Great Salt Lake as an Ecologically Significant Natural Area
Presented by: Great Salt Lake Institute

Summary

This White Paper summarizes research from the scientific literature, the brine shrimp industry, the mineral extraction industry, and ongoing research at all of the major institutions of higher education in Utah. Great Salt Lake Institute has collected this information and complied it for the Utah Division of Waste Management and Radiation Control to guide decision-making regarding permitting of a Class V landfill operation in close proximity to the Lake.

Great Salt Lake (GSL) is a unique, fragile, critical, and economically viable body of water, the largest lake in the U.S. west of the Great Lakes. As a terminal lake, any input into the water remains there in perpetuity, including minerals but also pollutants. Industry and development efforts around the shores of Great Salt Lake must operate without damaging this fragile ecosystem and with the understanding that a terminal lake is particularly sensitive to potential contaminants. In particular, a landfill operating close to the Lake and near the fault lines beneath the lake, should not contain any materials deemed hazardous to life in the Lake or dangerous to humans as airborne contaminants. Any benefit derived from such operations does not outweigh the potential detriment to the GSL ecosystem, which is an ecologically significant natural area.

Outline:

I. Great Salt Lake Overview

The Bonneville Basin and formation of Great Salt Lake

The Great Basin is the largest contiguous inland watershed in North America, surrounded by mountain ranges and the Wasatch fault zone (Cohenour and Thompson, 1966). In this setting, Great Salt Lake (GSL) lies in one of the lowest depressions, the Bonneville basin. During the Pleistocene (32 to 14 thousand years ago), this terminal lake locale held the large freshwater Lake Bonneville, covering about 20,000 square miles of western Utah and extended into eastern Nevada and southern Idaho (Oviatt et al., 1999; Shroder et al., 2016). However, the Bonneville basin primarily held shallow lakes such as GSL, or mudflats and playa, likely over the last several million years (Atwood et al., 2016). The transition of Lake Bonneville to GSL occurred over just a few thousand years. The water evaporated and leaked out in stages, resulting in the current GSL lake margins about 13,000 years ago.

Modern GSL

GSL is the largest lake in the western United States, the fourth largest meromictic lake in the world and the second saltiest lake on Earth next to the Dead Sea (Keck and Hassibe, 1979). The major sources of freshwater inflow to GSL comes from the Bear, Weber, and Jordan Rivers, which are part of an extensive drainage (Jones et al., 2009). Though the precise margins of GSL vary with seasonal precipitation and drought cycles, it measures approximately 122 km in length and 50 km in width with an average depth of 4.3 m and a maximum depth of 9 m (Keck and Hassibe, 1979; Stephens, 1990). This lake experiences significant seasonal temperature variation, from 0.5°C in January to 26.7°C in July (Crosman and Horel, 2009) and up to 45^{o}C in the shallow margins (Post, 1977) due to its elevation and desert setting.

Terminal Basin fluctuations

GSL is a terminal lake. Its elevation or water level is subject to rising and falling in response to precipitation, drought, and temperature. Water may leave the Lake through evaporation, but any compound that enters the Lake is there forever and may become a part of the food web (Wurtsbaugh et al., 2011; Naftz et al., 2008). Years of recent drought set a new historic low in 2016 (USGS, 2018a), the lowest elevation since 1963 (Stephens, 1990). In addition, climate change could be dire for the communities surrounding GSL because air quality is an issue with shrinking terminal lakes (Wurtsbaugh, et al., 2017). If the Lake continues to recede, the former lake bottom may become part of the salt playa, and contaminants stored there may enter the air. Understanding this dynamic terminal lake is key to managing both potential water quality and air quality problems.

Local conditions are impacted by haloclines, where brines of different concentrations do not mix, resulting in brine stratification (Naftz, 2008; Meuser et al., 2013). Unpredictably, weather systems can cause mixing and turnover to alter the salinity in local regions, and mixing of brines can impact the specific ion concentrations of an area (Spencer et al., 1985). As noted, the GSL elevated terminal lake ecosystem is subject to natural desiccation cycles and seasonal temperature fluctuation. However, current upstream water demands coupled with climate change are having an enormous impact on the lake elevation, and 2016 saw a historic low for GSL (Fritz, 2014; Deamer, 2016; USGS, 2018a). Empirical drought reconstruction predicts a catastrophic mega-drought in the southwestern the United States, which includes Utah, and suggests that GSL may not recover water elevation gain in the near future (Cook et al., 2015).

Anthropomorphic Impacts
The construction of a rock-filled railroad causeway from 1955-1959 bisected GSL and isolated the NorthArm of the Lake, restricting exchange and creating an artificial salinity gradient (Madison, 1970; Cannon & Cannon, 2002; Baxter et al., 2005). Within seven years, a salinity gradient was noted as the North Arm was approaching saturation, while the South Arm, which receives the freshwater input from the watershed, was less saline (Greer, 1971). In recent years, the open waters of the South Arm of GSL ranged from 110-150 g/L (USGS, 2018a) while the North Arm is at saturation (280 g/L-340g/L, dependent on the temperature) (e.g. Baxter et al., 2005; Almeida-Dalmet et al., 2015).

The causeway separating the north and south arms has also caused brine stratification. The formation of a deep brine layer is occasionally observed, due to the denser North Arm water seeping under the causeway, leaking through porous material, below the less saline South Arm water (Naftz, 2008; Meuser et al., 2013). Recently (December, 2016), the breach in the railroad causeway was restored, allowing South Arm water to pour into the North Arm through a repaired opening large enough to let boats pass (USGS, 2018b). As the lake equilibrates from this action, we may see the formation of new salinity gradients and microniches.

Other damming events created critical bird habitats and are part of the Western Hemispheric Shorebird Reserve Network. As early as the late 1800s, damming and diversion of water upstream of GSL to create agricultural lands resulted in freshwater marshes or brackish pools with a lower salinity. In 1928, the U.S. Congress passed an act to make the Bear River delta a National Wildlife Refuge (US Fish and Wildlife, 2018). Later, federal agencies diked and dammed the Bear River, to produce bird habitat there (United States Bureau of Reclamation, 1962). The structures are on the margin of the Lake and have been maintained over time, preventing inflow of this water to GSL. For similar reasons, the Farmington Bay Wildlife Management Area was constructed beginning in 1935, which created a brackish bay to the east of Antelope Island in GSL (State of Utah, 2018a). Carp can survive in the 2-3% salinity, and they are food for large waterfowl such as Bald Eagles and American White Pelicans.

II. Geology and Hydrology
Lake Bed Geology and Seismic Data
The Utah Geologic Survey (UGS) has not completed updated intermediate-scale or detailed-scale geologic maps of the area. Preliminary site-specific geologic data along the east side indicate a thin cover of Quaternary surficial deposits overlying faulted and fractured bedrock (Crittenden, 1988).

Fault lines have been mapped beneath GSL. Geologists at the UGS suggest that in an earthquake event, seismic impacts that would affect the GSL area include ground-shaking, slope stability, tsunamis and seiching. They report that the fault-mapping is currently incomplete as fault zone data from recent projects has not been reconciled with older data. Potentially active faults impact future transportation routes, especially the proposed railroad spur.

Microbialites, calcium carbonate organosedimentary mounds precipitated by associated microororganism, are distributed near fault lines on the GSL lake bottom (Baskin, 2014; Chidsey et al., 2015; Lindsay et al., 2017). Current models of microbialite formation, in sites around the world, suggest that they form where groundwater seeps occur as the groundwater would be necessary to bring calcium and form the calcium carbonate material. The microbialite structures in GSL are the densest on the shallow shelfs bounded by faults as the water depth changes. This suggests that the major faults under GSL along the west side, may be seeping groundwater.

Groundwater

Since contaminants have staying power in GSL, connections between the shoreline and the Lake via groundwater are of deep concern. Any industry operation near the Lake that stores, produces or utilizes materials toxic to the food web or dangerous to human health, such as the proposed Class V landfill, cannot assure leaching of materials into GSL will not occur. Proximal lakebed spring systems and fractured bedrock may present a preferential flow-path for groundwater, and any contaminant emanating from the proposed landfill, to reach this unique and significant ecosystem. **<u>See Appendix i</u>** for more information on the hydrogeology of the GSL system.

According to the UGS, the bedrock aquifer system has not been fully characterized. For any industrial site under consideration, a study should be undertaken including performing a seismic survey to locate major fractures, faults, and other structural features in the subsurface. Furthermore, a map should be generated of the water table, perched aquifers, and hydraulic conductivity of the bedrock aquifer. There is potential for storm-water runoff to breach any industrial operation and transport waste directly to GSL. Therefore, dangerous materials, such as the coal ash or other toxic waste materials proposed to be stored in the Class V landfill, should be prohibited from the lake shore.

III. Ecology and Food Web

Algae and Microbialites

The food web is simple from the macro viewpoint, and the more complex biochemistries of autotrophy and turnover of nutrients occur in the microbial communities. Like in most aquatic ecosystems, algae are the primary producers that harvest energy from sunlight and provide the first trophic level of the food web. The microbial community of the water column, especially the algae, provide nutrition for the invertebrates (e.g. Belovsky et al., 2011), which feed the birds of the Lake.

Some algae are associated with the microbialites (e.g. stromatolites, "bioherms," "biostromes"), which are structures precipitated on the lake floor by the chemistry of photosynthesis (e.g. Lindsay, et al., 2017). As these microorganisms turn sunlight into food, the power the Lake's ecosystem. Other microbes in the microbialite community may methylate mercury (Boyd, Lindsay and Baxter, unpublished), making the heavy metal more toxic to living systems. These are feeding stations for brine fly pupae, which are eaten by diving birds (e.g. ducks, phalaropes). For these reasons and others, microbialites are a key focal point of understanding the Lake ecology.

Invertebrates

The primary invertebrate organisms in the Lake, brine shrimp (*Artemia franciscana)* and brine flies (*Ephedra* spp.) (Verrill, 1869; Packard, 1871; Aldrich, 1912; Collins, 1980; Wurtsbaugh and Gliwicz, 2001; Roberts, 2013). While *Artemia* spend their entire life in the brine, the fly eggs and larvae mature in the water, and they pupate on the lake bottom, primarily on the microbialite structures (Wurtsbaugh et al., 2011). The adult flies fly above the surface and on the shore. The phytoplankton and periphyton of GSL feed these invertebrates (e.g. Felix and Rushford, 1979; Collins, 1980; Wurtsbaugh and Gliwicz, 2001; Belovsky et al., 2011; Barnes and Wurtsbaugh, 2015). *Artemia* and *Ephedra* may bioaccumulate metals and other pollutants (e.g. Gebhardt, 1976; Wurtsbaugh et al., 2011), which impacts the food chain above them. An in-depth discussion is attached regarding the role that *Artemia* plays in the food change and the sensitivity of this invertebrate to pollutants and biomagnification of metals. **See Appendix ii.**

Birds

GSL is one of the most important places for birds, not in the state, nor in the nation, but in *the world*. Our Lake features six sites out of the 720 named global "Important Bird Areas" (National Audubon Society, 2018). Thus, GSL was designated part of the "Western Hemisphere Shorebird Reserve Network," which gives international recognition to critically important shorebird habitats and promotes cooperative management and protection of the sites as part of an international reserve network (Western Hemisphere Shorebird Reserve Network, 2018).
A critical stop on the Pacific flyway, GSL hosts around ten million waterbirds (around 250 species) that spend at least part of the year here, making the Lake the most important shorebird site in North America (Bellrose, 1980; Oring et al., 2000; Paul and Manning, 2002; Aldrich and Paul 2002; Neill et al., 2016). GSL offers its avian inhabitants and visitors an enormous biomass of food in the form of *Artemia* and *Ephedra* in the open water, the benthic areas and the shoreline. In addition, the Lake has several bays and wetland areas which amplify the significance of this site for avian travelers. Bear River Bay (Bay) lies in the northeast arm of the GSL between the Promontory Mountains to the west and the Wasatch Mountains to the east and includes Bear River Migratory Bird Refuge (Refuge), the largest freshwater marsh complex on the GSL, and the Willard Spur. Together, these biologically productive shallow wetland habitats of the Bay support a significant portion of the avian population. **See Appendix iii** for an exhaustive list of species which utilize the Lake and its wetlands in their life cycle.

Toxins entering GSL can be a serious concern to migrating birds. Several studies have evaluated selenium and mercury concentrations in migratory birds using GSL including migrating Eared Grebe (Conover and Vest 2009a), wintering ducks (Vest et al. 2009), and breeding California Gull (Conover and Vest 2009b). "A healthy system also includes one in which very few birds die each year from avian botulism and in which other emerging toxins of concern are kept at low levels that are not harmful to birds or their food resources" (Great Salt Lake Advisory Council, 2012a).

Food Web Considerations

We now have a great deal of data on the sequestration and bio-accumulation of mercury and selenium in the Lake (e.g. Naftz et al. 2008; Beisnera et al., 2009; Boyd et al., 2017). Water samples from the GSL have exceeded the total mercury standard for protection of aquatic life in marine systems and were among the highest values observed for marine systems (Naftz et al. 2008). Additionally, high selenium concentrations have been reported in GSL water samples

(Brix et al. 2004). We know about the potential for bioaccumulation in microbialites (Wurtsbaugh et al., 2011). We know how heavy metals work their way up the food-chain and end up in ducks that are eaten by humans (Naftz et al. 2008) or moved to the terrestrial ecosystem into spiders and then birds (Saxton et al., 2013). We know the impact of heavy metals is a serious concern for not just those of us in Utah but also those on the paths of migrating water fowl who may eat these birds.

In the past, the biota of GSL were not given the protection of even the Clean Water Act due to the absence of fish. Given what we know about the biomagnification of toxins in the system today, *we are charged with preventing further contamination of this sensitive and fragile ecosystem.* A Class V landfill is contrary to this charge.

IV. Air Quality

As a terminal basin, GSL serves to collect and concentrate any contaminants that enter the system. As climate change and upstream water demands reduce the elevation of GSL (e.g. Wurtsbaugh et al., 2017), we are left with increasing amounts of salt playa and dust that may blow into the valley along the Wasatch Front. Dust storms impact air quality and also deliver toxic substances, both of which are concerns for the people who live next to the source (Griffin and Kelloff, 2004). The drying of Owens Lake, which provided Los Angeles water, has been a remarkable case study of the significant impact on air quality from blowing dust emanating from a dry lake bed (Barone et al., 1981).

Particulates and heavy metals have consequences for human health. **See Appendix ii** for a discussion of the health hazards of heavy metals. Placement of any toxins that threaten human health adjacent to GSL, such as the coal ash and other toxic wastes proposed to be stored in the Class V landfill, should take into account not only wind impacts on fugitive waste and potential groundwater leakage, but also the type of waste and its potential health risk as an airborne contaminant.

V. Economy

Industry and Economic Services

Beyond the concerns for maintaining the GSL ecosystem and preserving human health, the economy of Utah relies on our salty body of water. The brine shrimp industry contributes to and supports the Division of Wildlife Resources (DWR) GSL Ecosystem Project and is an active participant in the prudent and sustainable management of GSL (State of Utah, 2018d), **see Appendix ii**. 99% of brine shrimp harvested on GSL are exported and the *Artemia* cysts produced by the GSL brine shrimp industry provide one of the most important ingredients for the global aquaculture industry and, therefore, provides food and protein to the world. The GSL mineral extraction industry produces sodium chloride (road and softener salt), magnesium chloride (for steel production), and potassium sulfate (fertilizer) (Behrens, 1980; Bingham, 1980; State of Utah, 2018b). Similarly, these products are shipped globally. Therefore, we should consider the economic impact of threats to GSL.

GSL contributes more than \$1.32 billion annually to the state of Utah (Great Salt Lake Advisory Council, 2012b) with the following break-down of use categories:

Mineral extraction industry: $1,130,800,000
Artemia harvesting industry: $56,700,000
Recreation: $135,800,000

The 2012 report showed 7,706 jobs were connected to these activities. Also, the "net economic value" was calculated for the value attributed to GSL from people who recreate on the Lake (who would agree to pay a fee for use) and also for the publicly owned treatment works (POTW) who discharge into the lake system with less expensive treatment standards than would be required for freshwater. For both recreational and POTW use of GSL, the report estimated an additional economic benefit to Utah, in the range of $46 million to $95 million annually. These "economic services" provided by the Lake should be valued and protected, and maintaining a healthy Lake (Great Salt Lake Advisory Council, 2012a) is connected to the financial rewards GSL brings as a resource.

Tourism and Recreation

Tourism represented an important industry in Utah, and among tourist inquiries for the state, GSL is one of the top destinations (State of Utah, 2018c). Bird watchers attend the GSL Bird Festival (Davis County, 2018) or to visit the Lake on their own to watch birds at the Bear River Refuge, Farmington, Antelope Island or other locations. *The Spiral Jetty*, an artwork created by Robert Smithson, built into the North Arm of GSL, attracted more than 13,000 vehicles annually (Great Salt Lake Institute and Dia Foundation car counting data, unpublished).

Hunting is often used as an indication of the health of a population of wildlife. Millions of hunters follow reports to maximize their sport. Only areas with significant numbers of waterfowl rank as top hunting destinations, and GSL is one of these spots (Ducks Unlimited, 2018). Utah Division of Wildlife Resources issues approximately 16,500 duck hunting licenses per year (Realtree, 2018).

VI. Appendices (to follow references)

i. Hydrogeology of Great Salt Lake
ii. Pollution risks to biota (esp. brine shrimp)
iii. Birds of Great Salt Lake

VII. References

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3 March 2018

Mr. Rob Dubuc
Western Resource Advocates

Re: Promontory Point Landfill Classification Proposal

Dear Mr. Dubuc,

I have studied hydrogeology and water quality in the Great Salt Lake watershed over the past twelve years. Much of my Ph.D. dissertation was focused on understanding trace element cycling in the Great Salt Lake and adjacent freshwater wetlands. My research has also included quantifying groundwater fluxes in the arid Silver Island Mountains of western Utah, with a similar climate and topography as the Promontory Mountains. To fund my research program, I have received grants from local, state, and federal agencies. I also teach an upper-level undergraduate course on groundwater and graduate-level courses on contaminant hydrogeology and aquifer test analysis. I have attached my CV, which lists my publications, grants, and teaching experience.

Based on my expertise and experience working on Great Salt Lake, I want to raise concerns about Promontory Point Resource's March 2017 application for a permit to operate a Class V landfill and October 2017 permit modification request to relocate downgradient monitoring wells. According to the Utah Division of Environmental Quality website, a shift from Class 1 to Class V would allow the landfill to operate as a commercial facility, collect waste from out-of-state customers, and receive higher volumes of special waste such as coal ash, petroleum-contaminated soils, and waste asphalt. Groundwater contamination from the landfill could negatively impact the lake ecosystem and water quality. Therefore, the potential impact of the landfill on Great Salt Lake deserves a higher level of scrutiny.

My main concerns about the proposed landfill site and groundwater monitoring plan are outlined below.

- **There is evidence for an active groundwater system beneath the landfill site.** First, shoreline springs and saline marshy areas on the west side of Promontory Point are likely groundwater-fed, with minimal surface water inputs. There needs to be enough groundwater flow to sustain these wetland areas year-round, meaning that groundwater inputs must be equal to evapotranspiration in the marshes. This represents a significant amount of annual groundwater flux in the aquifer system. Second, the high density of microbialites on the lakebed to the west of Promontory Point indicates that there may be offshore groundwater seeps. Current research suggests that microbialites form along faults in the lakebed of Great Salt Lake, and seeps along the faults carry minerals and nutrients that support microbialite growth. Currently, not enough is known about the groundwater system at Promontory Point to risk contamination from the landfill.

- **Fractured bedrock beneath the landfill site may present a preferential flowpath for contaminants from the landfill to reach Great Salt Lake.** According to the permit application, the bedrock beneath the landfill site is composed of "highly- to intensively fractured/jointed" quartzite. Any contamination reaching the bedrock could be efficiently transported to the groundwater system and Great Salt Lake. Before a landfill is built at the Promontory Point site, the bedrock aquifer system needs to be characterized by a variety of geophysical and hydrogeological methods. These methods include performing a seismic survey to locate major fractures, faults, and other structural features in the subsurface, drilling deeper boreholes to map the surface of the water table and perched aquifers, and performing aquifer tests to determine the hydraulic conductivity of the bedrock aquifer. This would allow for a better understanding of the connections between the landfill site and the lake and potential travel times/flow pathways of a contaminant plume.

- **The proposed monitoring well network is inadequate to detect groundwater contamination.** Given the complex geology underlying the landfill site, three downgradient wells does not seem sufficient to detect groundwater contamination. Furthermore, not enough detail is provided about the proposed monitoring well depths and screened intervals. Ideally, there would be nested monitoring wells surrounding the landfill site with well screens at multiple depths in the alluvium and bedrock portions of the perched and water table aquifers. If the wells are not completed deep enough and at only one depth, it is possible that contaminated groundwater could travel through the fractured bedrock below the well screen of the monitoring well. The permit modification request does not provide enough details about the locations and depths of the new monitoring wells. The monitoring well network needs to includes screens at a variety of depths and with greater spatial coverage.

- **There is potential for stormwater runoff to breach the landfill and transport waste directly to Great Salt Lake.** Appendix V of the application states that "The onsite landfill drainage including the diversion channels was designed to carry the 25 year, 24 hour event." Given that the landfill has a projected lifespan of 100+ years, why isn't the landfill drainage designed for at least a 100-yr event? Ideally, it would be designed for a 500-yr event. A flood event could be catastrophic for the landfill and Great Salt Lake.

In my opinion, Promontory Point is a poor location for a landfill. The landfill is located adjacent to a lake that experiences dramatic changes in water level, which could influence groundwater flow paths and aquifer characteristics. The landfill site has pronounced topography with flash-flooding capability. The complex geology of the site, with fractured bedrock and nearby active faults, makes it nearly impossible to insure that the site is stable and that the groundwater would be protected. Finally, the landfill is too close to Great Salt Lake, which is a bird habitat of hemispheric importance and has great economic value to the State. Groundwater or surface water contamination from the landfill could greatly impact Great Salt Lake and cause irreparable damage.

Please feel free contact me for additional information.

Sincerely,

Greg Carling, Ph.D.

Document:

White Paper on Promontory Point Landfill (PPL) and Promontory Point Resources LLC (PPR) Application for Class V Waste Permit

Title:

Risks to Biota and the Ecosystem of Great Salt Lake from the PPL with Particular Emphasis on Potential Harm to the Brine Shrimp (*Artemia franciscana*) Population

Stakeholder:

Great Salt Lake Brine Shrimp Cooperative (GSLBSC)
Ogden, UT 84403

Author:

Brad Marden, Senior Research Scientist
Parliament Fisheries, LLC
Ogden, UT

Date:

February 23, 2018

Statement of Problem

Promontory Point Resources (PPR), a Utah based LLC, purchased a 2000-acre parcel of land in 2016 on Promontory Point Utah and, along with the land acquisition, it acquired a Class I landfill permit. PPR has applied for a modification of the landfill permit to include Class V waste. A Class V waste permit would allow PPR to receive and dispose of all materials included under Class V waste guidelines including, but not limited to, coal ash residues (CCR). PPR is preparing to dispose of coal ash waste on their site and have opined that they have more than a 120-year capacity that can be filled at a rate of 1 million tons of waste per year. The initial permit for landfill development on Promontory Point was based on the clearly defined intention to only receive local municipal waste and that the waste would not contain hazardous materials.

Opinion Regarding the Request for Class V Permit by PPR

The recent request for a Class V permit allowing for the disposal of known hazardous waste, even if the waste is not correctly designated as such due to economic considerations in the law (Korb, 2012; Lemly, 2014), presents an unacceptable and imminent risk to the biota of Great Salt Lake (GSL), as well as to the GSL ecosystem, and it threatens existing commercial activities that extract resources from GSL. Landfills with Class V waste have a history of failure and, in the case of PPL, the proximity to GSL renders this location particularly vulnerable to such calamities (Lemly and Skorupa, 2012). Any miscalculation in design or engineering features that results in leaching, leaking or catastrophic discharge of waste into GSL, will be highly consequential to the ecology and economic value of GSL. The risks, therefore, far outweigh the benefits for the economy of Northern Utah and for the long-term integrity of the GSL ecosystem.

Because of this pronounced mismatch between perceived benefits and known risks, the application for a Class V permit should be denied.

Relevant Facts

- PPR has applied for a permit to accept Class V waste and included in this designation is the ability to receive coal ash waste from coal powered energy plants.
- Coal ash contains heavy metals, toxic elements and carcinogens that have pronounced detrimental impacts on biological systems.
- In spite of the fact that coal ash contains numerous contaminants, it is not designated as hazardous waste due to economic concerns—regulatory agencies have determined that the economic cost of designating coal ash as hazardous waste is too expensive to justify.
- The history of coal ash disposal sites is replete with case studies of intentional, unintentional, and accidental discharge into surrounding terrestrial and aquatic systems causing acute and prolonged harm to aquatic organisms, populations, habitats and ecosystems.
- The economic impact of a spill from coal ash disposal is in the hundreds of millions to billions of dollars in cleanup costs, and these cleanup efforts can have limited positive results.
- GSL is a highly valuable saline system that delivers $1.3 billion in economic benefit to Northern Utah each year, with 7,706 full and part-time jobs comprising a total labor income of $375 million annually from GSL. In addition, the passive use value of GSL is in excess of $100 million. These jobs and the economic benefit to Utah that are derived from GSL are *contingent upon the ecological health of GSL* and maintenance of essential ecosystem functions. Damage to the ecosystem from failures of the proposed Class V PPL site would be devastating for these families and could collapse the annual revenues generated by GSL.
- GSL is already under the influence of multiple anthropogenic and natural stressors and in such a condition additional stressors, such as inundation with toxins or fine particulate matter from CCRs, can send the ecosystem into functional decline or potentially into an irreversible collapse.

Class V Waste, Coal Combustion Residue (CCR) and the PPL Site

The PPR has applied for a permit to accept Class V waste. Class V waste allows for regional waste to be deposited in the landfill and such waste could include a variety of wastes from other states that, under the source state's regulations could be considered hazardous. All of the waste included under a Class V permit is of concern and increases risks to GSL. The most alarming is the possibility of coal combustion residues (CCRs) being allowed within the scope of a Class V permit. Because of enormous concern about CCR waste being deposited at the PPL location, these comments focus on this type of waste, its history of harming human and wildlife health, and destroying habitats and ecosystems across the country.

Although representatives of the PPL have provided mixed messages regarding the type of Class V waste they intend to accept, PPR representatives have indicated that among the Class V waste that they plan to dispose of at the PPL site will be coal ash residue from coal energy producing plants (SLTRIB, 2017). Coal ash is just one component of coal combustion residue (CCR) that is produced in the combustion of coal for energy production. Coal combustion residue consists of fly ash, bottom ash, boiler slag and flue gas desulfurization materials ((USEPA, 2017; Connor, 2015). It is essentially all of the material that remains after combustion of coal and its disposal record has been beset with multiple cases of colossal environmental damage. Although reuse applications have been identified and implemented that seem to have minimal adverse impact on environmental quality, there remain hundreds of millions of tons of waste produced annually that need to be disposed of in a safe and prudent manner. The ability of waste disposal sites to carry out judicious and effective disposal is highly questionable as demonstrated by the poor record of such waste facilities—the history of these sites is littered with numerous accounts of failed systems that caused widespread destruction of waterbodies, aquatic and terrestrial wildlife and that adversely impacted human health.

Priority Coal Ash Contaminants of Concern: Leachability and Toxicity

Most of the known contaminants found in coal ash are toxic to aquatic organisms and can cause long-term damage to both lentic and lotic systems (Rowe, 2014; Rowe, Hopkins and Congdon, 2002; Rowe, Hopkins and Coffman, 2001). The many well-documented contaminants in coal ash have the potential to adversely alter habitat structure and function, decrease biological productivity, and are known to cause a full range of sublethal, lethal, carcinogenic, or metabolically altering impacts on the physiology of resident biota (Lemly, 2015; Lemly and Skorupa, 2012). The scientific literature provides a surfeit of rigorous studies linking coal ash constituents to impacts on biological systems (Ruhl et al., 2012; Ruhl et al., 2010; Karuppiah and Gupta, 1997; Carlson and Adriano, 1993; Yount and Niemi, 1990). The question is not whether or not coal ash contains contaminants, because there is no doubt it contains many, but the question is whether or not such contaminants present an unreasonable risk to the GSL ecosystem and the commercial viability of other resources extracted from GSL. An informed inquiry of the inherent risk in locating massive quantities of coal ash in the PPL site unequivocally indicates that there is a pronounced and indisputable risk of contaminants moving from the PPL site and into GSL. Based on the quantity of coal ash proposed for disposal and the history of leakage from such sites, permitting the PPL site for Class V waste poses an unreasonable and unacceptable risk to the GSL ecosystem.

As a means of specifically evaluating the potential consequences of leakage from a Class V landfill into GSL, I evaluate each common contaminant found in coal ash in terms of the established range of concentrations found in coal ash, its toxicological properties, impacts on aquatic biota and consequences for the valuable invertebrate *Artemia franciscana*. Particular emphasis is placed on impacts on *Artemia* because they form the foundation for the food web of GSL and exert a pronounced influence on water quality, nutrient cycling, energy production, carbon flow, and ecological integrity for the entire GSL ecosystem. Any harm to the *Artemia* population is vectored throughout the ecosystem, potentially causing entire populations of dependent vertebrate taxa to collapse.

Table 1.0 summarizes multiple studies that have evaluated the concentration of contaminants in coal ash. For each priority contaminant of concern, the range of identified concentrations is reported. Major elements found in coal ash include: aluminum, calcium, iron, magnesium, phosphorous, potassium, sodium, silicon, sulphur, and titanium. The trace elements provide a longer list and include some of the most toxic components (in **bold**) of coal ash including: antimony, **arsenic**, barium, beryllium, boron, **cadmium**, chromium, cesium, cobalt, **copper,** fluorine**, lead**, manganese, **mercury**, molybdenum, nickel, rubidium, **selenium**, thallium, thorium, tin, tungsten, uranium, vanadium and **zinc** (Rowe, 2014; Vassilev and Menendez, 2005). Although these components are identified in coal ash, a number of them are chemically bound quite tightly to other components and are not readily mobilized into the environment. Among the factors that exert a dominant effect is pH, with low pH favoring the release into the surroundings while elevated pH mitigates to some extent the release of specific elements (Izquierdo and Querol, 2012). Higher pH tends to stabilize toxic elements such as cadmium, lead, mercury and zinc whereas under similar conditions arsenic, chromium, molybdenum, and selenium are favorably released. Among the contaminants found in coal ash, mercury, arsenic, lead, copper, cadmium and selenium are of substantial concern for GSL biota due to the fact that these are already present in GSL water, albeit at concentrations below critical levels causing adverse ecological impacts. Increased concern for these particular harmful elements stems from the fact that additional inputs of presently occurring contaminants in GSL can elevate levels to the point where impacts occur throughout the food web.

Table 1.0. Documented concentrations in leachate releases for priority contaminants in coal ash from a variety of coal sources (Izquierdo and Querol, 2012). Values are shown in parts per million dry weight when available. Standard abbreviations are used according to the Periodic Table of the Elements or from established scientific norms.

Symbol	**As**	**Cd**	**Cr**	**Cu**	**Pb**	**Hg**	**Mo**	**Se**	**Zn**
Element	**(Arsenic)**	**(Cadmium)**	**(Chromium)**	**(Copper)**	**(Lead)**	**(Mercury)**	**(Molybdenum)**	**(Selenium)**	**(Zinc)**
Concentration of Releases	0.30 to 3.00 mg/kg	0.01 to 0.10 mg/kg	Around 5.0 mg/kg	Around 0.05 mg/kg	Around 0.4 mg/kg	<0.2 mg/kg	2.0 to 30.0 mg/kg	>1.0 mg/kg	0.02 to 2.0 mg/kg
Leachability	0.3% to 20.0%	3.0% to 9.0%	1.0% to 4.0%	2.0% to 5.0%	<1.0%	<1.0%	15% to 50%	10% to 50%	3% to 9%
Max Mobility pH[2]	pH: 7-11	pH: 4	pH: 8-12	pH: 3	pH: 1 or > 12	pH: 4 to 13 (minimal pH dependence)	pH: 7-12	pH: 10-12	pH: 2
Relative Risk[1]	Moderate to High	Moderate to High	Moderate	Moderate to high	Moderate	Moderate	Low	High	Moderate
Toxicity to Aquatic Organisms	High	High	Moderate	High	High	High	Low	Moderate	Low
Comment	Ca plays significant role in leachability	Cl shifts solubility curve to higher pH	Cr^{+6} is carcinogenic Cr^{+3} is most common form	Highest mobility occurs at non-environ-mental pH of 1	Ca influences toxicity of Pb. Pb leaching is independent of concentration and is a low environmental risk	Extremely volatile element. Most is lost during combustion of coal. Lack of correlation between concentra-tion and leachability.	Indicator of leachate mobility into ground water	Most strongly enriched element in coal.	Synergistic reaction with Cd. Solubility increases again at higher pH.

1.0 Relative Risk takes into account leachability, toxicity, and current levels already present in GSL water and sediments.

2.0 pH indicates the acidity/alkalinity of fluids in the confines of the waste facility. CCR waste generates its own unique chemical conditions when in a liquid or dry state. The resulting pH will influence the leachability of contaminants from the CCR into a mobile aqueous phase, thereby directly affecting the potential for mobilization of contaminants and discharge into surrounding matrices.

Financial Costs of CCR Waste Facility Failures

There is a tragic legacy of more than 45 years of documented contamination from coal combustion residue (CCR) storage and waste facilities that have caused major impacts on wildlife, human health, drinking water, lakes, streams and other waterbodies (Lemly, 2014). The economic cost of damage from coal ash waste ranges from tens to hundreds of millions on an annual basis for each site and extends into the billions of dollars for some sites (Gottleib, Gilbert and Evans, 2010; Lemly, 2015). For example, at the Kingston Fossil Plant in Tennessee, the spillage of more than a billion gallons of CCR contaminated water resulted in aquatic and terrestrial damage that will cost at least $US 1.2 billion in mitigation and cleanup efforts (Deonarine et al., 2014; TVA, 2009). The Duke Energy coal ash waste site leaked into the Dan River, North Carolina and caused a “short-run” cost estimate of $US 300 million (Lemly, 2015). Duke Energy has filed a projected cost expense of $US 3.5 billion for the remediation costs and further estimates were in the range of $US 2-10 billion. In Lake Sutton, North Carolina the financial impact on the fisheries due to CCR contamination cost an accumulated $US 217 million in lost revenue (Lemly, 2014). The 6-month breakdown of costs from the Dan River leakage included: $US 113.4 million for ecological damage, $US 31.5 million for recreational impacts, $US 75.6 million for human health and consumptive use damage, to name a few impacted economic categories and associated costs. Damage to other lakes in North Carolina from CCR waste impacted fisheries and other wildlife collectively cost $US 2.944 billion (Lemly, 2014; Lemly and Skorupa, 2012). It is especially noteworthy that this cost of nearly $US 3 billion was only determined for 5% of the surface waters contaminated by CCR waste in just one state. The unknown costs among other facilities are likely extraordinary. When CCR sites are closed down there are associated costs in the range of $US 100,000 to 300,000 per acre (Connors, 2015). This means that future closure costs of the proposed 2,000-acre PPL site would cost a minimum of $US 200 to 600 million in today's dollars. These are but a few of the many examples of the astronomical costs of failures associated with CCR waste facilities.

There is abundant evidence of the high probability of failure and massive costs that result from leakage from a CCR waste site that are far beyond any perceived benefit to Box Elder County or Northern Utah that would justify the PPL to receive such high-risk waste. There is no sufficient economic, or other reason to allow such waste to be stored at the PPL site. The Utah Department of Environmental Quality has pointed out that the state currently has a sufficient number of Class V waste facilities and is not in any need of more of this type of landfill. Many more jobs would be at risk than the minimal number of jobs produced. The evidence is clear: CCR waste sites typically fail and cause irreparable damage to ecosystems. Such a failure at PPL would dramatically impact industries that depend on GSL resources and the integrity of the GSL ecosystem. The possibility is so real that one can confidently predict that if PPR is awarded a Class V waste license and receive CCR wastes, then other GSL dependent businesses may eventually be ruined, thousands of jobs lost and Northern Utah residents and citizens of the USA left with a destroyed GSL ecosystem and billions in cleanup expenses.

History of Contaminants Leaking from CCR Waste Facilities and the Inability of the USEPA to Issue Necessary Regulations to Protect Water Resources, Humans, Wildlife and the Environment

Failures of CCR waste storage facilities are common worldwide and within the U.S. and are attributable to factors such as overtopping, slope instability, earthquakes, structural and foundation failures, seepage, erosion, and other factors (Blight and Fourie, 2005). These authors have documented reports of CCR waste facilities from around the world that caused environmental devastation, destruction of communities, property loss, and extraordinary accounts of human mortality. Among their conclusions is the opinion that all of the costs of waste disposal operations, and their short or long-term impacts, must be realistically assessed and borne by the revenue producing operation that created the waste. Such costs should not be passed on to innocent victims of poor planning and improper storage of CCR wastes and it must be acknowledged that these costs can be in the *billions of dollars*. Across the U.S. there are examples of major coal ash contamination events about every 3 years, and include such disasters as the 2011 Wisconsin CCR waste site that resulted in the collapse of an existing bluff that released contaminated coal ash into the drinking water of 40 million people (Korb, 2012). CCR disposal sites have become "attractive nuisances" and "ecological traps" for wildlife as they serve to attract various wildlife and then expose those wildlife to contaminants (Rowe, 2014). Accounts of damage from CCR storage facilities are readily apparent in the USA and must be taken into consideration in terms of the probability of occurrence, the severity of damage and the astronomical cost of remediation for such catastrophic events. Allowing for a Class V waste facility adjacent to GSL ignores the evidence of harm to waterbodies from such waste facilities and favors the economic enhancement of a select few while putting the entire GSL ecosystem, as well as thousands of jobs, multiple industries, and the health of Northern Utah residents in jeopardy.

In a 2007 study reported by the USEPA, the results of a 5-year investigation of 85 potential cases of damage from CCR sites revealed that 24 were definitively caused by leakage of contaminant laden CCRs and that another 43 cases were highly likely to also be linked to contaminant leakage from CCR disposal sites (USEPA, 2007). An additional 31 sites in 14 other states were identified by the Environmental Integrity Project (EIP, 2010) and were shown to be linked to contamination from CCR waste sites. An April 2012 report by the Environmental Integrity Project added to this number 116 sites from 49 coal powered power plants that were found to have ponds or landfills that contaminated surrounding waterbodies causing exceedances in maximum groundwater standards for various contaminants (EIP, 2012). Leakage from sites (such as the city of Pines, Indiana) has been so severe that the contaminated surroundings had to be declared as a superfund site (Maloney, 2014). In a 2009 USEPA assessment of coal ash sites around the U.S., 676 sites were evaluated and among these 559 were designated as a high or significant hazard. Lemly and Skorupa (2012) examined 22 contaminated sites, and on the basis of their findings reported to the USEPA administration on the urgent need for coal ash disposal regulations to prevent such contamination from happening in the future. Their investigation showed a cost of over $2 billion in cleanup expenses, most of which were not successful at resolving the damage caused by leakage from coal ash storage facilities. They strongly lobbied the USEPA to end all surface disposal of coal ash as the only viable means to forestall further damage from such sites. In their 2012 report, Lemly and Skorupa conclude their evaluation of coal combustion waste sites with the definitive statement that "…*surface impoundments pose unacceptably high ecological risks regardless of location or design.*" Regardless of location or design, it emphasized none of the sites evaluated were without failure or imminent risk of failure. Another USEPA study determined that of 1161 coal ash sites, 563 were in a high-risk category of leakage due to poor construction and other risk factors (USEPA 2012). Taking into consideration the concerns about coal ash waste disposal, the USEPA was supposed to issue new guidelines by 2010. The 2014 ruling by the USEPA on the issue of CCR waste designation determined that it would only be regulated under Subtitle D of the Resource Conservation and Recovery Act (RCRA) instead of under Subtitle C. Subtitle C would have classified it as hazardous waste with the associated requirements for proper disposal. Instead, under pressure from a powerful energy lobby, the classification of CCR under Subtitle D only categorizes the waste as "non-hazardous." Because of this series of events, there remains an enormous gap between unequivocal scientific evidence of harm to humans, wildlife, the environment and water quality caused by CCR disposal and the implementation of effective regulations to cease this highly destructive practice. Without proper regulations in place the damage will continue, with the only foreseeable remedy resting in the hands of state regulators and legal advocates to prevent permits from being issued. It is for these and other reasons that the Director of the Division of Waste Management and Radiation Control (DWMRC) should deny a Class V permit to PPR.

Physical Harm from CCR Particulate Matter

Not only are the toxins in CCRs damaging to the biochemistry of aquatic organisms but the physical "blanketing and smothering" of the benthos by CCR particulate matter can cause as much, or even more damage to an ecosystem than the contaminants (Lemly, 2015). For example, a spill of only 39,000 tons of CCR into a waterbody has the potential to cover 52,000 acres of substrate. Just two inches of CCR covering the benthos (lake-bottom organisms and related substrate) of a waterbody effectively suffocates the benthic biota and destroys the water/substrate interface necessary for essential nutrient cycling dynamics. This is a sobering reality when one considers that the PPL may receive 120 million tons, or more, of CCR for their waste site and the surface area of Gilbert Bay is only about 405,000 acres at the current elevation of 4193.7 feet above sea level (Baskin, 2005). The critical microbialite surface area will be substantially smaller than the surface area of Gilbert Bay at a given elevation thereby increasing the risk of harm from a spill of even a minute portion of the proposed quantity of CCR to be stored at the PPL site. Based on these values, and the desire of PPL operators to collect and store over a hundred million tons of CCR waste, it would only require a fraction (less than 2%) of the waste facility to fail to disrupt the entire benthic environment of Gilbert Bay.

In the GSL, the benthic system is characterized by highly productive microbialite structures that have been shown to be one of the major contributors to nutrient cycling and limnological processes that are essential for biotic growth in the GSL. The critical role that the substrate structure of GSL serves for the maintenance of the ecosystem is only recently being appreciated. Detailed bathymetric research done by the USGS under the guidance of Dr. Rob Baskin (Baskin, 2005), excellent limnological research of Dr. Wayne Wurtsbaugh (Wurtsbaugh, Gardberg and Izdepski, 2011; Wurtsbaugh and Berry 1990) and Dr. Gary Belovsky (2011), have shed light into the incredibly important role that benthic structures and biological diversity of benthic organisms serve for the GSL. Further detailed research by Dr. William Johnson and his team of researchers at the University of Utah Geological department has illustrated the complex nutrient and trace element cycling dynamics that take place at the substrate-water interface, and the role that this plays in terms of internal cycling of nutrients for GSL (Johnson et al., 2015; Dicataldo, Johnson and Naftz, 2009; Oliver et al., 2009; Diaz et al., 2009). All of this research supports one important and irrefutable fact—the GSL benthic structure and organism abundance and diversity is absolutely essential for the GSL ecosystem to function effectively. Inundation by CCR particulate matter caused by failure of the PPL would result in "blanketing and smothering" of this essential benthic system causing decimation of these critical ecological processes of GSL. This alone could destroy the GSL ecosystem independent of the many known toxins in CCR.

GSLBSC Contributes Significantly to the Economy of Utah, Provides Jobs, Supports the Utah Division of Wildlife Resources (UDWR) Research and Provides the Nutritional Foundation for the Global Aquaculture Industry.

GSL provides $1.3 billion in total economic output annually and contributes 7,706 jobs to the State of Utah. The brine shrimp industry both directly and indirectly provides $57 million of this economic output. The brine shrimp industry further contributes to and supports the Division of Wildlife Resources (DWR) and the GSL Ecosystem Project and is an active participant in the prudent and sustainable management of GSL. 99% of brine shrimp harvested on GSL are exported and the *Artemia* cysts produced by the GSL brine shrimp industry provide one of the most important ingredients for the global aquaculture industry and, therefore, provides food and protein to the world. By serving the aquaculture industry GSLBSC must produce a product that complies with dietary standards for the production of finfish and crustaceans suitable for human consumption. Any form of additional contamination of GSL waters with the many known contaminants in CCR can move through the food web and be partitioned into the tissue of brine shrimp. Contaminants such as arsenic, cadmium, copper, lead, mercury, selenium, cobalt and chromium that are found in CCR may accumulate in brine shrimp tissue rendering the product unusable as a dietary component for the aquaculture industry. Rigorous nutritional feed standards are established in countries where aquaculture production takes place, and routine testing of feed products is a normal requirement to meet health-related standards. Critical tissue values in feeds or aquaculture products produced for human consumption such as mercury, selenium, and lead (i.e., contaminants that are commonly found in CCR) can be as low as 1.0 to 0.5 ppm (tissue dry weight) (European Commission, 2002; World Health Organization, 1989). Exceedances in these levels disallows the product to enter aquaculture production. Because of this close scrutiny to the quality of GSL brine shrimp cysts by the aquaculture industry and by importing countries, any discharge, whether by aerosolized dust, leaching, leakage or catastrophic flooding, dumping or inundation of contaminant containing CCR into GSL would place the globally important GSL brine shrimp cysts at risk of failing to meet feed product standards.

Toxicity and Risk from Trace Elements in Coal Ash to *Artemia* and Other Aquatic Biota

CCR is laden with toxic trace elements that impact biological systems and that can have lasting consequences for contaminated water bodies. Among the many problematic constituents there are a few that are especially of concern for the GSL and its associated biota. The contaminants of main concern are listed below and their impacts on aquatic systems are briefly highlighted with special reference to impacts on GSL *Artemia*. The elements presented are just a partial list of potential contaminants found in Class V waste and especially in CCRs. We have concern for all of the toxic constituents of Class V waste that could be received by PPL if a permit for such waste is secured by the PPR. However, for the purpose of this document we are providing a cursory view of the kind of toxic potential that CCR waste has on the aquatic biota of GSL and on the brine shrimp population. Certainly, many more details for each toxin could be provided but it is beyond the scope of these initial comments. In addition to metal toxins that can leach out of CCR sites, other chemicals, such as ammonia, can leach from waste sites in such high concentrations that they can be toxic to aquatic biota such as *Artemia* (Svensson et al., 2005). Suffice to say there is a long list of toxins that have a high probability of entering the GSL ecosystem due to the myriad possible reasons for failures in the disposal, engineering or structures that could occur to the PPL facility should millions of tons of Class V waste be stored in such close proximity to GSL. Leakage, leaching, dust deposition, fugitive waste, flooding, earthquakes, human error or any other cause of an accidental discharge could cause widespread and prolonged damage to the valuable and productive GSL ecosystem.

ARSENIC

Arsenic Impacts on Biological Systems

Arsenic causes both acute and chronic health impacts including the promotion of cancer. Cancer promoting mechanisms associated with arsenic include genotoxicity, abnormal methylation or alteration of DNA, oxidative stress, enhanced cellular proliferation, proto-oncogene stimulation, and tumor suppressor gene (Hughes et al., 2011; Hughes, 2002). Arsenic is known to cause birth defects, cardiovascular damage and kidney disorders among humans and other vertebrates. Citing USEPA statistics, Earthjustice (2009) reports that people living near coal ash disposal sites have a 1/50 (2000/100,000) chance of developing cancer from the arsenic in coal ash. This is in contrast to the nationwide prevalence rate for *all cancers among all ages* which is 454/100,000 (National Cancer Institute, 2017). Among this total, a small fraction is attributable to arsenic within the general population, which means that living in close proximity to a coal ash waste site dramatically increases one's cancer risk. The toxicity of arsenic to aquatic organisms includes oxidative stress, protein alteration, enzymatic, genetic, and immune system dysfunction (Kumari et al., 2016). Elevated concentrations of arsenic in liver and kidney cause metabolic disruptions and increasing disease risk.

The toxicity of arsenic is related to its molecular form--inorganic forms are more toxic than the organic forms. Arsenic contamination in biota occurs through the food web and via direct dermal exposure (Bissen 2003; Smedley, 2002). Arsenic in marine invertebrates and fish is generally in the range of 1-100 ug/g dry weight. The ocean has a level of 1.7 ug/L (Williams, 2001). The USEPA limit in water for human consumption is: 10 ug/L (USEPA, 2017b). Characteristic environmental and tissue forms of arsenic are: arsenate, arsenite, and multiple organoarsenic forms. Arsenic toxicity depends on other factors such as water quality, type of organism and co-exposure to other stressors (Spehar et al., 1980). Aquatic invertebrates have been shown to accumulate arsenic above water concentrations and to transmit elevated levels of arsenic through the food web.

Arsenic Toxicity to GSL *Artemia*

Brix, Cardwell, and Adams (2002) evaluated the chronic toxicity of arsenic to the Great Salt Lake brine shrimp. The GSL water used for the study had an existing concentration of 0.24 mg/L. Arsenic concentrations were adjusted in the test water and growth, reproduction and survival responses were evaluated. Their results indicate that the No Observed Effect Concentration (NOEC) was 8 mg/L. The Lowest Observed Effect Concentration was 15 mg/L for survival and 56 mg/L for growth and reproductive effects. Survival was the most sensitive endpoint. These authors opined that the F1 generation was significantly more tolerant to arsenic that the P1 generation. The authors conclude that risk to GSL *Artemia* from current levels of arsenic in the GSL is low in terms of sediment concentrations. Although these authors infer that under existing conditions the risk to *Artemia* from arsenic is low, further contamination that elevates arsenic levels above impact thresholds can pose a significant risk to *Artemia* especially if there is in fact a trend of increasing arsenic in GSL sediments. Arsenic levels in coal ash can be between 8 – 1385 ug/g depending on the coal ash source, thus containing potentially substantial concentrations of arsenic that could enter the GSL ecosystem (Rowe et al., 2002). Wurtsbaugh et al. (2012) report that arsenic levels have been increasing in recent years due to anthropogenic and natural sources and have reached 30-50 ug/g in sediments. It is not clear what the mechanisms are that would be resulting in an increase in arsenic in GSL, but it is a matter that requires continued monitoring. The possible introduction of arsenic from CCR waste would constitute a dangerous contribution to the mass balance of arsenic in the GSL ecosystem.

CADMIUM

Cadmium Impacts on Biological Systems

There are no known benefits of cadmium and it is toxic to plants, fish, mammals, invertebrates and microorganisms (Eisler, 2000). Among humans it causes cancer, birth defects and is genotoxic (Nordberg et al. 2009). In most organisms, cadmium causes vascular injuries, blood alterations, hepatic damage, and neurological impacts. Among the metals and metalloids found in CCR, Cd may be one of the most toxic. Cadmium bioaccumulates and biomagnifies in the food web and has bioconcentration factors of up to 18,000 for invertebrates and 4,190 for aquatic organisms (Levit, 2010). Cadmium is typically low in GSL water with a reported concentration of <2 ug/L (Brix et al., 2006), however, due to its propensity to bioaccumulate the brine shrimp body burden may be between 0.1 and 0.5 ug/g dw, suggesting that even at low exposure levels cadmium may accumulate and magnify throughout the food web. In aquatic organisms, cadmium has been linked to higher mortality rates, slower growth rates, reproductive impairment, and population level alterations (Mebane et al. 2012; McGeer et al, 2000; Chapman, 1978a; Chapman and Stevens, 1978b). Hollis et al. (2000) have documented a protective effect of calcium against cadmium. Stephens and Gillespie (1976) have reported on calcium levels (840 mg/L) in GSL which may confer some reduction in cadmium impacts on *Artemia*. The federal fresh water chronic criteria for cadmium for fresh water is quite low at 0.72 ug/L and 7.9 ug/L for estuarine/marine waters (USEPA, 2016).

Cadmium Toxicity to GSL *Artemia*

In a comparative study of various *Artemia* species, Sarabia et al. (2001) found that *Artemia franciscana* (the species found in GSL) was the most sensitive to cadmium and exhibited the highest mortality rate. Susceptibility to cadmium was strain specific and A. *franciscana* was as much as two times as sensitive as other *Artemia* taxa. Cadmium toxicity is influenced by salinity because the bioavailability of cadmium is altered by chloride levels by the formation of weak Cd-Cl complexes, and these complexes influence uptake by brine shrimp (Blust, Kockelbergh and Baillieul, 1992). The ability of *Artemia* to tolerate cadmium may also be a function of metallothionein (low molecular weight proteins involved in metal detoxification) induction which is higher than other crustaceans (Martinez et al., 1999; Del Ramo et al., 1993). The higher metallothionein levels in *Artemia* suggest that exposure to other contaminants or stressors that may inhibit metallothionein release could exacerbate toxic risk to *Artemia* (Klaassen et al, 2009). Gebhardt (1976) determined a 320 hour LC50 for cadmium of 3.3 mg/L. This is in contrast to other studies that showed LC50 values between 93.3 and 280 mg/L (Sarabia et al., 2006). One should consider that lethality is just one endpoint of cadmium's toxicity to *Artemia* and it may not be the most sensitive indicator. Sublethal behavioral, reproductive, immunological, or metabolic impacts can be more sensitive alterations than lethality and many of these have not been tested on brine shrimp. Another consideration is that toxins can be synergistic, or antagonistic, in their impact on *Artemia* and other aquatic organisms. In a study on various forms of organotin on *Artemia*, Hadjispyrou, et al. (2002) determined that cadmium and organotin compounds interacted synergistically and increased mortality rates. Tin is another component often found in CCR waste that could enhance toxicity when combined with other metals.

In a study of the developmental impacts of cadmium on *Artemia,* Rafiee et al. (1986) determined that cadmium disrupts development in a dose-dependent manner and that effects could be observed with as little as 0.1 to 1.0 uM concentration. Above the 1.0 uM concentration, development continues but emergence is severely delayed or arrested. At 10.0 uM cadmium, abnormal development was observed. A group of researchers investigated other toxic endpoints for cadmium in brine shrimp and found that hatchability of brine shrimp eggs is much more sensitive to cadmium than is mortality, and that the EC50 for Cd using the hatching endpoint was a mere 7 ug/L (MacRae and Pandey, 1991; Bagshaw et al., 1986). The values obtained by these authors indicates a highly sensitive outcome to a metal in an aqueous environment. Later research by Brix et al. (2006), found that tolerance for Cd in GSL water, which has a high dissolved solute level, may confer a protective influence on the hatchability of cysts and that under natural conditions of GSL the toxic threshold is higher than indicated in previous toxicity tests. The broad range of results from these various groups indicates that under specific conditions brine shrimp cysts can be highly sensitive to cadmium.

COPPER

Although copper is a component of enzyme systems in living organisms and is therefore essential for life, it is considered a hazardous metal and is especially toxic to aquatic organisms. The USEPA water quality criteria for copper to protect aquatic life is 2.3 ug/L (USEPA, 2007). The revised salt water limit for the protection of aquatic organisms and their uses is 3.1 ug/L for the 24-hour dissolved copper concentration. The pH of water exerts an influence on the toxicity of copper in which lakes with elevated pH contain copper in a more bioavailable and toxic form (Guthrie and Perry, 1980). Copper absorption is associated with mucous membranes in gills of fish and increased pH of a waterbody increases the absorption of copper by fish. Copper toxicity causes damage to plasma membranes of cells and can cause adverse effects on survival, growth, reproduction, brain function, enzyme activity, blood chemistry and altered metabolism. Copper has been shown to be lethal to aquatic invertebrates and finfish at low concentrations. For example, the LC50 value for Daphnia magna was determined to be 9.1 ug/L in a 24-hour toxicity test (Nebeker, 1986). Toxicity values for other Cladocerans are in the range of 2.83 to 19.36 ug/L, while fish values are found in the range of 5.92 to 27.77 ug/L (Seim et al. 2010; Koivisto et al., 1992; Belanger et al. 1989 Chapman, 1978; Winner et al., 1977; Winner and Farrell, 1976). Chronic values for invertebrate species range from 2.83 to 34.6 ug/L and 5.0 to 60.4 ug/L for fish. Extensive studies of the toxicity of copper are available in the scientific literature detailing hundreds of investigations on the toxicity of copper to aquatic organisms. There is no doubt that copper exerts a toxic effect on aquatic organisms when exposure exceeds the minimal biochemical requirements. It is also important to recognize the properties of copper as an algicide and that due to its toxicity to algae it can greatly diminish primary production in a waterbody. Any decrease or impact on primary production in an ecosystem can be vectored throughout the food web causing multi-trophic level impacts (Levy, 2007; Real et al., 2003; Wright and Welbourn, 2002).

Copper Toxicity to GSL *Artemia*

Total copper levels in the GSL are usually between 3.7 to 10.2 ug/L (Brix et al., 2006). This is above the USEPA standards for dissolved copper associated with the protection of aquatic life and is close to some of the toxic thresholds identified in acute and chronic testing of copper impacts on *Artemia*. Notwithstanding the proximity to acceptable limits for copper in GSL, it should be kept in mind that the salt matrix and other dissolved elements influence the outcome of the calculations of concentration and toxicity. In a study on the developmental effects of copper, MacRae and Pandey (1991) determined that copper had similar impacts as lead and that at concentrations of 0.01 uM (0.64 ug/L) it reduced the rate and development of *Artemia* nauplii. These authors recognized that emerging nauplii are much more sensitive to metal toxicants than are dormant embryos or older life stages—thus reinforcing the need for conservative interpretation of adult toxicity studies. In studies of hatchability of brine shrimp cysts Brix et al. (2006), site previous research that determined an EC50 of 5 ug/L for copper. Brix et al. (2006) performed a series of similar tests using a variety of prepared saline solutions and found similar, albeit slightly higher, results in which copper concentrations ranging from 12 to 28 ug/L caused hatching impairment. When these authors used GSL water the EC50 increased to 50 ug/L and was interpreted to be a result of dissolved organic carbon loads in the GSL water.

Concern about Class V waste and copper toxicity to brine shrimp and other GSL arises from the fact that copper is already approaching a critical toxic threshold for brine shrimp and it would only require a small inoculation of copper laden waste or leaching of copper from waste sites to push the copper threshold over the limit. An example of copper input from CCR waste is found in the Dan River CCR spill which resulted in increased copper levels in ambient water of 7.5 to 46 ug/L over a distance of 10 miles (USEPA, 2014). The Dan River example indicates the extent to which discharge from a CCR site can contaminate vast areas and volumes of receiving waters. CCR can have copper concentration of 45-1452 ug/g dry weight and can therefore be a significant source of copper to a waterbody if CCR waste is discharged or leaks into water systems (Rowe et al., 2002). This known content of copper in CCR coupled with the current concentration of copper in GSL water, and the sensitivity of *Artemia* to copper exposure, renders any additional copper introduction into GSL a very high risk of causing damage to the *Artemia* population.

MERCURY

Mercury Impacts on Biological Systems

Mercury causes a host of physiological, reproductive, and biochemical impacts on mammals, fish, and invertebrates. Mercury has no known beneficial biochemical function in animals; rather it is a highly potent toxin. Mercury is a mutagen, teratogen, carcinogen, and neurotoxin, and causes embryocidal, cytochemical, and histopathological effects (Evers et al., 2005; Seewagen, 2003; Siegel and Siegel, 1997; Spalding et al., 1994; Eisler, 1987). The most consistent and pronounced impacts are on the central nervous system. Mercury can cause sublethal behavioral modifications that in turn lead to reduced reproductive output and fitness (Burgess and Meyer, 2008). Methylmercury (the organo/methylated form of mercury) has a steep dose response relationship (in other words a small increase in dose results in a substantial increase in the impact) attributable to the bioavailability of this form. Methylmercury is always more toxic than inorganic mercury compounds. The formation of methylmercury from its elemental form occurs more readily in anoxic environments, especially in the presence of reduced sulfur species, as is the case when the monimolimnion (deep brine layer) is present in the deeper sections of a hydrochemically stratified Gilbert Bay (Boyd et al., 2017; Naftz et al., 2008; Weiner et al., 2003). In the environment, mercury partitions itself primarily in the soils and sediments over water or atmospheric levels. In living organisms, mercury accumulates primarily in the kidneys but is found in elevated levels in the liver, brain, other internal organs, as well as the blood—bound to erythrocytes (red blood cells) (Liu, Goyer and Waalkes, 2008). Mercury readily bioaccumulates or bioconcentrates (measured with exposure to the contaminant in water only) in higher trophic levels. Some representative examples of bioconcentration factors (BCFs) range from 28,300 to 238,000 for methylmercury and from 2,300 to 68,600 for inorganic mercury presented to aquatic organisms in the water. In marine copepods, BCFs of 14,360 for inorganic mercury and 179,200 for methylmercury were observed. The USEPA shows bioaccumulation factors (BAFs) of 27,900 (trophic level 3) and 140,000 (trophic level 4) based on Great Lakes research (USEPA, 1998). Bioaccumulation of mercury in the GSL food web has been identified and documented in a variety of studies (Wurtsbaugh, Gardberg and Izdepski, 2011, Conover and Vest, 2009; Vest et al. 2009). Mercury is persistent, not readily degraded by biochemical detoxification mechanisms therefore tends to bioaccumulate in food webs. An accumulation of scientific evidence indicates that wetlands may be particularly sensitive to mercury impacts (Hurley et al., 1995; St. Louis, 1994). Mercury has been tracked through the GSL food web and the presence of mercury in resident and migratory birds that use the GSL ecosystem for foraging and reproduction is a matter of tremendous concern because mercury is quite possibly the most harmful contaminant to birds (Conover and Vest, 2009; Heinz and Hoffman, 1998; Eisler, 1987; Hoffman and Moore, 1979).

The toxic potential of mercury and its ability to move through the food web must be taken extremely seriously because it can cause widespread and lasting damage to ecosystems and human health. Risks associated with mercury input into GSL are elevated due to the cyclical meromictic condition (i.e., stratification) of Gilbert Bay and the role that this plays in providing a unique microenvironment suitable for converting mercury into its more toxic and bioavailable form--methylmercury. Knowing that these conditions exist in GSL, and that the GSL ecosystem is perhaps the most important location for migratory birds in North America, renders the idea of situating a landfill in close proximity to GSL that may receive millions of tons of mercury containing waste an unfathomable mistake---which may not be resolvable.

Mercury Toxicity to GSL *Artemia*

Terminal lakes such as GSL may accumulate persistent heavy metals such as mercury, and mercury is found in low concentration in the oxic waters of GSL. A 5-year study of atmospheric, water and brine shrimp mercury concentration by Peterson and Gustin (2008) showed epilimnetic water concentrations of 3.6 ng/L for total mercury and 0.93 ng/L for methylmercury. These authors documented that total mercury concentrations in GSL surface waters was consistent throughout the year and, compared to other natural waters, were not significantly elevated. Wurtsbaugh, Gardberg and Izadepski (2011) reported 5.0 and 1.2 ng/L for total mercury and methylmercury respectively. To put these levels into perspective, waterbodies known to be contaminated by industrial waste are often in the range of 10-40 ng/L and those contaminated by mining waste can be as high as 1000 ng/L (Weiner et al, 2003). Levels measured for GSL that are in excess of expected values for a terminal lake have periodically been identified in the anoxic monimolimnion (deep brine layer) of Gilbert Bay that occasionally develops due to exchange of heavy brines from Gunnison Bay (Naftz, et al., 2008). Although mercury in the monimolimnion was elevated above the oxic (upper) layer of Gilbert Bay, this deep layer was representative of only a fraction of the total lake volume. There is much speculation and some evidence that mercury in microbialite dependent biota may be conveyed through upper trophic levels (Wurtsbaugh, Gardberg and Izadepski, 2011). However, investigations of exposure of brine shrimp to an anoxic layer contaminated with mercury did not in fact show greater accumulation of mercury in the brine shrimp, and instead showed a decrease in the body burden attributed to detrital dilution of the mercury (Jones and Wurtsbaugh, 2014). There currently exists some exposure potential for GSL *Artemia* to mercury and evidence of tissue accumulation in adult brine shrimp (Naftz et al, 2009). GSLBSC has documented mercury in brine shrimp biomass and cysts over the past decade and has tracked a within year pattern of accumulation in adults but no temporal pattern in cysts. This increase in concentration among adult brine shrimp was also observed by Peterson and Gustin (2008). In spite of a within year trend of increasing *Artemia* tissue mercury concentration, the long-term GSLBSC data on mercury in *Artemia* cysts do not show evidence of an accumulating pattern over multiple years. The level in GSL brine shrimp cysts remains below threshold levels for the aquaculture industry and below levels known to cause impairment. However, any increase in mercury in GSL could elevate the water concentration to above threshold limits causing widespread harm to the *Artemia* population, and could impact its use for the global aquaculture industry thus ruining the GSL brine shrimp industry. Mercury values in cysts taken from the open water of the GSL are well below the EPA dietary standard for methylmercury (0.3 mg/kg ww) and are also below the European Union Directive on heavy metals in aquaculture feed (0.5 mg/kg dw). Because mercury is one of the major concerns regarding GSL, and is a known contaminant in CCR, it is of grave concern that disposal of CCR at the PPL waste site could eventually lead to increased contamination of GSL waters with mercury.

Mercury impacts on brine shrimp have been studied and have provided a variety of outcomes. Developmental delays occur in *Artemia* at levels as low as 0.01 micromoles per liter. At higher concentration, mercury can have more dramatic effects such as inhibition of development at emergence and hatching stages (Go et al., 1990). In the presence of 0.1 micromoles of diphenylmercury, rates of emergence and hatching are adversely affected (Pandey and MacRae, 1991). The LC50 for *Artemia* exposed to mercuric chloride is 27.1 µg/l and the standard for total mercury in freshwater by the USEPA is 12 ng/L (USEPA, 1985). Cunnigham and Grosch (1978) exposed *Artemia* to mercuric chloride and documented significant reduction in lifespan at 0.01 ppm and a reduction in cyst viability produced by mercury exposed adults. Because of the persistence of mercury in biological systems, and the fact that there are presently low levels of mercury in GSL water and elevated levels in sediments of GSL, any further inputs of mercury into waters or sediments of GSL poses a high risk to the GSL food web. The current levels of mercury remain just below the critical point at which enhanced bioaccumulation and adverse impacts are likely to occur.

SELENIUM

Selenium Impacts on Biological Systems

Selenium is an essential element and is a component of selenoproteins. In this capacity, selenium is a required element for biochemical processes (Liu, Goyer and Waalkes, 2008). However, selenium is a well-known and increasingly problematic environmental contaminant. Selenium occurs naturally in the earth's crust and enters the environment mainly through human activity (such as mining and agricultural irrigation) (Ohlendorf, 2003). Combustion of coal is the main source of airborne selenium and selenium is found in substantial concentrations in CCR. Selenium is one of the top priority elements of concern in CCR and therefore in the proposed Class V waste that PPR desires to dispose of at the PPL site. Selenium contamination from CCR waste facilities throughout the U.S. is thoroughly documented in the scientific literature (Besser et al., 1996; Lemly, 1985; Furr et al., 1979; Cherry and Guthrie, 1978) and is brimming with accounts of extensive environmental damage. Although selenium is a required element, the toxic dose is just slightly higher than the required dietary quantities (Hilton et al., 1980). Exposure and dietary ingestion of excess amounts of selenium have caused widespread impacts on wildlife (Skorupa, 1998; Presser and Barnes, 1984; Eisler, 1985). Most of the adverse impacts were on bird reproduction and on fish development (Ohlendorf, 2003).

Normal background levels in surface waters are usually in the range of 0.1 to 0.4 ug/L and in sediment are 0.2 to 2.0 mg/kg (USDI, 1998). Selenium concentrations in GSL are 0.4 and 0.8 ug/L (UDWQ, 2008). Selenium bioaccumulates in aquatic and terrestrial food webs with the highest BAF (e.g., 1000X BAF) occurring between the aqueous phase and plants/algae or invertebrate grazers (Ohlendorf, 2003). The ability of selenium to bioaccumulate within food webs and to cause severe reproductive impairment in birds is one of the most pernicious impacts it has on wildlife. Harm to birds with long life spans is particularly notable because it only requires a 3% drop in reproductive output (e.g., caused by exposure to an environmental contaminant) in long lived species to have population level impacts (Mitro et al., 2008). Long lived species tend to have low reproductive rates which makes them more vulnerable to population level impacts. The scale of the problems associated with selenium in the environment are accumulating every year and are exacerbated by energy requirements necessary to meet the needs of a burgeoning population. The perception of selenium is changing as scientists, resource managers, wildlife advocates, and the general public recognize the linkage between contamination sources, energy demands (mainly those provided by non-renewable sources such as coal and fossil fuel), the ever-increasing presence of selenium in the environment and the widespread and severe harm it is causing (Tan et al., 2016; Lenz and Lens, 2009; Lemly, 2003). Combined with this awareness are more effective and definitive prevention and mitigation measures, such as regulatory limits on selenium in water bodies. Additionally, there is much greater scrutiny and concern regarding the immense problem of selenium in CCR waste.

Along with issues pertaining to mercury, discussions about GSL and selenium have been at the forefront of scientific, public, political, and resource management issues related to GSL over the past decade. A water quality standard for selenium was established for GSL that is a regulatory landmark—the first of its kind. The regulatory water quality criteria standard for selenium for GSL is the first water quality criteria established solely on the basis of a wildlife health criteria and not in terms of human health. A 4-year public participation effort directed by Utah DEQ/DWQ, that included a Selenium Steering Committee and distinguished science panel, convened to review the extant literature on selenium and the GSL ecosystem and to establish a water quality criteria for selenium. The outcome of this scientific effort was the establishment and implementation of a standard for selenium of 12.5 mg/kg concentration (avian egg/embryo tissue standard) (UAC R317-2-14). This standard was based on extensive scientific evidence, combined with reviews of the fate and effects of selenium in GSL and its associated biota, and was ultimately intended to be protective of the most sensitive endpoint—avian reproductive effects (Ohlendorf et al., 2009). Recognizing the effort behind this standard and the importance of such a standard, both locally and within a national framework, it is bewildering to think that it is acceptable to dispose of millions of tons of CCR waste contaminated with selenium adjacent to the shores of GSL. It requires a full suspension of disbelief to read the justification for the Class V permit submitted by PPR and to recognize that the colossal effort to protect the GSL ecosystem from selenium can so easily and utterly be disregarded or summarily dismissed.

Selenium Toxicity to GSL *Artemia*

The USEPA chronic criterion of 5 ug/L and the acute criterion of 20 ug/L are intended to be protective of aquatic life. Nationwide, waterbodies often exceed these values placing aquatic organisms at risk (Canton and Derveer, 1996). Discharge of CCR waste in other locations has demonstrated that even when other constituents of CCR that enter receiving waters are not significantly elevated, selenium has stood out as one the most pronounced in terms of increases in receiving waters and in the associated biota (Rowe et al., 2002). Because of this characteristic of selenium in CCR waste discharges, and the unique situation at GSL with respect to its hemispheric importance for birds, any increase in water concentrations and in the body burden of *Artemia* is a substantial risk to the entire food web. The GSL brine shrimp and brine fly populations form the nutritional basis of the ecosystem and therefore must be protected in order to sustain critical ecosystem functions of GSL. The site-specific avian eggshell/embryo-based water quality criterion for selenium in GSL is intended to not only protect birds from harm but is assumed to be the most sensitive endpoint which should then translate into protection of other biota including brine shrimp. In a study by Brix et al. (2004) a water quality standard that is protective of aquatic birds (5 mg/kg) would translate into a site-specific water quality standard of 27 ug/L. Whereas this value is substantially higher than the USEPA chronic criterion, it was viewed as being quite safe to brine shrimp because of their higher tolerance for selenium in saline water with an appreciable sulfate concentration (e.g., 5,800 mg/L). The particular features of the tested GSL water conferred some protection against the toxic impacts of selenium on *Artemia* (Brix et al., 2004). These authors calculated a chronic selenium 96-hour LC50 that was dependent upon sulfate levels. Their findings indicated LC50 values of 1.4 and 82 mg/L selenium in the presence of 50 and 14,000 mg/L sulfate respectively. Among the GSL aquatic species studied, *Artemia* were the most sensitive to selenium and growth and reproduction were the two most sensitive endpoints. In spite of this relative sensitivity, the authors opined that a water quality standard of 27 ug/L would be protective of birds and *Artemia*.

If GSL waters are protective of brine shrimp exposed to selenium, then the relevant issue becomes transfer of selenium from *Artemia* into the food web. Selenium bioaccumulates in the food web and is known to bioconcentrate in *Artemia*. In the Brix et al. (2004) study, they found BAFs of 129 to 3,380 in regions of high selenium concentration (120 ug/L) to low selenium concentration (1 ug/L) respectively. Others report bioaccumulation factors of 100-400 times for inorganic selenium and in excess of 350,000 times for organic selenium (Tan et al., 2016). The ability of brine shrimp to bioaccumulate selenium poses a risk for predators that feed on *Artemia* and on its use as a feed source for aquaculture. Increased body burden of selenium could result in exceedances of international regulatory limits on contaminant levels in aquaculture feeds thereby shutting down the GSL brine shrimp industry.

In its own multiyear research, GSLBSC has tracked selenium in the tissue of brine shrimp and has recorded concentrations in the range of 2.0 to 8.0 mg/kg dw. These values are consistent with other scientific investigations of GSL *Artemia* and for brine fly larvae which have a reported selenium concentration of 1.2 mg/kg dw (Wurtsbaugh et al., 2011). Although no adverse impacts on the *Artemia* have been observed at these concentrations, they are not far below the avian tissue threshold of 12.5 mg/kg dw. Because tissue selenium concentration is in close proximity to the threshold in bird tissue, combined with bioaccumulation capacities of brine shrimp for selenium, any increase in the levels of selenium in the water column of GSL would be highly problematic for the biota. Additionally, it has been demonstrated that birds consuming prey containing both selenium and mercury (10 mg/kg Se and 10 mg/kg Hg in the feed items) had liver levels of selenium 1.5 times higher than if there was no mercury in the feed (Conover and Vest, 2009). This scenario is consistent with conditions in GSL—levels of mercury and selenium in water and brine shrimp that currently don't pose significant risk to other biota, but if they increase even slightly, then adverse impacts can be expected. Leaching or leakage from CCR waste sites have been known to increase selenium concentrations in receiving waterbodies and in resident biota along with adverse impacts on the biota (Lemly, 2014, Lemly and Skorupa, 2012; Rowe et al., 2002; Lemly, 1997). It is therefore, as with other toxic trace elements in CCR, again an unacceptable risk to dispose of Class V coal ash waste that contains selenium at the PPL site.

LEAD AND ZINC

Lead and Zinc Impacts on Biological Systems

Zinc and lead are two other metals that are found in CCR waste and that are known toxins to biological systems. Whereas there is no purpose of lead in biological systems, zinc is a component of enzymes and serves a biochemical role in protein structure and function. Zinc is ubiquitous in cells and has a role in growth, development and immunological function and is the second most abundant trace metal in cells—second only to iron (Roohani et al., 2013). Lead, on the other hand, is always deleterious to biological systems and lead poisonings are some of the oldest known causes of poisonings in human history (Pattee and

Pain, 2003). Lead is known to be found in the leachate from CCR waste site and can be found in solution with concentrations of 10.5 to 89.9 ug/L (Wang et al., 2008) and 1.2 to 30.0 ug/L in a separate study by Karuppiah and Gupta (1997). According to these studies, leachate concentrations are a function of pH with the highest levels occurring in more acidic solutions. Although GSL is an alkaline waterbody with a pH in excess of 8, the pH of leachate solutions is a function of the stored CCR waste and its interaction with water in the landfill or receiving pond, thus having little to do with the final waterbody in terms of initial mobilization. Lead uptake in aquatic organisms is through direct absorption and via ingestion. Among terrestrial animals it is primarily through ingestion and inhalation (Vighi, 1981; Varanasi and Gmur, 1978). Organic lead is generally more toxic than the inorganic forms and lead is bioaccumulated in organisms (Pattee and Pain, 2003). In both terrestrial and aquatic organisms, lead exerts its toxic potential via inhibition of heme producing enzymes—the result of which is impaired oxygen carrying capacity of the blood and hypoxic damage to cells and tissues throughout the body (Pattee and Pain, 2003). The LC50 for Daphnia for lead is 612 ug/L (178). The risk that lead presents to aquatic biota is a function of the hardness and the concentration of calcium in the water. Zinc and lead interact in the sense that when zinc is low in the diet the uptake of lead increases (Eisler, 2000; Morrison, et al., 1977), and when lead is elevated in the blood zinc replaces iron in heme synthesis formation—causing dysfunction of oxygen carrying capacity of the blood (Eisler, 2000). Lethality is not always the most relevant concern regarding lead exposure as it can cause behavioral modifications and metabolic distress that can render the organism highly susceptible to other stressors or disease. In research by Demayo et al. (1982) and Mason and Fitzgerald (1993), sublethal impacts on aquatic invertebrates were sufficient to eliminate entire populations. Lead causes anemia and all related physiological impacts pertaining to anemia—which are extensive. Lead causes neurological damage, deformities, excess mucous production that interferes with gas exchange in gills, hypoxia and anoxia leading to death (Aronsen, 1971). Lead is, in short, a highly toxic metal that is found in concentrations between 21 and 2120 ug/g in CCR (Rowe et al., 2002). Lead causes impacts throughout every level of contaminated food webs.

Toxicity of Lead and Zinc to GSL *Artemia*

Lead poses risks to all GSL biota including the brine shrimp. Lead bioaccumulates and has BAFs of 930 to 3630 from water concentrations to phytoplankton (Soto-Jimenez et al., 2010). BAFs above this trophic level are not as dramatic and therefore are not as responsible for lead accumulating throughout the food web though lead continues to accumulate up trophic levels of the food web. In a study of the BAF for *Artemia* consuming lead contaminated algae, Soto-Jimenez et al., (2010) found that the BAF for this particular study was <1.0. MacRae and Pandey (1990) examined the effects of lead and zinc on *Artemia*. These authors used the sensitive endpoint of emergence and hatching of *Artemia* nauplii/embryos as their toxicological endpoint. In their study they found that at very low concentrations 0.1 uM of lead, rate and extent of *Artemia* development was impaired and that zinc was less toxic than lead. These authors compared toxic results for previous tests of metals on *Artemia* and concluded that the prelarval stages are the most sensitive and a better indicator of metal impacts on *Artemia* than results for adult and juvenile stages. In a study by Gajbhiye and Hirota (1990), combinations of metals (copper, zinc, cadmium, nickel, lead, iron and manganese) were tested on *Artemia*. Their study showed that there were additive and synergistic effects among the metals on *Artemia*. The order of toxicity determined through the studies was: Pb>Cd>Cu>Ni>Zn>Fe>Mn. Lead therefore ranked at the top of the list of toxic metals to *Artemia*. In their test of lead impact on *Artemia* survival they calculated a 24-hour LC50 of 1.7 ppm with 100% mortality at 2.5 ppm concentration, whereas for zinc the LC50 is 17.8 ppm. These authors opined that larval stages of *Artemia* are more resistant than adult stages.

Zinc concentrations in GSL range from 8.0 to 14.1 ug/L (Brix et al., 2006). In a study by Bagshaw et al., 1986, and cited by Brix et al., the EC50 for zinc on *Artemia* is 28 ug/L. With an EC50 of this concentration there is reason to posit concern about impacts of zinc on *Artemia* in GSL. Brix et al., (2006) repeated the experiment using alternative test solutions that were a closer representation of GSL water and discovered the EC50 was substantially higher at 307 ug/L. The impact on brine shrimp by zinc will be dependent upon the specific microenvironment in which exposure takes place with higher risk being associated with lower levels of dissolved organic matter and solutes. These authors suggest that as long as the background concentration for most metals is below 15 ug/L then there is little concern of toxicity. Discharges of CCR waste streams into GSL that elevate metal levels above this 15 ug/L concentration would create a situation of increased threat to *Artemia* and to the biota of GSL.

SUMMARY

The information presented in this document represents but a small portion of the available scientific literature on the widespread and deleterious impacts that toxins and material entering the environment from landfills that contain Class V waste (especially CCR waste) can have on biological systems. The amount of information available is immense and the impacts detailed in those reports are staggering.

Costs of damage and remediation for contaminated sites can be in the billions of dollars and ecosystems remain impacted for decades or longer. In some instances, entire populations of resident biota are obliterated. The damage can be from the large number of metals contained in such waste or just from the physical characteristic of CCR waste that can physically smother biologically active areas or cloud water with particulate matter such that primary production shuts down entirely. Many of the contaminants in CCR waste are persistent chemicals that do not biodegrade in the environment and instead remain in the exposed system continuing to cycle and exert toxic impacts for decades.

The outcome of waste facilities disposing of CCR waste is entirely clear—the probability of failure is unacceptably high especially when the receiving location is near any waterbody.

In the case of the PPL site, allowing the PPR to receive Class V waste would be a mistake and would put the entire GSL ecosystem at risk from contamination by Class V waste. If there is the kind of discharge or leaking from the PPL site that is commonplace among CCR waste facilities, and the typical level of damage occurs, then the GSL as we know it may be ruined.

The probable adverse environmental effect of approving a Class V permit for PPR at its PPL site far outweighs any possible beneficial effect of such approval.

The benefits (e.g. goods and services that GSL provides to the residents of Northern Utah, to those families for whom this is their sole source of income, to the businesses that have carefully and steadily been built, and to the millions of dependent wildlife that simply cannot survive without GSL), could be wiped out in an instant by a disaster at the PPL waste site if it is allowed to receive millions of tons of CCR waste.

The Director should not allow a Class V permit to be awarded to PPR for their PPL site. The risks from Class I waste are already enough of a threat to the GSL ecosystem. The science is clear that any significant failure, which has a probability to occur, will result in pronounced deleterious biological effects on the brine shrimp population and the GSL biota. To increase this risk by allowing Class V waste to be disposed of at this facility is to deny the reality of the existing science and the history that has documented the incredible risks of disposing of CCR waste and the disasters they have created.

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Avian Use of Great Salt Lake
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The Great Salt Lake (GSL) and associated wetlands are nationally, internationally, and hemispherically important to a large and diverse array of migrating and breeding wetland-dependent birds. Bear River Bay (Bay) lies in the northeast arm of the GSL between the Promontory Mountains to the west and the Wasatch Mountains to the east and includes Bear River Migratory Bird Refuge (Refuge), the largest freshwater marsh complex on the GSL, and the Willard Spur. Together, these biologically productive shallow wetland habitats of the Bay support hundreds of thousands of waterbirds and shorebirds annually.

Shorebirds
The GSL ecosystem supports 1.4 million shorebirds annually representing 28 species (Paul 2010 in Utah Division of Forestry, Fire, and State Lands 2013) and is a designated Western Hemisphere Shorebird Reserve Network (WHSRN) site of hemispherical importance (https://www.whsrn.org/great-salt-lake). Shorebirds are mostly present during the non-winter months with spring migration numbers peaking in April/May and fall migration peaks occurring in August/September. Shorebirds primarily use the GSL ecosystem as a stopover site to rest and feed in the shallow, invertebrate-rich waters as they journey between breeding and wintering areas, which for some species can be very long distances, e.g., from the Arctic to the southern hemisphere. However, a number of species nest in GSL ecosystem habitats including American avocet, black-necked stilt, long-billed curlew, killdeer, snowy plover, spotted sandpiper, willet, Wilson's phalarope, and Wilson's snipe.

Easily recognizable, the American avocet and black-necked stilt are abundant breeding and migratory shorebird species in the GSL ecosystem. Spring migrating avocets begin arriving in March, while stilts generally begin arriving in April. Breeding, nesting, and brood rearing activities occur during April through July. Numbers of avocets and stilts generally peak in August as local juveniles have fledged and northern birds have arrived in preparation for the journey to wintering areas in the southern US, Mexico, Central and South America, and the Caribbean. Up to 67% of avocets and 38% of stilts in the intermountain west have been recorded on the GSL during peak fall migration periods (Intermountain West Joint Venture 2013). During the GSL Waterbird survey from 1997-2001, numbers of avocets from July-September averaged 94,000 over the 5 years, with a significant portion recorded on Bear River Bay (Paul and Manning 2002). That same survey yielded a mean of 25,500 for black-necked stilts, again with a significant portion recorded on Bear River Bay habitats.

Wilson's phalarope is an uncommon to common species in Bear River Bay during breeding and becomes more abundant during the fall migration as northern birds move in to rest and feed prior to traveling to wintering areas in South America. Up to 33% of phalaropes in the intermountain west have been recorded on GSL during peak migration periods (Intermountain West Joint

Venture 2013). The mean number of phalaropes recorded from June-August was 126,600 during the 5-year GSL Waterbird Survey (Paul and Manning 2002). A high count of nearly 350,000 was recorded in 2000. Most of these birds use more central and southern areas of the GSL e.g., Ogden Bay; however, Bear River Bay can host large numbers.

Other shorebird species that nest in or migrate into Bear River Bay habitats include killdeer, long-billed curlew, snowy plover, spotted sandpiper, willet, and Wilson's snipe. Generally, these species occur in lower numbers compared to other shorebirds such as avocets or stilts. Long-billed curlews are known to nest in the pastures and fields on the Promontory Peninsula (H Browers, pers. obs., Breeding Bird Survey https://www.pwrc.usgs.gov/BBS) and are considered a state Species of concern (Utah Division of Wildlife Resources 2017).

GSL has been considered as hosting one of the largest concentrations of breeding snowy plovers. An estimate of 10,000 plovers was reported in the early 1990s (Paton and Edwards 1992). The current Great Salt Lake breeding population is estimated to be 5500 or about 21% of the continental population. Snowy plovers are a state Vulnerable Species (Utah Wildlife Action Plan Joint Team 2015) and can be found in small numbers throughout GSL.

Great Salt Lake habitats are considered of regional importance for willets under the WHSRN criteria, i.e., hosting 20,000 birds annually or 1% (Intermountain West Joint Venture 2013b). Willets breed on the Promontory peninsula in the fields adjacent to the Bay (H Browers, pers. obs., Breeding Bird Survey https://www.pwrc.usgs.gov/BBS). Wintering areas include western, southern, and eastern US coasts, Mexico, Central and South America, and the Caribbean.

The marbled godwit is a relatively large shorebird species that nests on the prairies of the northern US and southern Canada. Winter areas include the coasts of western and southern US, Mexico, and Central and South America. GSL is considered the most important migratory stopover site for marbled godwits during both the spring and fall migrations. In spring, godwits are generally present from April-May and from July-September, though some can be linger into November. Up to 26% of godwits in the intermountain west have been recorded on GSL during peak migration periods (Intermountain West Joint Venture 2013). During the GSL waterbird survey from 1997-2001, the mean number of godwits recorded was 15,125 from July-August with the greatest concentration found in the Bear River Bay, primarily on the Refuge and the Willard Spur (Paul and Manning 2002).

The Long-billed dowitcher is a common spring and fall migrant to Bear River Bay that primarily breeds in western and northern Alaska in North America. Dowitchers winter on the Pacific, Atlantic, and Gulf coasts of the United States, throughout Mexico and into Central America. Arrival of spring migrating dowitchers generally occurs in early April and peaks in early to mid-May. The fall migration is more prolonged and extends from July through late October/early November. Larger numbers come through during the fall migration. The mean number of dowitchers recorded during the GSL waterbird survey from August through September was 14,370 with a majority being found in Bear River Bay habitats. The high count recorded in 1998 was 58,800.

Western sandpiper is a common to abundant fall migrant from western Alaska and Siberia averaging about 22,000 during July-August (Paul and Manning 2002). A high count of over 194,000 was recorded in 2000. Bear River Bay habitats, primarily on the refuge, hosted significant numbers of these migrants during the survey.

The red-necked phalarope breeds in Alaska, Canada and Greenland and winters in Mexico and Central and south America. An abundant fall migrant with as many as 240,000 was recorded in one day (Paul 1982). Though there is some use of Bear River Bay by red-necked phalaropes, the primary areas of use are Ogden and Farmington Bays (Paul and Manning 2002).

Other less common migrant shorebirds that use habitats in GSL and Bear River Bay (primarily Bear River Refuge) include greater and lesser yellowlegs, Baird's and least sandpipers. These species nest in Alaska and Canada and winter from the US coast to South America.

Waterfowl

Several million ducks, geese and swans representing about 40 species use the GSL Ecosystem annually. Population estimates from data collected during 2004-2005 yielded peak spring numbers at 1.8 million occurring in March 15-31 (Intermountain West Joint Venture 2013b). Dabbling ducks made up the majority of species occurring during spring migration with Northern pintail, American green-winged teal, and Northern shoveler being most abundant. Though present in smaller numbers, common diving ducks included lesser scaup, common goldeneye, ruddy duck, redhead, and canvasback. Spring migrating tundra swans also reach their peak in March.

Peak numbers for fall were estimated to be 2.8 million occurring in early to mid-September (Intermountain West Joint Venture 2013b). Though numbers do drop off after mid-September, estimates still remain high during October (1.5 million) and November (1.5 to 2 million) until wetlands freeze up in late November/early December in a typical year. Dabbling ducks made up more than 90% of waterfowl use during fall migration. Most common dabbling duck species included Northern pintail, Gadwall, American green-winged teal, and mallard. Tundra swans nest in Alaska and are an abundant migrant in the fall that can number up to 60,000 birds by mid-November, mostly found on the Refuge.

Waterfowl numbers using GSL drop off considerably during winter. As freshwater wetlands become ice covered, most waterfowl leave the area headed for California or other southern locales. Some species go as far as Mexico. Remining waterfowl seek out open water often using more saline parts of the Great Salt Lake that do not freeze over. Common winter species include Northern pintail, American green-winged teal, Northern shoveler, mallard and common goldeneye. A few Canada geese and tundra swans may remain during winter.

Though primarily a migratory stopover site for waterfowl, the GSL Ecosystem hosts several species that breed in significant numbers. Cinnamon teal, gadwall. mallard, and northern shoveler are common to abundant nesting dabbling duck species. Common nesting divers include ruddy duck and redhead. Canada geese are abundant nesters throughout the GSL Ecosystem.

Cinnamon teal are the quintessential western dabbling duck. Cinnamon teal are abundant nesters in the intermountain west and particularly the GSL Ecosystem. Historically, the marshes of northern Utah were considered the most important cinnamon teal breeding area (Bellrose 1980). Recent changes in habitat due to the flooding that occurred in the mid-1980s, an increase in mammalian predators, and the invasion of phragmites has impacted cinnamon teal production.

Waterbirds

Great Salt Lake habitats host a large variety of other wetland or water-dependent birds that nest including wading birds such as white-faced ibis, black-crowned night heron, great blue heron, sandhill crane, and snowy egret. More aquatic species such as grebes (eared, Clark's, pied-billed, and Western), double-crested cormorant and American white pelican are common to abundant. Terns (black, Caspian, and Forster's) and gulls (California, Franklin's and Ring-billed) are very abundant in the Great Salt Lake ecosystem.

White-faced ibis nest colonially in tall emergent vegetation such as hardstem bulrush. Ibis feed on invertebrates by wading and probing in shallow water and mud. Historically, GSL hosted the largest breeding colony of white-faced ibis in North America. When the Lake flooded in the 1980s, these birds were displaced and moved to other nesting locations farther north and west. When floodwaters receded, ibis began to return to GSL and are now an abundant nester and migrant. The 1997-2001 GSL waterbird survey tallied a mean of 25,500 ibis from July through August with Bear River Bay habitats supporting a significant portion (Paul and Manning 2002).

Franklin's gull is an abundant colonial nesting and migratory species in the GSL Ecosystem often breeding alongside or within Ibis breeding colonies. The 1997-2001 GSL waterbird survey recorded a mean of 46,500 gulls from July through September with Bear River Bay habitats supporting a significant portion (Paul and Manning 2002).

California gull, the state bird of Utah, is the most abundant breeding waterbird species in the GSL ecosystem. Surveys conducted from May-July, 2009 for the western colonial breeding waterbird atlas (Cavitt et al. 2014) recorded nearly 99,000 gulls in 20 colonies around GSL, the most for any of the colonial bird species surveyed. During the period April-September, the 1997-2001 GSL survey recorded a mean of 81,000 gulls (Paul and Manning 2002). Though gulls were abundant throughout the GSL, Bear River Bay accounted for many of the birds tallied.

Great Salt Lake hosts the largest American white pelican colony in North America (https://wildlife.utah.gov/pelican_webmap) estimated to be over 16,000 birds. The colony is located on Gunnison Island in Gunnison Bay. Because of the salinity in Gunnison Bay, foraging for fish near the colony is not an option. Adult pelicans have to leave Gunnison Bay to find food, often flying east over the Promontory Range to Bear River Bay which offers the most important foraging habitats for pelicans in the GSL ecosystem. After filling up on fish, adult pelicans return across the Promontory Range to the colony to feed their young. GSL hosts a large number of pelicans during the fall migration. The mean number recorded 25,480 for the 5-year GSL waterbird survey (Paul and Manning 2002). Many of these birds are using habitats in the Bear River Bay.

Eared grebes area a common nesting and abundant migrant species. These birds nest over water on mounds constructed of vegetation. The mean number tallied from August-September was 93,000. Bear River Bay accounted for a small portion of the total. The high count of 699,000 was recorded in 1997.

Raptors

Though a number of species nest in the GSL ecosystem, raptors are more abundant during the late fall, winter and early spring. Bald eagle, rough legged hawk and Northern harrier are common to abundant species around the GSL and on the Promontory Peninsula during the winter months. Peregrine falcons attracted by the abundance of potential waterbird prey occur frequently during the fall and spring migration. Nesting raptors include turkey vulture, golden eagle, American kestrel, prairie falcon, Northern harrier, Red-tailed hawk, and Swanson's hawk. Nesting owls include barn owl, burrowing owl, great horned owl, and short-eared owl. Both burrowing owls and short-eared owls are considered state Species of Concern (Utah Division of Wildlife Resources 2017)

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