrease fouling and frequent membrane maintenance. Requires post-treatment of reject stream.	For a 1 mgd system estimated $40 million capital cost and $3 million annual O&M cost.
Ion Exchange	An absorption process where undesirable ions in the water are exchanged or like charged ions by electrostatic attraction to sites of opposite change on the surface of granular chemicals (weak and strong base anionic resins).	Generally greater than 90%, but limited data on Se removal to < 5 µg/L	Pilot scale. Not well tested at full scale to treat selenium.	Has potential to treat selenium < 5 µg/L. Concentrates selenium, thus decreasing volume of post-treatment waste.	Relatively limited data for performance removing Se to low levels. Resin can become saturated and need to be "recharged".	1 mgd system estimated capital cost $28 million and $4 million annual O&M.
CHEMICAL						
Zero Valent Iron	Bioreactor with ZVI material (power, granual or fibrous forms of iron)	80-90% removal. Limited data to demonstrate Se removal to <5µg/L.	Pilot scale. Used in acid mine drainage. San Joaquin Valley has considered	Reduces Se effectively in lab.	O&M requirements for iron mass, including replacement and removal of spent ZVI and sludge disposal. Requires long residence time. Need pH 4 to reduce	1 mgd system estimated capital cost $13 million and $3 million annual O&M.

Technology	Description	Efficiency	Applicability/ Development Stage	Advantages	Considerations / Disadvantages	Capital and O&M Costs[1]
					selenate to selenite.	
Ferrous Hydroxide	Uses ferrous iron (Ferrihydrite or ferrous hydroxide) to reduce selenium to elemental form which precipitates out.	Se removal not proven to < 5 µg/L.	Full scale implementation.	Widely implemented full-scale for industry. Relatively simple technology.	Limited amount of data showing treatment < 5 µg/L. Sludge removal/ treatment needed. Requires pH 9 for operations.	300 gpm treatment system for a flue gas desulfurization (FGD) power plant was $15 million capital cost and $1.5-2 million annual O&M costs (in 2000 USD)
BIOLOGICAL						
Anaerobic Bacteria Removal	Bioreactor; reactor with carbon feed stock (molasses, methane, methanol, sugar products); low tech to high tech range. ABMet – bioreactor with an attached growth film.	Demonstrated Se removal to < 5 µg/L in pilot-scale and full-scale implementa-tion.	Full scale implementation	Commercially available, proven technology, Uses naturally occurring microbes. Demonstrated technology to reduce Se to < 5 µg/L.	May require pre-treatment to remove suspended solids. External carbon source (molasses-based) needed, especially if high nitrates. Can be tricky to maintain bacterial culture. Backwash water required. Sludge disposal required. Large footprint.	1 mgd system estimated $30 million capital cost and $3 million annual O&M cost. $1.23-1.48/ 1,000 gal for a 10 mgd system (Alberta Environment 2006)
Algal Treatment	*Algal Bacterial Selenium Removal (ABSR)* - algae as a food source to stimulate bacterial growth *Algal Assimilation approach* Controlled Eutrophication Process (CEP) - High-growth rate algae take up nutrients and selenium,	ABSR - about 80% selenate removal, but not proven to achieve Se levels < 5 µg/L CEP - not tested for selenium removal	ABSR pilot scale CEP pilot-scale - tested for nutrient reduction, but not for selenium removal.	Potentially low cost. Some direct volatilisation of Se. Can help nutrient removal. CEP can operate at wide range of salinity.	ABSR effluent contains higher levels of bioavailable Se forms. Seasonally limited by temperature and light. Requires further treatment to remove algae (settling, flocculent, fish grazing). Not yet demonstrated to achieve Se levels < 5 µg/L	In situ algal-bacterial systems costs $0.0008/ gal (CH2M HILL 2010) CEP pilot for 13 mgd system - capital cost $5-6 million, annual O&M cost $1.6-1.8 million (Kent BioEnergy)

Technology	Description	Efficiency	Applicability/ Development Stage	Advantages	Considerations / Disadvantages	Capital and O&M Costs[1]
	support bacterial processes					
Constructed Wetlands	Treatment cells planted with wetland vegetation to support microbes (on roots) that reduce or volatilize Se. Flow configurations can include surface flow, subsurface flow, or vertical upflow.	Highly variable, up to 96% Se removal and sometimes achieve <5 µg/L.	Pilot scale implementation. Studied at large scale	Low technology passive process with relatively low operational costs compared to other technologies. May not require pre-treatment. Potential to reduce Se up to 96%. Performs better in warmer temperatures. Able to treat large volumes of water. Potential to enhance volatilization. Could add carbon (e.g. manure, molasses, straw bales) to stimulate bacterial reduction and require less land (smaller wetlands).	Uncertainties about consistently achieving Se <5 µg/L. Requires long retention times (several days) and therefore large footprint. Could transform Se into more bioavailable forms and assimilate Se into vegetation, posing potential risk to wildlife, especially at surface flow wetlands. Performance less effective in cool weather. Subsurface flow wetlands can become clogged.	1 mgd system (subsurface flow) estimated $17 million capital cost and $150,000 annual O&M cost.

[1]Cost estimates in 2010 Year Cost (US Dollars) unless otherwise specified. Estimates are for a system to treat 1 (one) million gallons per day. However, the Salton Sea SCH Project would likely be using more water.

Sources: CH2M HILL 2010 (most cost data), Alberta Environment 2006, NSMP 2007.

SECTION 3

Conclusions

This report provides an updated review of the available technologies to aid in SCH project design and evaluation. Water treatment to remove selenium is but one potential tool to minimize exposure of ecological receptors, and would be applicable if other mitigation measures failed to sufficiently reduce ecological risk. The SCH Project selenium management strategy involves identifying wildlife at risk, identifying and quantifying sources and concentrations of selenium, characterizing foodweb pathways and bioaccumulation risk, development of source control and mitigation measures to reduce wildlife exposure, and possibly water treatment if necessary and feasible.

Although several physical, chemical and biological treatment technologies have the potential to remove selenium, few have reliably reduced selenium in water to less than 5 µg/L at any scale, and still fewer have been successfully implemented at full-scale for sufficient time to demonstrate the long-term feasibility of selenium removal technology (CH2M HILL 2010). Physical treatments (reverse osmosis, nanofiltration) can be very effective, but are largely incompatible for the SCH Project due to cost and complexity. Chemical treatment with iron is also expensive, and has not been demonstrated to reduce low levels of selenium. The expert panel concluded that physical and chemical treatments were not applicable or feasible for the SCH Project, and recommended a focus on simpler management strategies rather than reliance on complex or intensive water treatment technologies.

Of the treatment technologies reviewed, biological treatment, particularly constructed wetlands, appears to have the most applicability to the SCH Project. Biological treatments offer the advantage of being relatively low cost and maintenance compared to physical treatments. However, there is lack of consensus among experts and in the literature regarding the potential effectiveness and impacts of biological treatment techniques, particularly constructed treatment wetlands.

One issue is whether treatment wetlands can reliably reduce selenium levels to less than 5 µg/L or even 2 µg/L. Some panelists suggested investigating ways to increase treatment efficiency of wetlands. One approach is to enhance volatilization (Lin and Terry 2003) either by selecting wetland plant species that are more effective at volatilization or adding a carbon source (e.g. molasses) to stimulate bacterial processes. The removal of selenium by biological volatilization to the atmosphere is attractive because it leads to a net loss from the aquatic system, thereby preventing its entry into the food chain. Measurements of selenium volatilization by wetlands have shown that approximately 2-10% of inflow selenium may be volatilized annually, but these rates may increase to 50 to 60% during summer, or by the application of organic amendments and other management practices (e.g., de-topping) (Huang et al. in prep). Laboratory studies can evaluate the effectiveness of different plant species for volatilization (Huang et al. in prep.) Criteria for selecting plants include ability to sequester or volatilize selenium, rapid growth and spread, and suitability for the Salton Sea climate and habitat. Also, the Brawley and Imperial constructed wetlands provide an opportunity to pilot test enhancement methodologies that could be scaled-up to treat river flows before discharge to the future SCH ponds. Other biological treatment technologies such as CEP may further remove selenium and could be combined with constructed wetlands as a polishing step.

Another issue is whether biological treatment may transform selenium into more bioavailable forms (Fan et al. 2002, Lemly and Ohlendorf 2002). Concentrations of total selenium may decrease in the water (dissolved selenium) but increase in the particulates and bioaccumulate in invertebrates (Higashi et al. 2005).

Finally, concerns were raised regarding wildlife exposure at the treatment wetland itself, which would sequester and likely accumulate selenium within its sediments, detritus, and biota. Selenium monitoring of the SCH Project should include biota as well as water and sediment, for both the SCH ponds and treatment wetlands, if implemented. Design features and strategies to reduce wildlife exposure would need to be a part of a biological treatment technology if implemented. For example, wetlands could be designed with dense plantings to reduce amount of open water habitat. This may deter open water species such as waterfowl and terns, but is likely to less effective for other marsh species such as rails. Other deterrent methods are possible, but may be challenging to implement effectively (DWR 2005).

Many questions remain to be addressed if the SCH Project is to pursue implementation of treatment wetlands as part of its overall selenium management plan. The design requirements of a constructed treatment wetland would include size, shape and location of the wetlands; appropriate plant species; desirable target selenium concentration for the treated effluent; estimation of water requirements; and possible design features to minimize selenium eco-toxicity in birds (Huang et al. in prep.). Research can be undertaken to enhance capability to efficiently remove selenium as well as nutrients and total suspended solids from river water flowing into the SCH and to develop a conceptual design to minimize eco-toxic effects within the SCH-treatment wetland complex. Studies have been proposed using a combination of laboratory and field studies to test different ways of 1) increasing the immobilization and retention of selenium in non-bioavailable forms in the wetland sediments, and 2) accelerating the rate of wetland selenium removal by volatilization to the atmosphere (Huang et al. in prep.). Information from this and other sources will be used to refine and improve SCH Project design, evaluation and implementation within an adaptive management framework.

References

Alberta Environment. Evaluation of Treatment Options to Reduce Water-Borne Selenium at Coal Mines in West-Central Alberta. Pub No. T/860.

Alcoa Inc. 2009. Alcoa Inc. – Bauxite Environmental Improvement Project Notice of Intent. October 5 2009.

Amrhein, C. and W. Smith. 2010. Survey of sediments at prospective SCH sites. Report prepared by University of California Riverside for Department of Water Resources. July 30, 2010 Draft. 30 pp.

Anderson, T.W. 2008. Avian Use and Selenium Risks Evaluated at a Constructed Saline Habitat Complex at the Salton Sea, California. Masters Thesis. San Diego State University.

Carlberg, J. M., J. C. Van Olst, M. J. Massingill, R. J. Chamberlain, G. Schwartz, D. E. Brune, and J. R. Benemann. 2003. Use of the Controlled Eutrophication Process (CEP) to reduce nitrogen and phosphorus concentrations entering the Salton Sea. Salton Sea Biological remediation Program. Final Report.

CH2M HILL. 2010. Review of available technologies for the removal of selenium from water. Final Report. Prepared for the North American Metals Council, June 2010.

Department of Water Resources (DWR). 2005. Salton Sea Ecosystem Restoration Plan. Technologies and Management Techniques to Limit Exposures to Selenium. Final Report. April 2005.

DWR and California Department of Fish and Game (CDFG). 2006. Salton Sea Ecosystem Restoration Program Draft Programmatic Environmental Impact Report. Prepared by CH2M HILL for California Resources Agency, California Department of Water Resources and California Department of Fish and Game.

DWR and CDFG. 2006. Salton Sea Ecosystem Restoration Program Draft Programmatic Environmental Impact Report. Appendix F: Ecological Risk Assessment. Prepared by CH2M HILL for California Resources Agency, California Department of Water Resources and California Department of Fish and Game.

Eggert, D.A., J.H. Rodgers, Jr., G.M. Huddleston, and C.E. Hensman. 2008. Performance of pilot-scale constructed wetland treatment systems for flue gas desulfurization (FGD) waters. Environmental Geosciences 15(3): 115-129.

Fan, T.W.-M, S.J. Teh, D.E. Hinton, and R.M. Higashi. 2002. Selenium biotransformations into proteinaceous forms by foodweb organisms of selenium-laden drainage waters in California. Aquatic Toxicology. 57: 65-84.

Frankenberger, W.T., Jr., C. Amrhein, T.W.-M. Fan, D. Flaschi, E. Kartinen, Jr., K. Kovac, E. Lee, H.M. Ohlendorf, L. Owens, N. Terry, and A. Toto. 2004. Advanced treatment technologies in the remediation of seleniferous drainage waters and sediments. Irrigation and Drainage Systems 18:19-41.

Gusek, P.E., K. Conroy, and T. Rutkowski. 2008. Past, Present, and Future for Treating Selenium-Impacted Water. Annual Meeting of the West Virginia Surface Mine Drainage Task Force, Morgantown, WV.

Hamilton, S. J. 2004. Review of selenium toxicity in the aquatic food chain. *Science of the Total Environment* 326: 1-31.

Herbel, M.J., T.M. Johnson, K.K. Tanji, S. Gao, and T.D. Bullen. 2002. Selenium stable isotope ratios in California agricultural drainage water management systems. *Journal of Environmental Quality* 31: 1146-1156.

Higashi, R.M., T.A. Cassel, J.P. Skorupa, and T.W.-M. Fan. 2005. Remediation and bioremediation of selenium-contaminated waters. Pp. 355-360 in: J.H. Lehr and J. Keeley (Eds.), *Water Encyclopedia: Water Quality and Resource Development*. Wiley: Hoboken, New Jersey, 717 pp.

Huang, J., M. Verce, A. Stiles, and N. Terry. 2010. Development of a constructed wetland treatment system for the Salton Sea Species Conservation Habitat – A review. Draft report prepared by University of California, Berkeley, Ca. 42 pp.

Johnson, P. I., R. M. Gersberg, M. Rigby, and S. Roy. 2009. The fate of selenium in the Imperial and Brawley constructed wetlands in the Imperial Valley (California). Ecological Engineering 35: 908-913.

Lemly, A.D. 2002. Selenium Assessment in Aquatic Ecosystems. Springer-Verlag, New York, pp. 18–32.

Lin, Z., and Terry, N. 2003. Selenium removal by constructed wetlands: quantitative importance of biological volatilization in the treatment of selenium-laden agricultural drainage water. Environ. Sci. Technol. 37: 606–615.

Lin, Z., N. Terry, S. Gao, S. Mohamed, and Z.H. Ye. 2010. Vegetation Changes and Partitioning of Selenium in 4-Year-Old Constructed Wetlands Treating Agricultural Drainage. International Journal of Phytoremediation. March 26 2010.

Luoma, S. N., and T. S. Presser. 2009. Emerging opportunities in management of selenium contamination. *Environmental Science & Technology* 43: 8483-8487. Available online at: http://pubs.acs.org/doi/pdfplus/10.1021/es900828h?cookieSet=1.

Masscheleyn, P.H. and W.H. Patrick, Jr. 1993. Biogeochemical processes affecting selenium cycling in wetlands. Environmental Toxicology and Chemistry 12: 2235-2243.

Miles, A.K., M.A. Ricca, A. Meckstroth and S.E. Spring. 2009. Salton Sea Ecosystem Monitoring Project. U.S. Geological Survey Open-File Report 2009 – 1276, 150 pp.

Nitrogen and Selenium Management Program (NSMP). 2007. Identification and assessment of selenium and nitrogen treatment technologies and best management practices. Prepared for the NSMP Working Group. March 30.

Nitrogen and Selenium Management Program (NSMP). 2008. Final Report. Pilot Test Report for Nitrogen and Selenium Removal Technologies. Newport Bay Watershed. March 21 2008.

Ohlendorf, H.M. and G.H. Heinz. (in press) Selenium in birds. In: W.N. Beyer and J.P. Meador (eds.) *Environmental Contaminants in Biota: Interpreting Tissue Concentrations.* Second Edition. Taylor and Francis, Boca Raton, FL.

Quinn, N. W. T., T. J. Lundquist, F. B. Green, M. A. Zárate, W. J. Oswald, and T. Leighton. 2000. Algal-bacterial treatment facility removes selenium from drainage water. *California Agriculture* 54(6) 50-56.

Robberecht, H. and R. Van Grieken. 1982. Selenium in environmental waters: determination, speciation, and concentration levels. Talanta 29: 823-844.

Saiki, M. K., B. A. Martin, and T. W. May. 2010. Final report: Baseline selenium monitoring of agricultural drains operated by the Imperial Irrigation District in the Salton Sea Basin: U.S. Geological Survey Open-File Report 2010-1064, 100 p.

Salton Sea Species Conservation Habitat Project Draft Implementation Plan. March 15 2010.