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ECOLOGICAL RISK ASSESSMENT OF NESJAVELLIR GEOTHERMAL POWER PLANT WASTEWATER DISPOSAL IN LAKE THINGVALLAVATN IN SW-ICELAND

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ABSTRACT

Nesjavellir geothermal power plant is located in SW-Iceland southwest of Lake Thingvallavatn. Utilisation of the geothermal resource for production of electricity and hot water for district heating brings to the surface geothermal fluids with trace elements. The waste fluid is either pumped into shallow drillholes that connect to underground water or disposed of in the Nesjavellir stream, where it disappears into the lava and finds its way into Lake Thingvallavatn. Temperature and quantities of various trace elements were measured in the wastewater discharged from the power plant and at Eldvík, Varmagjá, Markagjá and Grámelur springs on the shore of the lake. Trace elements were also measured in samples of an aquatic plant (*Myrophyllum alterniflorum*), a gastropod snail (*Lymnaea peregra*), a fish (arctic charr *Salvelinus alpinus*), from lake sediments at Varmagjá and at a control station, Vatnskot, on the north shore of the lake. Hg is mostly lost to the atmosphere before it reaches the lake. Other trace elements are modified through chemical reactions and diluted to such an extent that there is little reason for concern except in the case of arsenic. All trace elements in lake shoreline springs were within the international and Icelandic water quality criteria for protection of aquatic life except for arsenic. Heated geothermal discharge is usually cooled to a temperature within upper incipient lethal temperature limits for most organisms before reaching the lake. Wave action also ensures efficient mixing of the spring water at geothermally influenced sites with cold lake water, thus precluding any effects of thermal pollution.

There was no detectable accumulation of trace elements in biological samples from the lake. Trace elements in sediment samples were all within the international sediment quality guidelines for protection of aquatic life. Mercury in fish tissues is within the accepted international limit for protection of human health. No potential adverse ecological effects on the lake are perceptible due to surface wastewater disposal except possible contamination of cold groundwater at Grámelur. Sound management of the wastewater through deep reinjection and regular monitoring of arsenic and other trace elements is recommended.

1. INTRODUCTION

Aquatic ecosystems are composed of the biological community (producers, consumers and decomposers), the physical and chemical (abiotic) components and their interactions. Within aquatic ecosystems, complex interactions of physical and biochemical cycles exist, and changes do not occur in isolation. However, ecosystems have usually developed over a long period of time and organisms become adapted to their environment. In addition, ecosystems have an inherent capacity to withstand and assimilate stress based on their unique physical, chemical, and biological properties. Nonetheless, systems may become unbalanced, by natural factors, which include drastic changes in climatic variations, or by factors due to human activities.

Any changes, especially rapid ones, could have detrimental effects. Adverse effects due to human activities, such as the release of toxic chemicals in industrial effluents, may affect many components of an aquatic ecosystem, the magnitude of which will depend on both biotic and abiotic site-specific characteristics. In evaluation, aquatic ecosystems should be viewed as whole, isolated organisms affected by one or few pollutants not in terms of complex system with aquatic and terrestrial components.

As chemicals or substances are released into the environment through natural processes or human activities, they may enter aquatic ecosystems and partition into particulate phase. These particles may remain in the water or be deposited into the bed sediments where the contaminants accumulate over time. Sediments may, therefore, act as long-term reservoirs for contaminants. Because sediments comprise an important component of aquatic ecosystems providing a habitat for a wide range of benthic and epibenthic organisms, exposure to certain substances in sediments represents a potentially significant hazard to the health of organisms. Effective assessment of this hazard requires understanding of the relationship between the concentration of sediment-associated chemicals and the occurrence of adverse biological effects (CCME, 2001). For example Ólafsson's 1992 study of Lake Thingvallavatn revealed a varied chemical composition of geothermal drillhole fluid and lakeshore springs with the former recording high arsenic concentrations in separator water of 5.6-310.0μg/l and 134-160 μg/l in 1983 and 1991, respectively. At the Varmagjá and Eldvík springs, which are influenced by effluents from the power plant, concentrations rose from 0.6-0.7μg/l in 1984 to 2.2-4.7 μg/l in 1991 (Ólafsson, 1992). Although the increased concentration of As in affected springs was within limits considered safe for fresh water biota, precautionary monitoring measures were recommended.

If hot wastewater from a standard steam cycle power plant is released directly into an existing natural waterway, the increase in temperature can have a very significant impact on communities of aquatic plants and animals. In serious cases this can result in a complete change of the community whereby high temperature tolerant species take over. In milder cases, water temperature variations among sites may create differences in the physiological and behavioural advantages among aquatic organisms, hence influencing their competitive ability and distribution. Thus, Taniguchi et al. (1998) showed that competitive ability measured as food consumption and aggression could be temperature mediated. In the fish study, brown trout was more aggressive than the creek chub (*Semotilus atromaculatus*) at 3-22°C, but above this temperature the creek chub had a competitive advantage over the trout. The creek chub had increasing success obtaining forage items, decreasing the amount of forage available to the trout at higher temperatures.

When spring water affected by effluents from a power plant reaches the lake, it mixes with lake water causing dilution of solutes and lowering of temperature. The effectiveness of the mixing process depends on wind driven currents and on local sheltered conditions at the spring site, e.g. to what extent the site is sheltered from mixing currents. In Thingvallavatn, wind action is frequent and spring water is quickly and effectively mixed with lake water (Snorrason, 1982). Therefore, any effects of high temperature or potentially harmful solutes in springs affected by geothermal effluents or wastewater are predicted to be local and restricted to the spring sites (Snorrason and Jónsson, 1995).

The report harmonises data from various independent studies on the chemistry of Nesjavellir geothermal power plant wastewater and Lake Thingvallavatn ecology. A comparison of temperature and the concentration of trace elements in the wastewater discharge at the separator station, Lake Thingvallavatn shoreline springs and biological samples (lake sediments, aquatic plants, gastropod snail and fish) were made. The concentration levels recorded were compared with the national and international environmental quality criteria and guidelines for protection of aquatic life. Attempts are also made to evaluate if the geothermal developments at Nesjavellir geothermal field are a potential source of chemical contaminants that have been recorded in various biological samples from Lake Thingvallavatn. A prediction is made on the potential ecological outcome of Nesjavellir geothermal power plant wastewater management option in Lake Thingvallavatn.

1.1 Ecological risk assessment

An ecological risk assessment evaluates the likelihood that adverse ecological effects may occur or are occurring as a result of exposure to one or more stressors. The process evaluates and organizes data, information, assumptions, and uncertainties in order to help understand and predict the relationships between stressors and ecological effects in a way useful for environmental decision-making. An assessment may involve chemical, physical, or biological stressors, and one or many stressors may be considered. Most ecological assessments have been developed within a risk management context to evaluate human-induced changes that are considered undesirable. It usually focuses on stressors and adverse effects generated or influenced by anthropogenic activity. The definition of adversity is usually vital as a stressor may cause adverse effects on one ecosystem component but be neutral or even beneficial to other components. Changes that are often considered undesirable are those that alter important structural or functional characteristics or components of an ecosystem. In evaluation of adversity, this may encompass consideration of the type, intensity, and scale of the effect as well as the potential for recovery (U.S. EPA, 1998).

From the environmental perspective, ecological risk assessments are important tools in predicting the likelihood of future adverse effects from a given activity. They are designed and conducted to provide information to risk managers about the potential adverse effects of alternative management decisions. Attempts to eliminate risks associated with human activities in the face of uncertainties and potentially high costs, present a challenge to risk managers. Although many considerations and sources of information may be used by risk managers in the decision making process, ecological assessments are unique in providing a scientific evaluation of ecological risk that explicitly addresses uncertainty.

An ecological risk assessment evaluates the potential adverse effects of human activities on the plants and animals that make up ecosystems. When conducted for a particular place such as a watershed, the ecological risk assessment process can be used to identify vulnerable and valued resources, prioritise data collection activities and link human activities with their potential effects. Assessments provide a focal point for cooperation among different stakeholders, and also a basis for different management options.

Ecological effects may be local, regional or global and may involve a specific type of plant or animal, a community of organisms (the fish in the lake), or an ecosystem (all of the physical and biological components of the lake). The high variability in environmental factors, combined with cumulative interactions of physiochemical and biological processes in an aquatic ecosystem, may complicate the interpretation and evaluation of contaminant-related stressors on the organisms.

1.2 Contribution of ecological risk assessment to environmental decision making

Ecological risk assessment supports many types of management actions, including the regulation of hazardous waste sites, industrial chemicals, and pesticides, or the management of watersheds or other ecosystems affected by multiple non-chemical and chemical stressors. Ecological risk assessment in itself has several features that contribute to effective environmental decision-making:

- Through an iterative process, new information can be incorporated into risk assessments, which can be used to improve environmental decision-making. This feature is consistent with the adaptive management principles (Holling, 1978) used in managing natural resources.
- Ecological risk assessments can be used to express changes in ecological effects as a function of changes in exposure to stressors. This capability may be very useful, particularly to the decision maker who must evaluate tradeoffs, examine different alternatives, or determine the extent to which stressors must be reduced to achieve a given outcome.
- Assessments explicitly evaluate uncertainty. Uncertainty analysis describes the degree of confidence in the assessment and can help risk managers to focus research on those areas that will lead to the greatest reductions in uncertainty.
- Assessments provide a basis for comparing, ranking and prioritising risks. The results can be used in cost-benefit and cost-effectiveness analyses that offer the additional interpretation of the effects of alternative management options.

2. LITERATURE REVIEW

For thousands of years mankind has functioned as an integral part of the environment, and until recently has had no greater impact than any other animal species. However, with increased technological skills, especially in the last century, the capacity to cause environmental changes has increased dramatically. Such changes are not necessarily bad as such but have been unpredictable and irreversible in the short term, mainly due to poor understanding the environment and of environmental processes (Hunt, 2001).

In recent years concern about industrial prospects has shifted somewhat from an overwhelming emphasis on economic viability to considerable emphasis on environmental viability (Ármannsson and Kristmannsdóttir, 1992). In comparison with other fossil fuel energy sources, geothermal energy has generally been accepted as being an environmentally benign energy source. Geothermal developments in the last 40 years have, however, shown that it is not free of adverse impacts on the environment. These impacts are of increasing concern, and to an extent that they may limit future geothermal developments. Hiding or ignoring such environmental problems may be counterproductive to the development of the industry as it may lead to loss of confidence in the industry by the public, regulatory, and financial sectors. A good example of the consequences of ignoring problems is the nuclear power industry. To further the use of geothermal energy, all possible environmental effects should be identified, and countermeasures devised and adopted to avoid or minimise their impacts.

In order to mine heat contained in the rock most geothermal energy developments bring fluids to the surface. In high-temperature liquid-dominated geothermal fields, the volume of the resultant liquid waste involved may be large. For example at Nesjavellir geothermal power plant, it is currently about 258 kg/s of brine, 126-140 kg/s condensate and 343-1776 l/s of cooling water (Gíslason, 2000). For vapour-dominated systems it is less, and for low temperature systems it is very much less. The waste fluid is usually disposed of in waterways or evaporation ponds, or re-injected into the ground. Surface disposal causes more environmental problems than reinjection.

Unless all waste borewater and cooling water blowdown is reinjected, geothermal fluid discharges may have an impact on the local and regional surface water such as rivers, lakes and estuaries. Environmental effects are not only due to the volumes involved, but also to the relatively high temperatures and toxic chemicals in waste fluid. For example, at Wairakei (New Zealand) the wastewater has a temperature of about 140°C (Hunt, 2001) while at Nesjavellir it is in the range of 46-100°C. The chemistry of the fluid discharged is largely dependent on the geochemistry of the reservoir and the operating conditions used for power generation, and varies from one geothermal field to another (Webster, 1995). For example, fluids from Salton Sea field (USA), which is hosted by evaporative deposits, are acidic and highly saline ($pH < 5$, $[Cl] = 155,000$ ppm) while those of Hveragerdi field (Iceland) are alkaline and of low salinity ($pH < 9$, $[Cl] = < 200$ ppm). Most high-temperature geothermal water may contain high concentrations of at least one of the following toxic chemicals: lithium (Li), boron (B), arsenic (As), cadmium (Cd), lead

(Pb), mercury (Hg) and sometimes ammonia (NH_3), as shown in Table 1. If released into river or lake, the contaminants can potentially have negative impacts on aquatic life, terrestrial plants and humans. This can also adversely affect both surface and groundwater quality.

TABLE 1: Contaminant concentrations in selected geothermal fluids and in a world average freshwater, in mg/kg (Webster, 1995)

	Li	B	As	Hg	H_2S	NH_3
Freshwater	0.003	0.01	0.002	0.00004	bdl	0.04
Deep well water						
Salton Sea (US)	215	390	12	0.006	16	386
Cerro Prieto (Mexico)	-	19	2.3	0.00005	0.16	127
Wairakei (New Zealand)	14	30	4.7	0.0002	1.7	0.2

bdl = Below detection limit

The effect of geothermal development in the Carson desert and Reno-Sparks area, Nevada (USA) on the surrounding water quality has also been reported with As, B and Hg concentrations being high in comparison to other groundwater sources in the area. For example, Carson desert area concentrations of As and B were 130 μg/l and 18,000 μg/l, respectively (Bevans et al., 1998).

Most of the chemicals are present as solute and remain in solution from the point of discharge, but some are taken up in river or lake bottom sediments, where they may accumulate. The concentrations in such sediments can become greater than the soluble concentration of the species in water, so that re-mobilisation of the species in sediments, such as during earthquakes or floods, may result in a toxic flush of the chemical species into the environment (Hunt, 2001). Chemicals which remain in solution may be taken up by aquatic vegetation and fish (Webster and Timperly, 1995), and some can also move further up the food chain in birds and animals residing near the lake/river. For example, in New Zealand, annual geothermal discharge into the Waikato River contains 50 kg mercury, which is partly responsible for the high concentrations of mercury (often greater than 0.5 mg/kg of wet flesh) in trout from the river and high sediment mercury levels (> 200 μg/kg).

Disposal of hot wastewater from a standard steam cycle power plant directly into an existing natural waterway, leads to an increase in water temperature, which may have a negative impact on aquatic plants and animals. Release of untreated chemicals may also result in aquatic poisoning and contamination of shallow groundwater supplies, making them unfit for human use, as shown in Figure 1.

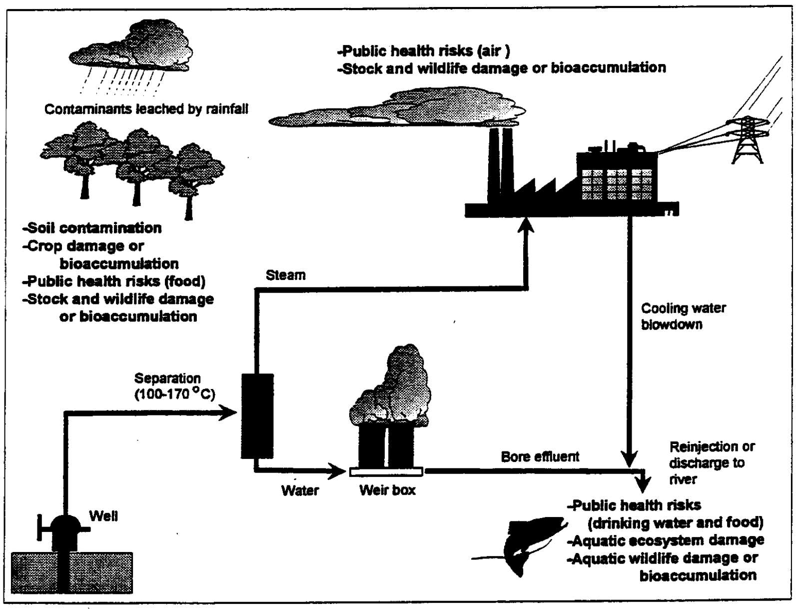

FIGURE 1: Potential biological impacts of geothermal development (Webster and Timperly, 1995)

The research history of Lake Thingvallavatn goes back in time to 1706-1711, when the Icelandic farm codex was established with valuable information on agriculture and fisheries around the lake. First limnological and biological observations are 150 years old, made by Icelandic naturalist and poet Jónas Hallgrímsson (1840-1841). Another study on the ecology of planktivorous charr due to increased fishing, was conducted in 1937 and 1938.

In 1974, on the occasion of the 1100th anniversary of settlement in Iceland, the chairman of the Thingvellir commission initiated another research project on the lake. The research conducted in the seventies focused on limnology, chemistry, vegetation, primary production and animal life of the lake (Jónasson, 1992). Ólafsson (1992) evaluated the input of dissolved matter (basically inorganic nutrients and their utilization) into the lake (Table 2). The study showed that high-temperature geothermal wastewater from Nesjavellir power plant has a chemical composition quite different from that of cold groundwater (Table 3) and has undesirably high concentrations of toxic trace elements, especially mercury and arsenic as depicted by trace constituents in spring water at Nesjahraun (Table 4). Trace element studies for Zn, Cu, Pb and Cd in separator water and at Markagjá, Eldvík and Varmagjá springs were also done in 1996 (Björnsdóttir, 1996) (Table 5).

TABLE 2: Mean chemical composition of water flowing into Lake Thingvallavatn (Ólafsson, 1992)

Group	pH	Alk. (meq/l)	Cl (mg/l)	SO_4 (mg/l)	Na (mg/l)	K (mg/l)	Ca (mg/l)	Mg (mg/l)	PO_4 -P (μg/l)	NO_3 -N (μg/l)	Si (mg/l)
1	8.0	0.47	8.0	2.4	8.2	0.59	4.5	1.8	26.2	25.2	7.4
2	6.9	0.28	7.6	1.8	5.4	0.35	2.3	1.4	0.7	24.5	2.4
3	7.4	0.57	7.1	2.1	6.4	0.54	4.2	3.2	21.5	14.0	7.1
4	7.7	0.79	7.0	15.0	7.1	0.46	12.8	3.3	23.5	10.5	8.9
5	8.9	0.39	4.9	1.9	6.8	0.53	3.2	1.0	26.4	30.8	6.8
6	9.2	0.55	6.5	2.3	11.0	0.47	4.3	0.7	20.4	41.3	6.2
7	7.8	1.17	7.8	8.6	10.9	1.30	9.6	5.6	58.6	47.6	11.5

Groups: 1) Brook at Heidarbær, spring at Skálabrekka; 2) Öxará, Lindin, Móakotsá, Torfadalslækur; 3) Villingavatnsá; 4) Ölfusvatnsá; 5) Vatnsvik, Vellankatla; 6) Flosagjá; 7) Varmagjá, Hagavík.

TABLE 3: Concentration of trace constituents in Nesjavellir geothermal discharge in 1991 (Ólafsson, 1992)

	Well NJ-11	Well NJ-13	Power station
PO_4 -P (μg/l)			
NH_3 -N (μg/l)	239	163	236
Zn (μg/l)	1.7	1.7	1.2
Pb (μg/l)	0.11	0.54	0.09
Cd (μg/l)	0.72	0.23	0.05
Cu (μg/l)	0.3	0.3	1.2
Mn (μg/l)	3.9	1.7	1.2
Ni (μg/l)	1.0	0.2	1.3
As (μg/l)	134	160	120
Hg (ng/l)	44	143	261

Trace element concentrations in the earlier studies compare closely with natural concentrations in unpolluted Antarctic ice (Fergusson, 1990). The Antarctic is considered to be the most remote and unpolluted place on earth with respect to inorganic metal (especially heavy metals) levels. Other places, such as the arctic, correspond to some global contamination. In unpolluted freshwater, however, the background concentration of most elements may be low (Table 6).

TABLE 4: Trace constituents in spring water at Nesjahraun in 1984 and 1991, respectively, in µg/l (Ólafsson, 1992)

	Markagjá (1984/1991)	**Varmagjá (1984/1991)**	**Eldvík (1984/1991)**
NH_3-N	0.3	0.2	0.2
PO_4-P	32.6/38.9	58.7/65.6	65.5/44.4
NO_3-N	28	81	65
SiO_2	11.5/14.9	32.3/37.9	33.6/47.7
Ni	0.1/0.1	0.2/0.1	0.1/0.1
As	1.1/0.5	0.6/2.2	0.7/4.7
Cd	0.005/0.11	0.006/0.04	0.004/0.06
Pb	0.07/0.07	0.06/0.03	0.05/0.1
Hg	0.0018/0.0008	0.0009/0.008	0.0009/0.0025
Mn	1.0/1.0	0.1/0.2	0.1/1.2
Cu	0.4/1.0	1.2/1.2	0.7/1.5
Zn	0.2/2.9	0.2/1.1	0.3/0.3

TABLE 5: Trace constituents in spring water at Nesjahraun, in µg/l (Björnsdóttir, 1996)

	Markagjá	**Varmagjá**	**Eldvík**	**Separator water**
Zn	4.2	3.0	1.7	2.8
Cu	1.0	0.6	0.8	0.4
Cd	0.2	0.04	bdl	bdl
Pb	0.4	1.0	0.8	0.6

bdl = Below detection limit

TABLE 6: Typical levels of heavy elements in freshwater, in µg/l (Fergusson, 1990)

Elements	**Fresh water**	**Non-polluted rivers**	**Polluted rivers**	**Lakes**	**Rivers mining areas**	**Geothermal water**
Cd	< 1	0.01 - 1	1 - >10	0.01 - 20	100 - 700	0.01 - 0.5
Hg	0.02 - 0.1	0.0001 - 1	> 1	0.02		0.05 - 60
Pb	< 10	< 1 - 10	20 - 100	0.1 - 30	100 - 1000	1 - 10
As	< 1 - 5	< 1 - 10	10 - 1000	1 - 70	100 - 5000	1000 - 5000
Se	< 0.1	0.1 - 0.3				

The earlier supposition by Ólafsson (1992) of potential ecotoxicological effects of Nesjavellir geothermal field developments on Lake Thingvallavatn was followed by studies of Snorrason and Jónsson (1996, 2000) of trace metals in biological samples (sediments, aquatic plant *Myrophyllum alterniflorum*, fish - small benthivorous arctic charr *Salvelinus alpinus* and gastropod snail *Lymnaea peregra*) from Lake Thingvallavatn. Gastropod snails (*Lymnaea peregra*) are benthic dwellers feeding on epilithica of epiphytic alyae and sediments and form 80-90% of the diet of small benthivorous charr (Snorrason et al., 1992; Malmquist et al., 1992). Evidence of any bioaccumulation of metals from sediments and water should be detectable in either or both of the two organisms.

3. ECOLOGY OF THINGVALLAVATN

3.1 Location of the study area

Iceland has at least twenty-eight high-temperature geothermal fields with a temperature of more than 200°C at 1 km depth. Such areas have been shown to be favourable to exploration and development of geothermal power plants, such as Nesjavellir power plant located in the Hengill high-temperature area southwest of Lake Thingvallavatn and 27 km east of Iceland's capital, Reykjavík (Figure 2).

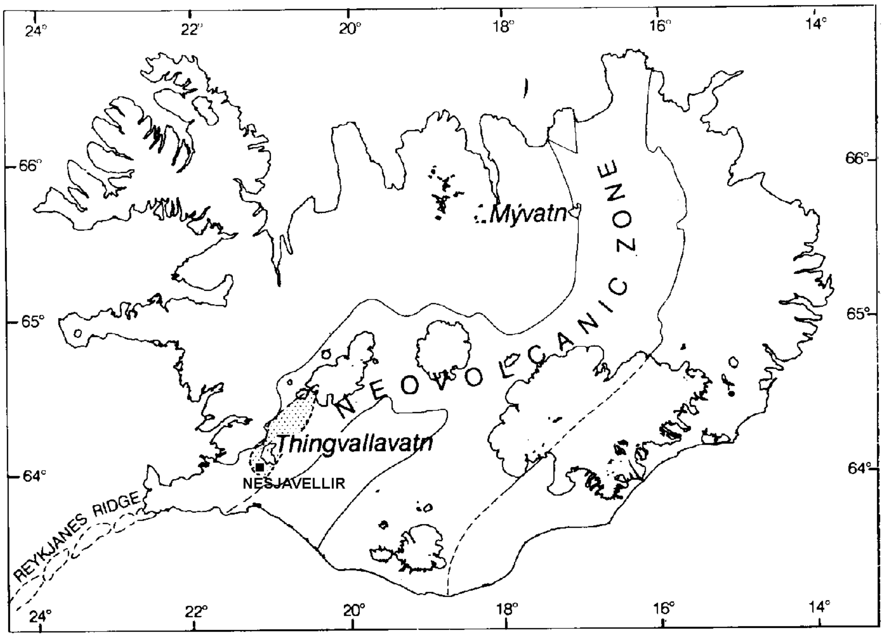

FIGURE 2: Map of Iceland with the location of Lake Thingvallavatn and Nesjavellir geothermal power plant

3.2 Physical characteristics of Lake Thingvallavatn

Figure 3 shows the physical characteristics of Lake Thingvallavatn. The lake has an area of 83 km^2 at an elevation of 100.5 m above sea level. The mean and measured maximum depths are 34 and 114 m, respectively. The lake catchment area is estimated to be around 1000 km^2, of which approximately 50% is covered by postglacial lava. The mean discharge in the last 50 years is 100 m^3 /s and mean annual precipitation has been estimated to be 1000-1200 mm. Most of the precipitation percolates through the porous lava. Hence, the lake is 90% fed by underground springs. The main spring areas in the north show a constant temperature of 2.8-3.5°C. Warmer groundwater enters the lake from southwest, owing to the nearby Hengill geothermal area (Adalsteinsson et al., 1992; Einarsson, 1992).

3.3 Geology

Thingvallavatn occupies a NE-SW elongated graben within the western branch of the volcanic rift zone in SW-Iceland. The topography of the graben floor slopes off from the Langjökull ice cap and Skjaldbreidur lava shield of over 1000 m altitude in the northeast to below sea level in the western part of the lake. The bedrock of the catchment is composed of Postglacial lava, Pleistocene pillow lava and hyaloclastites, all of which are basaltic. As a result, drainage from the main part of the catchment usually seeps into the lava to emerge as springs on the shores of the lake. The Nesjavellir geothermal field, located south of the lake is part of the Hengill volcanic system, comprised of hyaloclastites and olivine tholeiitic lava. There have been 3-4 eruptions within the Nesjavellir fissure swarm in Postglacial times. The last was around 2000 years ago, forming an 11 km^2 lava field known as Nesjahraun and the crater island Sandey in the lake (Saemundsson, 1992) (Figure 4).

3.4 Vegetation and soils

Soils around the lake are mainly of aeolean origin, well drained with a low clay content and a mineral fraction consisting mainly of volcanic ash. The soils are also very susceptible to wind and water erosion. Any of the lowland areas are covered by birch wood (*Betula pubescens* Ehr.), and in some areas exotic species of Sitka spruce (*Picea sitchensis* Bong.) and lodgepole pine (*Pinus contorta* Loud.) have been planted. The most extensive plant communities of this open rangeland area are moss heath (*Racomitrium* spp), dwarf shrub heath (*Betula nana-Vaccinium* spp-*Empetrum nigrum-Calluna vulgaris-Dryas octopetala-Salix* spp associations), graminoid-heath (Grasses and *Carex* spp associations) and wetlands (halfbogs, bogs and fens) (Thorsteinsson and Arnalds, 1992).

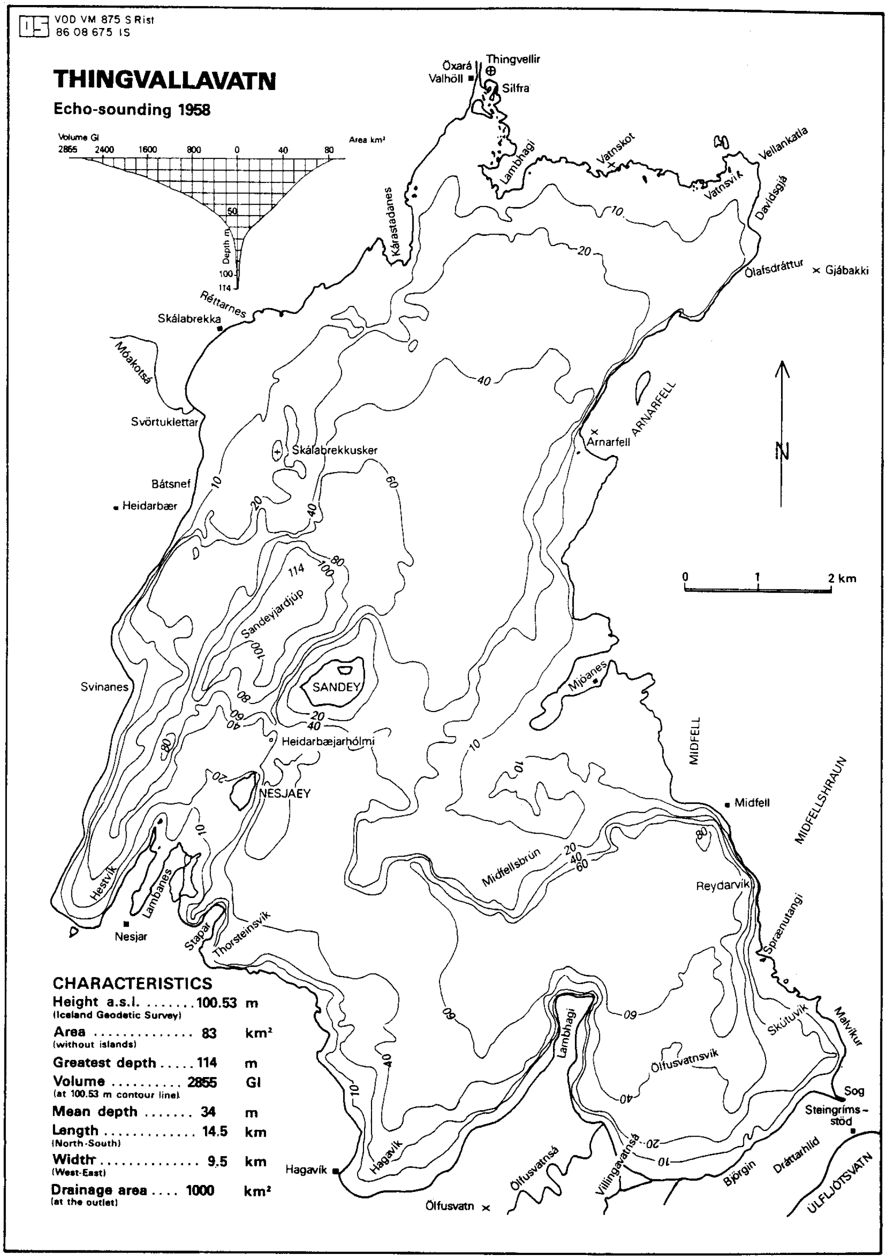

FIGURE 3: Physical characteristics of Lake Thingvallavatn (Adalsteinsson et al., 1992)

3.5 Hydrology

The Thingvallavatn catchment area has been estimated to be 1,000 km^2, stretching from the Hengill mountain in the southwest to Langjökull in the northeast. Lava covers about half of the catchment area. Half of the Thorisjökull and Langjökull glaciers are drained to the Thingvallavatn groundwater system. Surface runoff from the glaciers percolates through the glacial deposits north of the Skjalbreidur shield and through the postglacial lavas, as does the precipitation that falls in the entire area northeast and east of the lake. The lake catchment is surrounded by mountains: Botnssúlur, Skjaldbreidur, Hlödufell and Thórisjökull (Adalsteinsson et al., 1992). Underground water flow from Langjökull in the north and to Hengill in the south towards the lake has been verified (Figure 5).

Inflow into the lake cannot be fully quantified but taking into account the known springs, the lake is likely to be 90% spring-fed as shown by earlier isotopic study. The main cold water springs enter the lake at the northern shore, while at the southwestern shoreline several springs have been observed that have elevated temperature due to the geothermal activity of the Nesjavellir high-temperature field. The oxygen isotopic composition of this water is slightly heavier than observed elsewhere within the catchment area due to contamination by the Nesjavellir thermal fluid, which is enriched with the heavy isotope ^{18}O due to oxygen isotopic exchange with the bedrock (Sveinbjörnsdóttir and Johnsen, 1992).

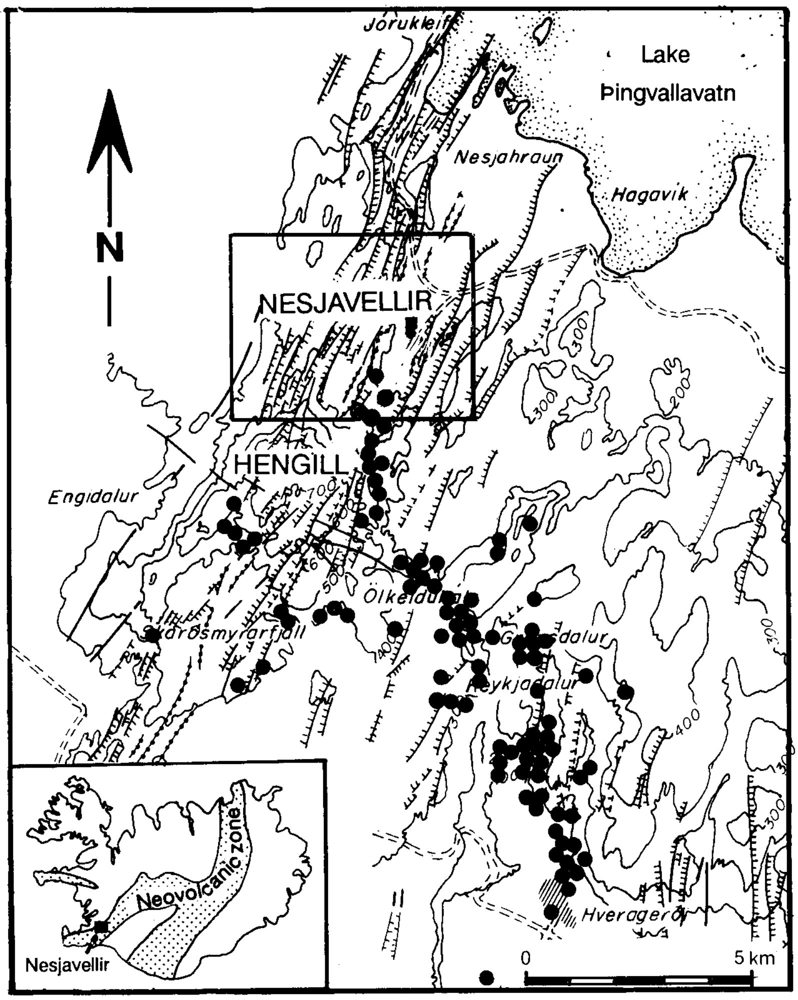

FIGURE 4: Tectonic map of the Hengill area, the location of active geothermal manifestations is shown by black dots; the Nesjavellir field is located within the square (Gunnarsson et al., 1992)

3.6 Climatic conditions

The mean temperature is rather low in the Thingvallavatn area in comparison with the lowlands in SW Iceland. Monthly mean temperatures range from -2.4 - 11.2°C at Thingvellir with January and July being the coldest and warmest months, respectively. A large part of the precipitation occurs in the mountains, with decreasing values on the leeward side. Annual values of 3000 mm and 1000-1200 mm have been recorded in the mountains and lowland areas, respectively. Northerly and southerly winds are common in the area. The southerly winds blow along Thingvallavatn towards the northern end of the lake. Monthly mean values of wind speeds of 2.7-3.1 m/s and 2 m/s in winter and summer, respectively, are experienced (Einarsson, 1992).

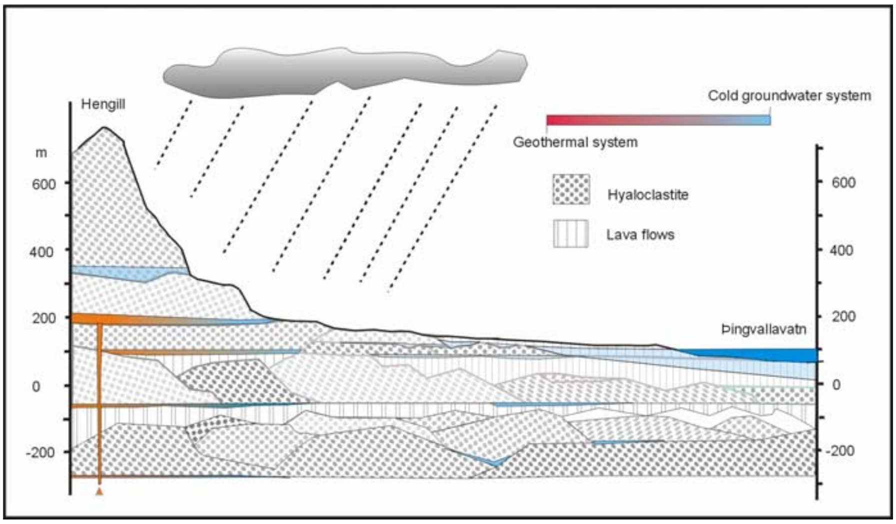

FIGURE 5: Hengill groundwater system (Franzson, 2000)

3.7 Phytoplankton

Secchi disc transparency ranges of 6-15 m and a temperature range of 0-13°C have been recorded in the lake. The nutrients N, P and Si are known to enter the lake at varying concentrations and nitrogen has been seen to limit phytoplankton production. Main phytoplankton species in the lake are: large diatom species such as *Melosira islandica* and *M. italica*, which dominate except in mid-summer (Jónasson et al., 1992). In July and August chrysophyceae and pyrrophyta may dominate the production of the phytoplankton, with large diatoms returning in autumn.

3.8 Zooplankton

The zooplankton is dominated by three species of rotifers and three species of crustaceans. The rotifers *Polyarthra dolichoptera* (Idelson) constitute up to 90% of the rotifer biomass in summer, while in autumn *Keratella cochlearis* (Gosse) and *Conochilus unicornis* (Rousselet) together contribute 55-80% of the rotifer biomass. *Cyclops abyssorum* varies from a minimum of 30 to 100% of the crustacean numbers during winter. *Leptodiaptomus minutes* and *Daphnia longispina* are also present. The crustacean zooplankton, mainly *Daphnia* and Cyclops, form a great portion of the food for the planktivorous morph of arctic charr (*Savellinus alpinus* (L.)) in the lake (Antonsson, 1992; Malmquist et al., 1992).

3.9 Epilithic algal communities

Epilithic algal communities in Lake Thingvallavatn have different species and depth specific patterns for growth, photosynthesis and respiration. Their photosynthesis is very low and decreases with increasing depth. Main algal communities found in the lake are *Cladophora* sp., *Ulothrix* sp. and *Nostoc* sp. The littoral zone (0-20 m depth), constituting 18% of the total lake surface area, provides over 32 km^2 of substrate covered by green algae, blue-green algae and diatoms. Therefore, the contribution of this community to the total primary production of the lake is of great importance (Jónsson, 1992).

3.10 Zoobenthic community

Macrozoobenthos of Lake Thingvallavatn consist of approximately 60 taxa dominated by 24 Chironomidae and 16 Oligochaeta species (Lindegaard, 1992). By abundance, they can be categorized as shown in Table 7.

TABLE 7: Average abundance of macrozoobenthos (no. of individuals per m^2) in the littoral and profundal zones of Lake Thingvallavatn (modfied from Lindegaard, 1992)

Family	Littoral zone (0-20 m)	Profundal zone (20-114 m)
Hydra sp.	435	2
Planaria	14	31
Nematoda	418	0
Mollusca	8489	116
Oligochaeta	17224	3567
Hirudinea	143	0
Cladocera	1476	0
Plecoptera	616	0
Trichoptera	140	0
Chironomidae	20926	170
Empedidae	2458	0

3.11 Fish species

Lake Thingvallavatn supports three fish species: arctic charr (*Salvelinus alpinus* (L.)), brown trout (*Salmo trutta* L.) and threespined stickleback (*Gasterosteus aculeatus* L.); arctic charr is dominant in terms of biomass. The charr exhibit four different, trophically specialized morphs: small and large benthivorous charr, piscivorous charr and planktivorous charr. The planktivorous arctic charr is the numerically dominant morph (Snorrason et al., 1992).

3.12 Avifauna (waterbirds)

Birdlife of Thingvallavatn had not received much attention in the past, as there has never been an overall ecological study of the waterfowl population there as compared to Lake Myvatn in northern Iceland. Species found here are mainly waterbirds i.e. the birds that are dependent on the lake and on the lakeshore for food, nesting and breeding. Among the birds found in the area are some unusual species such as the great northern diver (*Gavia immer*), barrow's goldeneye (*Bucephala islandica*) and Gyrfalcon (*Falco rusticolus*). The most common birds however are mallards (*Anas platyrhynchos* L.), merganser (*Mergus serrator* L.), tufted duck (*Aythia fulgula* L.) and grey-lag geese (*Anser anser* L.). Also found here are land birds which are known to be regular visitors of the lake (Magnusson, 1992).

4. METHODOLOGY

4.1 Sample definition, preparation and analysis

Samples were collected from Vatnskot, a control station not influenced by thermal effluents, and in Varmagjá in Thorsteinsvík stations on Lake Thingvallavatn in May 2000. Before freeze-drying, samples were rinsed with ultra-pure water and visible dirt was removed. The submerged aquatic plant (*Myrophyllum alterniflorum*) from Vatnskot was light brown in colour while that from Varmagjá was bright -green. The aquatic plant samples were further defined by measuring dry material upon freeze-drying, as well as the ash, salt and moisture content. The gastropod snail (*Lymnaea peregra*) was defined in the same way (Audunsson, 1996).

Three samples each of arctic charr (*Salvelinus alpinus*) were taken from Varmagjá (Thorsteinsvík) and Vatnskot. The samples from Varmagjá contained over three times heavier individuals, on average, as compared to those from Vatnskot. The three *Salvelinus alpinus* samples from Vatnskot differed in composition, showing the greatest variation in size. To make a reasonable comparison, each sample was grouped according to size/age. This sample division also made results more reliable as more than one measurement was performed on each sample and local pollution of part of the samples does not affect the results of the whole sample (Audunsson, 1996).

When *Salvelinus alpinus* had been weighed and its length measured, its belly was opened and the liver extracted and weighed. The stomach contents were inspected, maturity stage and gender determined. Before measurement, the livers were homogenized. The fish was filleted after removal of otoliths and skin. Fish flesh was divided in sub-samples according to liver divisions, and fillets from each sub-sample were homogenized.

4.2 Metal analysis

Samples for mercury analysis were broken down by HNO_3 / H_2SO_4 / $KMnO_4$ and then titration of excess $KMnO_4$ with hydroxylamine, while other samples for determination of other metals were broken down by HNO_3 in quartz bombs. After breakdown, mercury was determined by *Cold Vapour Atomic Absorption Spectroscopy* (CVAAS) using $SnCl_2$ in HCl as a reducing agent. Lead was determined by *Graphite Furnace Atomic Absorption Spectroscopy* (GFAAS) on a 1 Vov scaffold using $Mg(NO_3)_2$+KH_2PO_4 as a matrix modifier and Zeeman background correction. Selenium and arsenic were determined by hydride generation *Atomic Absorption Spectroscopy* (AAS) after reduction of Se(VI) to Se(IV) using hydrochloric acid. Other metals, cadmium, zinc, copper and manganese were determined using *Flame Atomic Absorption Spectroscopy* (FAAS) with D2-background correction. Dry material was determined as weight loss upon drying at 102-105˚C for 16-18 hours (Audunsson, 1996).

4.3 Lake Thingvallavatn shoreline springs

Water samples from warm springs at Varmagjá, Eldvík, Markagjá and Grámelur were collected (see Figure 6). Water was collected directly into polythene bottles. Samples for analysis of trace metals other than mercury were acidified to pH < 2 with hydrochloric acid. Analysis was done by SGAB Analytica Laboratory of Sweden.

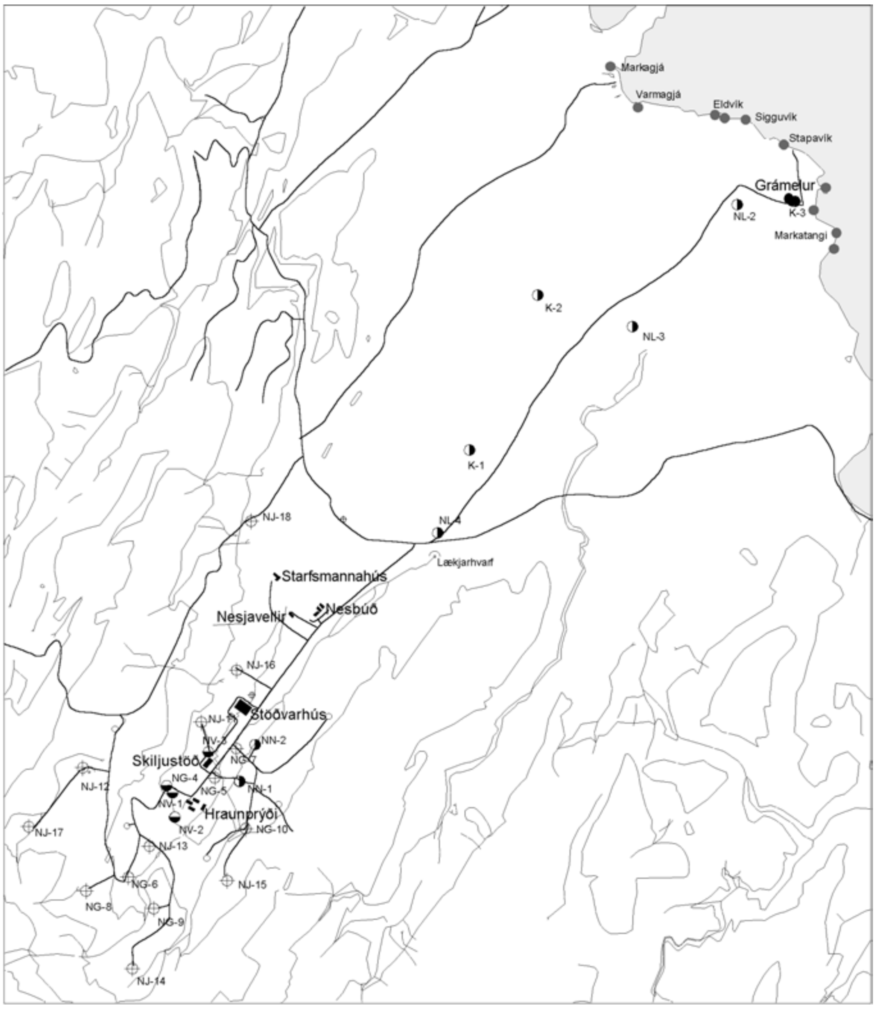

FIGURE 6: The Nesjavellir area and the sampling points of warm water springs on Lake Thingvallavatn shore

4.4 Separator water at Nesjavellir geothermal power plant

The Nesjavellir geothermal power plant was commissioned in 1990, utilising geothermal steam for heating fresh groundwater for district heating services. Since 1998 the power plant was also engaged in the production of electricity (Figure 7). Present installed capacity is 90 MWe and 200 MWt. The electricity production phase is a steam cycle design, which uses cold fresh water from boreholes near Grámelur for cooling condensed steam. In the heat exchangers the geothermal water heats cold water and is cooled to 55˚C. Used and unused brine at a flow rate of 115-143 kg/s at 46 - 100˚C is discharged into the nearby Nesjavellir stream that disappears into Nesjahraun lava at Laekjarhvarf. This mixes with groundwater, which flows some 3.8 km to Lake Thingvallavatn. About 126-140 kg/s of condensate at 48-68˚C, and 343-1776 l/s of cooling water at 49-69˚C is also discharge into shallow drillholes that connect to surface groundwater (Gíslason, 2000).

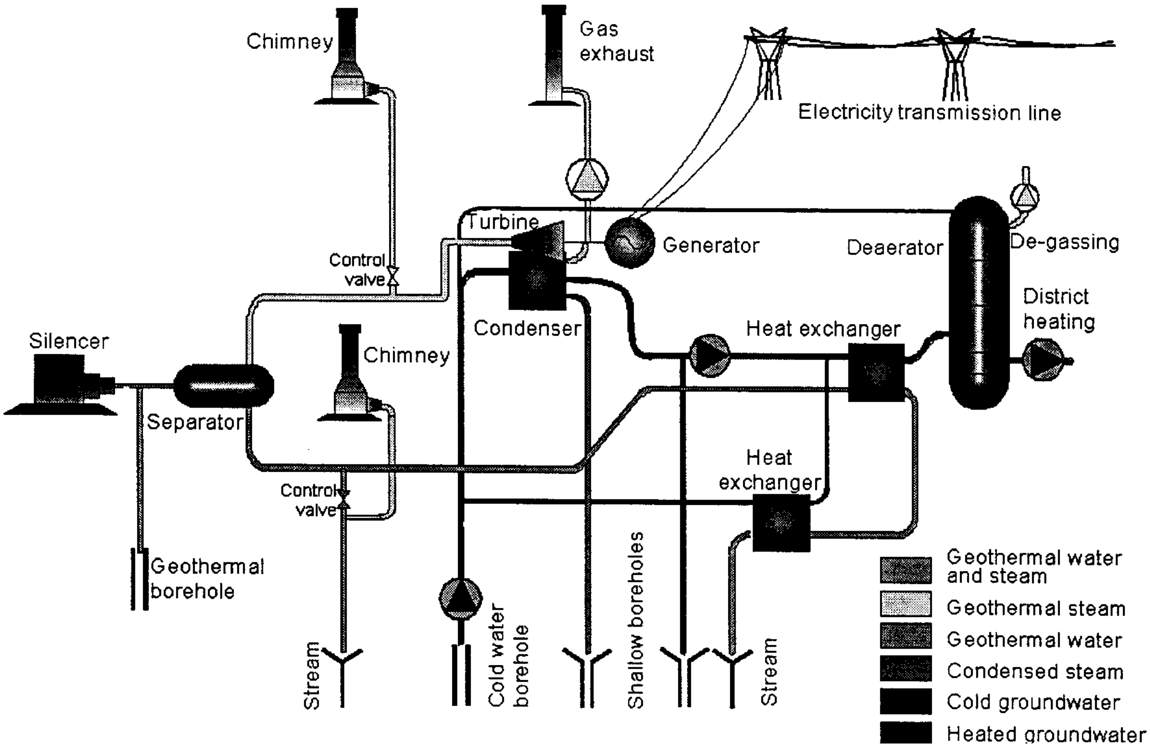

FIGURE 7: Flow diagram for the Nesjavellir geothermal power plant (Gíslason, 2000)

The high-temperature drillhole fluid was sampled at the separator station. To prevent volatilization losses of Hg from separator water, samples were preserved by the addition of oxidized $KMnO_4$ solution and HNO_3. A summary of the analytical methods employed for springs and geothermal wastewater is presented in Table 8.

TABLE 8: Analytical method for springs and geothermal water

Analyte	Technique
Zn, Pb, Ni, Se, Cu, Cr, Cd, As and Mn	ICP-SMS (*Inductively Coupled Plasma Sector Mass Spectrometry* with no pre-treatment other than acidification and dilution
Ca, Si, K, Mg, Na, Mn and Al	ICP-AES *(Optical Emission Spectrometry with Inductively Coupled Plasma*); comparable to flame atomic absorption
Hg	AFS (*Atomic Fluorescence*), mercury is determined after cold evaporation

5. RESULTS AND DISCUSSION

5.1 Geothermally heated discharge and underground cold water

Underground flow of geothermal wastewater is confined just above the groundwater table as shown by the temperature profile of experimental drillholes in the Nesjahraun wastewater run-off area. A drastic drop in the temperature profile in the drillholes is observed a few metres below the groundwater table, depicting a shift from warm to cold water. For example, drillhole NL-08 situated approximately 1 km from Grámelur showed a temperature drop from 16˚C at 15 m depth to 8˚C at 30 m depth. This inflow resulted in elevated spring water temperatures around Grámelur (Figure 8). There is a probability of groundwater contamination at Grámelur, which will render it unfit for use in district heating and cooling.

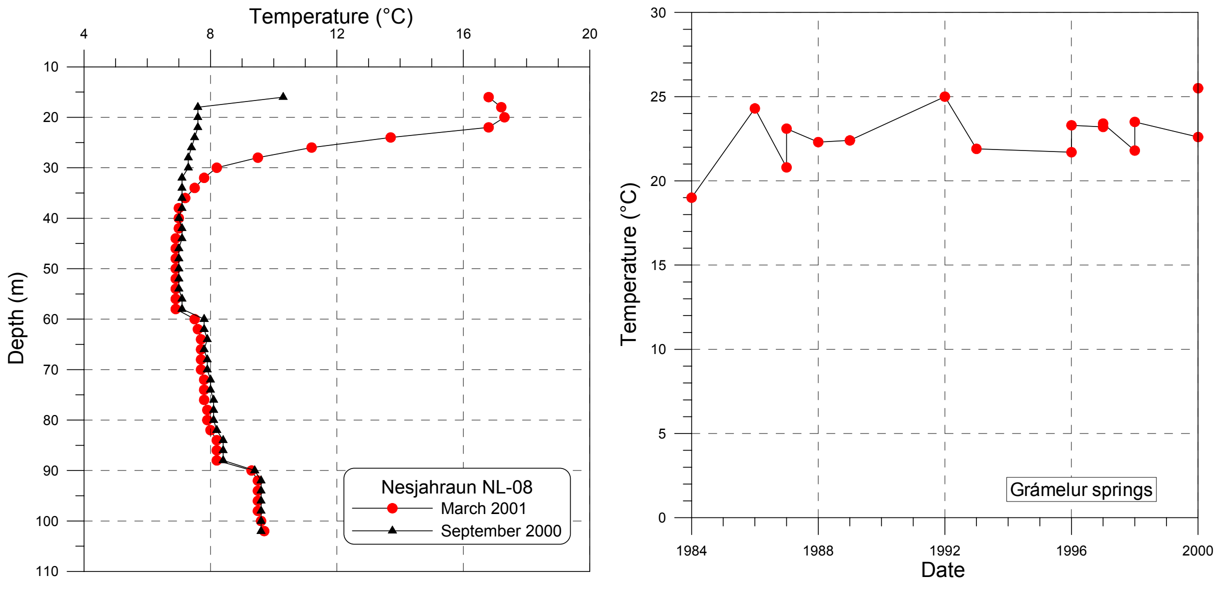

FIGURE 8: Temperature profile of drillhole NL-08 in Nesjahraun lava and water temperature of Grámelur springs (data from Reykjavík Energy)

5.2 Effect of heated discharge on lake water temperature

Figure 9 shows the graphical plots for the temperature of Varmagjá and Eldvík warm water springs on the shoreline of Lake Thingvallavatn from 1979 to 2000. The mean annual water temperature recorded at the warm springs on the shore of Lake Thingvallavatn from 1990-2000 varies between 19.7 and 26.0˚C for both winter and summer temperature readings (Appendix I). This temperature compares very closely to temperatures recorded at the same points during the period 1975-1990, before commissioning the power plant (Figure 9). The sampling points received some natural geothermal inflow before the development of the Nesjavellir geothermal field from springs whose outflow was estimated to be 15 MWt (Gunnarsson et al., 1992). Temperature profile measurements in experimental drillholes in the Nesjahraun wastewater run-off area indicate that geothermal wastewater is confined to the upper layer of the groundwater table, as observed in drillholes NL-11 and NK-02, located approximately 1 km from the lake (Hafstad, 2000 and

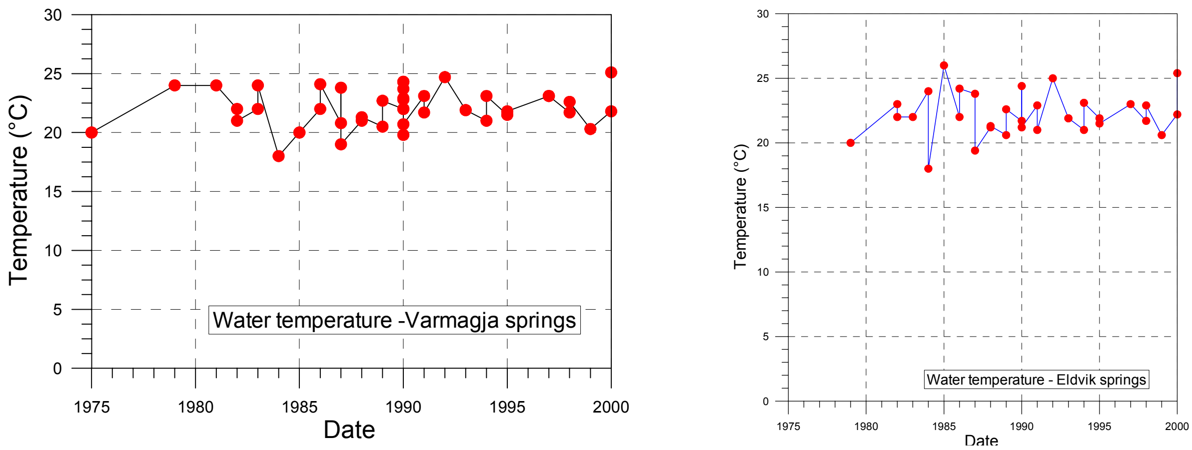

FIGURE 9: Water temperature of Varmagjá and Eldvík springs on the shoreline of Lake Thingvallavatn (1979-2000)

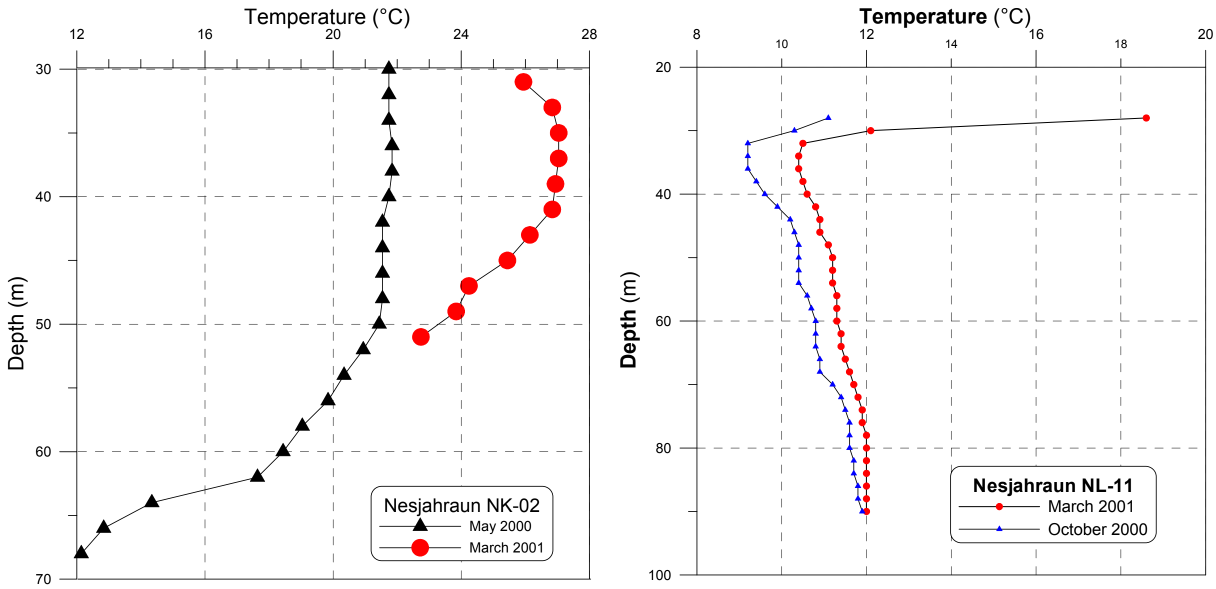

FIGURE 10: Temperature profiles of shallow experimental drillholes (NK-02 and NL-11) in the Nesjahraun lava near Lake Thingvallavatn

2001). An immediate drastic drop in the temperature profile in the drillholes is observed a few metres below the groundwater table (Figure 10). Geothermal water thus finds its way into Lake Thingvallavatn with a considerable temperature drop due to mixing with cold groundwater, leading to only moderate rise in water temperature in the springs.

5.3 Effects of heated discharge on lake ecology

5.3.1 Plankton community

The effects of heated discharge on an aquatic ecosystem can be both beneficial and detrimental depending on local environmental conditions. Effects may range from an impact on phytoplankton, increasing their population with heat or nutrients associated with the sources of heated effluents. The effect of artificially increasing the temperature regime of a species tends to increase growth and photosynthesis, so long as light is sufficient for these functions and the limits of temperature tolerance are not reached. As one approaches the limits of temperature tolerance for a species, cell division is repressed, as is photosynthesis, and the formation of reproduction cells. The cell size is often reduced and the oxygen required for respiration is increased, thus the pattern of growth may be greatly altered. When temperature rises above 35˚C, blue-green algae often become dominant and if this high temperature is maintained for a long period of time, the ecosystem may be severely damaged as these algae are a poor source of food. Blue-green algae are believed to be responsible for a variety of fish poisoning. Stimulation of phytoplankton growth may be beneficial, in that it provides food for zooplankton. However, such eutrophication increases turbidity, which may decrease the ability of visual-feeding fish to locate their prey. For example, salmon react poorly to different light intensity at higher temperatures. Turbidity from blooms may retard photosynthesis in benthic algae and vascular plants, which might form an integral part of the nursery ground of larval or juvenile fish (Ruth, 1968).

Under natural conditions, algae may be classified as thermophilic with optimum temperature ranges for growth above 30˚C; intermediate species, which prefer temperatures between 10 and 30˚C; and cold-water forms (Ruth, 1968). The thermophilic algae are commonly found in thermal springs and streams resulting from geysers. In Varmagjá and Eldvík, the temperature of inflowing springs in the range of 19.7-26.0˚C (see Appendix I) will probably have changed the species composition of the algal community in the immediate vicinity. A few metres distance from the inflow, efficient water mixing usually causes a temperature drop to normal lake temperature. Here we will find the normal cold water adapted algal communities (Jónsson, 1992). Other ecological conditions such as light and nutrients also influence the kind of communities which we find present at various seasons of the year (Ruth, 1968).

5.3.2 Effects of heated discharge on the benthic community

Most temperate-zone species of freshwater benthic organisms are eurythermal. In the course of the year they live at temperatures somewhere between 0 and 32˚C (Wurtz, 1968). This is not to say these organisms can casually plunge from one extreme to the other, but at any given time, most species can tolerate relatively wide temperature fluctuations. For example, surface-water temperatures as high as 33.3˚C of a river in Alabama (USA) had no detrimental effects on a community of water striders, giant waterbugs, water scorpions, midge larvae, mayfly nymphs, dragonfly and damselfly nymphs, aquatic beetles, sponge, amphipids and snails from the genera Helisoma and Lymnaea. This indicates that the communities were not biologically depressed. Species composition of benthos at the immediate point of warm water inflow at Varmagjá and Eldvík may be somewhat altered. The ecological stability observed in the Wurtz (1968) study reflects biological equilibrium between benthic fauna and the environment. Macro-invertebrate organisms may include species that can tolerate elevated temperatures, though less adapted ones will be unable to compete.

5.3.3 Effects of heated discharge on fish

The higher temperature at the geothermally influenced sites might lead to displacement of fish, especially arctic charr (Salvelinus alpinus). However, arctic charr were consistently observed in geothermally influenced areas both in winter and summer. This is due to the thorough mixing of geothermal and lake water (Adalsteinsson et al., 1992; Snorrason, 1982). The main factor to be considered in the assessment of temperature effects on fish is the upper incipient lethal temperature. This is the temperature at which 50% of a population of fish species will survive indefinitely and is usually in the range of 24-25˚C (Woodward et al., 2000). Studies of preferred temperatures suggest that brown trout prefer temperatures from 12 to 19˚C with embryonic temperature tolerance of about 1-11˚C (Garrett and Bennett, 1995). Because of efficient cooling by wind action, temperature effects will be so restricted that they will hardly affect the fish in Thingvallavatn in a significant way.

5.4 Effects of geothermal chemicals

Water quality analyses of geothermal separator water and Lake Thingvallavatn shoreline springs indicate arsenic as the most critical trace element to consider for ecotoxicological effects. The concentrations of other elements were consistently low and will have no effects on the lake ecology (Appendix I and Figure 11).

5.4.1 Trace element concentrations in separator water and lake shoreline springs

Arsenic in unpolluted water is usually below 5.0 μg/l (Fergusson, 1990). The level of As in the separator water in 1983 was 5.6-310 μg/l and in 1991 134-160 μg/l (Ólafsson, 1992), and in the year 2000, 20.9 μg/l. Concentrations of arsenic in Varmagjá and Eldvík springs in 1984 were 0.6-0.7 μg/l, but increased to 2.2-4.7 μg/l in 1991 (Ólafsson 1992). The As concentration level was 0.709 and 5.97 μg/l for Varmagjá and Eldvík, respectively, in the year 2000. This was slightly above the 5.0 μg/l *Canadian water quality guidelines for the protection of aquatic life* (CCME, 2001). This concentration is within level III

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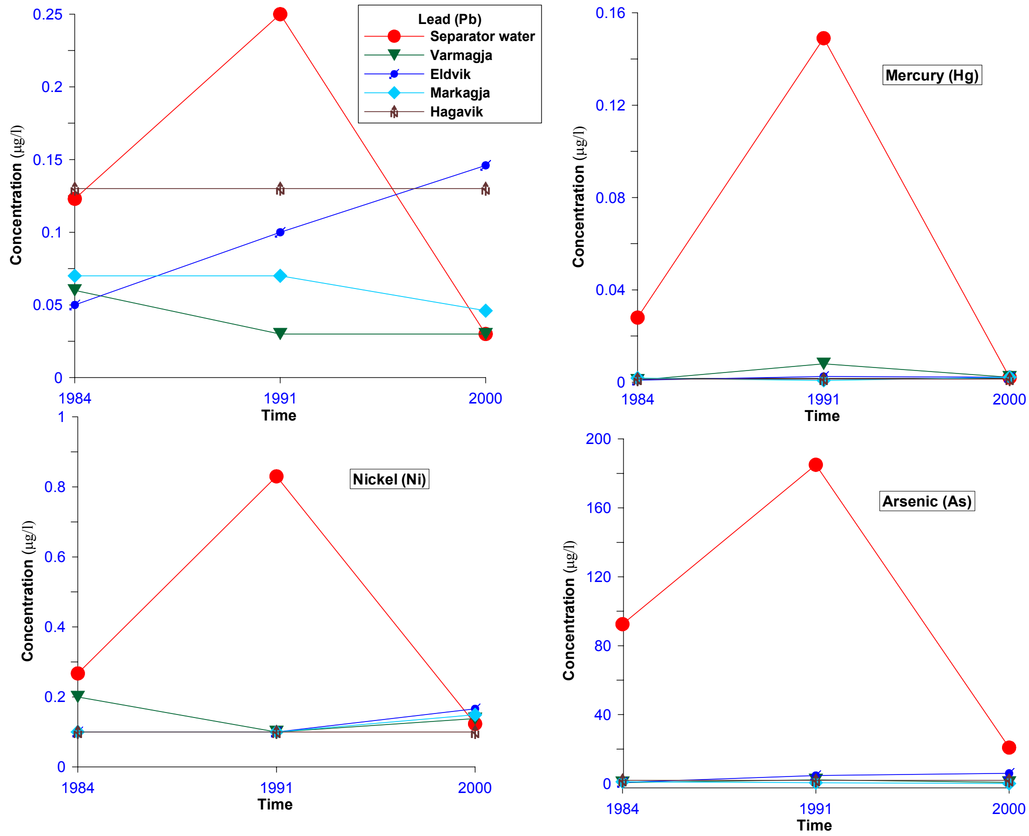

FIGURE 11: Temporal variation of trace elements concentration in Nesjavellir geothermal power plant separator water and Lake Thingvallavatn shoreline springs

µg/l) guidelines of *Icelandic surface water pollution protection regulations* (Government News, 1999). Higher As levels probably stem from geothermal effluents as the separator water had a high As concentration of. Arsenic exists in two states, arsenate (As^{v}) and arsenite (As^{III}). In geothermal fluids arsenic exists as arsenate, which is the thermodynamically stable form of arsenic, less toxic than arsenite. This form, however, can readily be reduced to arsenite (As^{III}) by bacteria associated with blue-green algae (cyano-bacteria) in lake shoreline springs. Its high level makes it significant due to its toxicity to aquatic organisms, which are also affected by other factors such as temperature, pH, organic matter content, phosphates and other water quality parameters.

In 1991, Cadmium concentration in separator water was 0.004-0.72 µg/l (Ólafsson, 1992). The concentration in 2000 was less than 0.005 µg/l. The concentration of Cd at Varmagjá and Eldvík was 0.004-0.006 µg/l in 1984 and 0.04-0.06 µg/l in 1992 (Ólafsson, 1992). The concentration levels at both sampling points was < 0.005 µg/l in the year 2000, which is below the 0.017 µg/l Canadian freshwater quality guideline for protection of aquatic life and within level I (≤0.01 µg/l) of Icelandic surface water pollution protection regulations.

Chromium concentrations were not measured during the 1984-1992 studies on Lake Thingvallavatn. The concentration in separator water was 0.031, 0.479, and 0.46 µg/l at the Varmagjá and Eldvík springs, respectively, in 2000. The concentration is below the Canadian water quality guidelines for protection of aquatic life for trivalent chromium, Cr (III) - 8.9µg/l, and hexavalent chromium, Cr(VI) - 1.0 µg/l, and within level II (0.3-5.0 µg/l) of Icelandic surface water pollution protection regulations. High Cr in the springs could be from natural sources and not from geothermal wastewater, which had very low levels. Chromium is commonly found in surface water in the two oxidation states Cr(III) and Cr(VI), with Cr(VI) being more toxic.

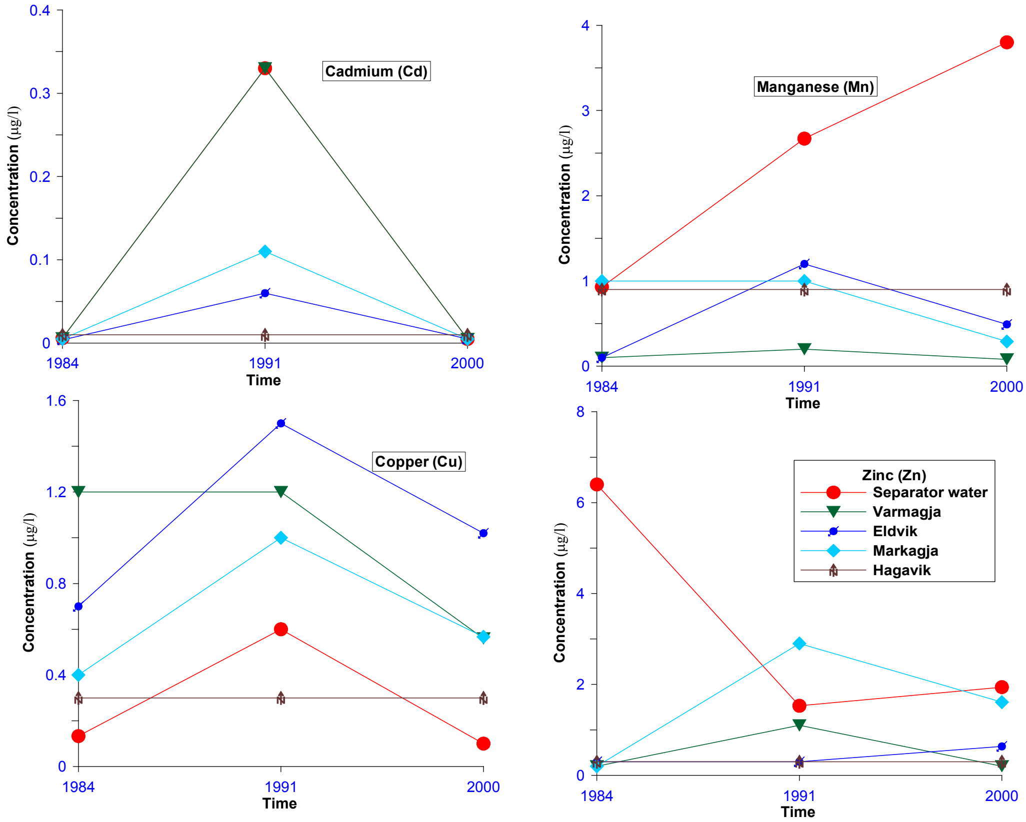

FIGURE 11: Continued

Copper in the separator water was 0.1-3.0 µg/l (1984) and 0.3 µg/l (1991) (Ólafsson, 1992). Copper concentration in separator water in year 2000 was < 0.10 µg/l. The concentration at Varmagjá and Eldvík in 1984 was 0.7-1.2 µg/l and 1.2-1.5 µg/l in 1991(Ólafsson, 1992). Concentration in 2000 was 0.56 and 1.02 µg/l for Varmagjá and Eldvík, respectively. No changes have occurred and the level recorded is below the permitted limit in fresh water (2-4 µg/l) for protection of aquatic life based on Canadian water quality guidelines and within level II (0.5-3.0 µg/l) of Icelandic surface water pollution protection regulations. Enrichment of spring water with Cu could be from an external source but not from geothermal effluent as its concentration in separator water was < 0.10 µg/l.

Nickel concentration in separator water was 0.1-1.3 µg/l in 1984-91 (Ólafsson, 1992), and 0.123 µg/l in 2000. The concentration at Varmagjá and Eldvík was 0.1-0.2 µg/l in 1984-91 (Ólafsson, 1992). Ni concentration in 2000 was 0.138 and 0.166 µg/l for Varmagjá and Eldvík, respectively. There has been no increase and the present concentration is low in comparison with the permissible Ni limit of 25-150 µg/l of Canadian water quality guidelines for protection of aquatic life. It is within level I (≤ 0.7 µg/l) of Icelandic surface water pollution protection regulations.

The concentration of lead was 0.03-0.54 µg/l in separator water in 1983-1991 (Ólafsson, 1992), and <0.030 µg/l in 2000. Pb concentration at Varmagjá and Eldvík in 1984 was 0.05-0.06 and 0.03-0.10 µg/l in 1991 (Ólafsson, 1992). The concentration level in 2000 was < 0.030 and 0.146 µg/l for Varmagjá and Eldvík, respectively. Pb concentration in Lake Thingvallavatn shoreline springs was very low in comparison with the 1-7 µg/l limit of Canadian water quality guidelines for protection of aquatic life. This has no ecological effects on Lake Thingvallavatn. There could be an external source of Pb, especially from dissolution of lead balls used on fishing gears/nets.

Zinc concentration in the separator water was 1.3-15.2 µg/l in 1984, 1.2-1.7 µg/l in 1991 (Ólafsson, 1992), and 1.94 µg/l in 2000. Concentrations at Varmagjá and Eldvík were 0.2-0.3 µg/l in 1984 and 0.3-1.1 µg/l in 1991 (Ólafsson, 1992). Zn levels in 2000 were < 0.20 and 0.636 µg/l at Varmagjá and Eldvík, respectively, all below the 30.0 μg/l limit Canadian water quality guidelines for protection of aquatic organisms and in level I (≤ 0.50 μg/l) of Icelandic surface water pollution protection regulations.

In 1991, Hg varied from 0.044 to 0.261 μg/l (Ólafsson 1992) in separator water, but in 2000 it was <0.0022 μg/l. The concentration at Varmagjá and Eldvík were 0.0025-0.008 μg/l in 1991(Ólafsson, 1992) and less than 0.0022 μg/l in 2000. Mercury exists as elemental Hg in deep geothermal fluids (metallic Hg). Once flashed, most of it ends up in steam which is discharged in condensate as elemental Hg. Elemental Hg is lost to the atmosphere through volatilisation, hence, the biological effects of Hg derived from geothermal wastewater are insignificant. Hg in water is readily oxidized to Hg^{2+} ions, implying that geothermal Hg has less environmental impact in the water column than would be predicted on the basis of mercury behaviour.

Compression of trace element constituents in separator water and geothermally affected springs in the period 1984 - 2000 show that for all elements, except As and Hg, the drillhole water does not differ from the springs, despite considerable variation. The levels of As and Hg were much higher in the drillhole water than in the springs. Mn concentration in geothermal wastewater has been increasing somewhat over the years (Figure 11).

5.4.2 Trace metals in lake sediments

Heavy metals exist in water bodies as dissolved species in water, suspended insoluble chemical solid or component of suspended natural sediments. Suspended sediments and metallic chemical solids are usually deposited at lake bottom after they aggregate to form large, denser than water particles that settle from the water when they can no longer be kept in suspension hence elevating metals concentrations in sediments. Elevated levels of trace elements in sediments can arise from dead aquatic plants, which assimilate the elements and release them in sediment during the decomposition process. When deposited, the elements will be adsorbed on the iron coated sediment particles thus increasing sediment trace element concentration. The ability of aquatic plant to accumulate these trace elements may pose a risk to some benthic organisms especially when released during the decomposition process.

Figure 12 shows trace elements concentration in sediments at the two sampling stations, Thorsteinsvík and the control station Vatnskot in the year 2000. Trace metal concentrations in sediments were low in comparison with levels recorded in unpolluted freshwater lakes in Sweden (SEPA, 1991) and Canada (CCME, 2001). All were within the *Swedish Environmental Protection Agency* (SEPA) quality criteria of metals in sediments of freshwater lakes and watercourses, Icelandic sediment quality guidelines

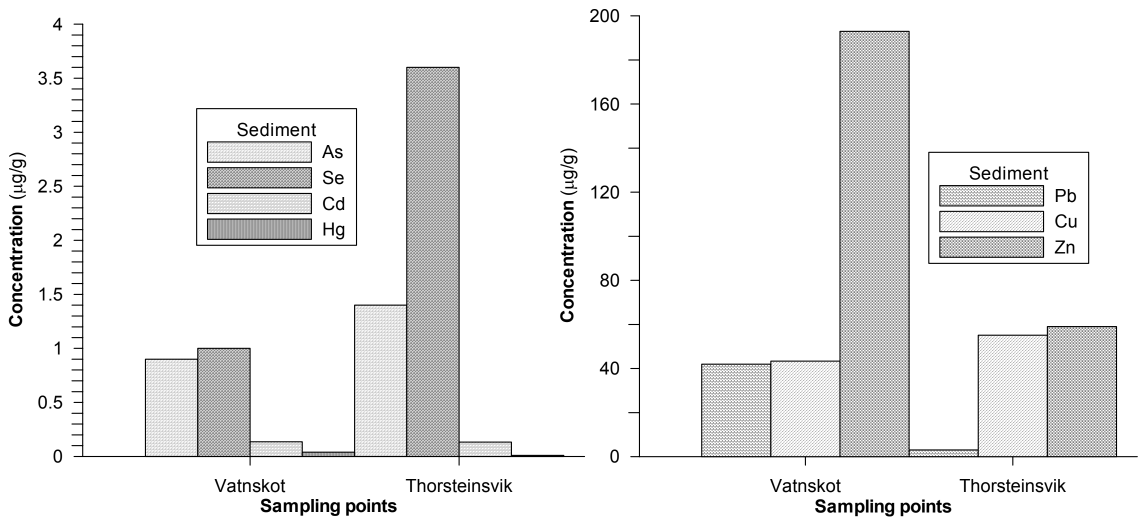

FIGURE 12: Trace element concentration in sediments at Vatnskot and Thorsteinsvik in 2000

(Government News, 1999) and in most cases also within the Canadian sediment quality guidelines for the protection of aquatic life. The latter defines safety limits in terms of *Interim sediment quality guidelines* (ISQGs) derived using threshold effects level (TEL) and probable effect level (PEL) which defines the level above which adverse effects are expected to occur frequently. Threshold effects level on the other hand defines the concentration level below which adverse biological effects are expected to occur rarely (Long et al., 1995). The concentration of Pb and Zn at Vatnskot were above the interim sediment quality guidelines (ISQs) but within the probable effect level (PEL) limit and the same applied for Cu at both stations (Table 9). As geothermal separator water and the geothermally affected spring water (Varmagjá and Eldvík) had very low concentration of these elements elevated sediment levels must be caused by other sources. For example the elevated Pb concentration in the year 2000 sediment sample from Vatnskot is most likely due to dissolution of lead balls from lost fishing gears/nets.

TABLE 9: Sediment quality guidelines for the protection of aquatic life and sediment concentration at Thorsteinsvík and Vatnskot in the year 2000, in mg/kg

	Canadian sediment quality guidelines		Icelandic sediment quality criteria	Thorsteinsvík	Vatnskot
	ISQs	PEL			
As	5.9	17	≤ 8.0*	1.4	0.9
Cd	0.6	3.5	0.11 - 0.30**	0.133	0.136
Cr	37.3	90	≤ 100.0*	26.2	35.3
Cu	35.7	197	40 - 70**	55.1	43.3
Hg	0.17	0.486	0.02 - 0.1**	0.01	0.04
Pb	35	91.3	15 - 50***	3	42
Zn	123	315	≤ 60*	59	193

* Level I; ** Level II; *** Level III.
ISQs: Interim sediment quality guidelines (CCME, 2001);
PEL: Probable effect level (CCME, 2001);
Icelandic sediment quality guidelines (Government News, 1999)

Both As and Hg are found in elevated concentration in water from drillholes (Figure 11) and theoritically one might expect to find eleveted levels of these metals in sediments in Thorsteinsvík. This was not the case and the concentrations of these metals were at the same low levels in Thorsteinsvík and at Vatnskot (the control station). As already discussed As concentrations appear to drop when the geothermal effluent mixes with ground water on the way down to the lake. Most of the elemental mercury in geothermal wastewater is usually lost to the atmosphere through volatilisation and readily oxidized to Hg^2. This means that geothermal mercury is unlikely to have significant biological effects on the Thingvallavatn ecosystem.

5.4.3 Trace elements in biological samples

Most organisms have evolved mechanisms to control absorption of some trace metal pollutants. For example the freshwater algae, *Chlorella vulgaris*, can develop a variety that significantly decreases its rate of Cu absorption, and a tolerant variety of the polychaete, rag worm (*Nereis diversicolor*) which shows lowered permeability to Cu and Zn (Coombs, 1980; Demuynck and Dhainaut-Courtois, 1994). Excretion mechanisms are also important but they vary and are organism dependent. These include:- secretion or diffusion over general body surface via sources such as fish skin mucus; excretion into faecal matter via intestine, liver and gall bladder in fish or via pancreas in crustacea; excretion into urine via the kidneys in bivalves; excretion through the gills; moulting which removes significant amount of metal via the cast shell and sessile bivalves excrete excess iron through the basal discs that permanently attach them to the substratum. In general, developmental stages are critical for sensitivity of invertebrates to trace metals. Embryonic stages have been established to be more sensitive to trace metals in the order Hg > Ag > Zn > Ni > Pb. Both crustaceans and bivalve molluscs show greater resistance to metal contamination with each successive developmental stage (Rand and Petrocelli, 1985). In comparison to freshwater fish and invertebrates, aquatic plants are equally or less sensitive to Cd, Cu, Pb, Hg and Zn. Water resources thus should be managed for the protection of fish and invertebrates.

In general, the concentration levels of trace metals in the biological samples were low and the temporal variation observed must be ascribed mostly to random, natural variability (Figures 13 and 14). In no case, except for Cu in the small benthivorous arctic charr, do samples from Thorsteinsvík show significantly higher trace metal levels than samples from the control station at Vatnsvík. Hence it can be concluded that the littoral community of Thingvallavatn in the Nesjavellir area shows no signs of changes in trace metal levels. The higher Cu levels in small benthivorous arctic charr is not readily explained. There is some interesting variation seen in Vatnskot. The 1995 sample of *Myriophyllum* showed a level of Pb that was approximately hundredfold higher than normal. The sporadic nature of this finding and the incidental elevation of Pb in the sediment sample from 2000 suggests that lead may be dissolving slowly from point sources consisting of lead balls and lead ribbons in fishing gear that has been lost in the lake. The high levels of Mn seen in samples of *Myriophyllum* and *Lymnaea* in Vatnskot may also stem from some kind of pollution at Vatnskot. In this respect it should be noted that the old farm at Vatnskot was situated near the sampling station.

6. CONCLUSIONS AND RECOMMENDATIONS

6.1 Conclusions

- Except for arsenic the trace element concentration levels in the Nesjavellir geothermal power plant wastewater are within the accepted guidelines of Icelandic laws and regulation on surface water pollution protection, and international environmental quality guidelines such as Canadian environmental quality guidelines and Swedish Environmental protection Agency (SEPA) guidelines on protection of watercourses and lakes. Although mercury levels in separator water were slightly elevated its ecological effects are not significant as most of it ends up in the atmosphere once discharged, with insignificant amounts left in the wastewater.
- A temperature range of 49-100˚C for discharged wastewater is particularly high at the point source of disposal but as it flows towards the lake, it blends with cold underground water, attaining moderate temperature before reaching Lake Thingvallavatn.
- There was no significant trace element pollution in springs below the power plant. Observed changes in concentration levels are attributed to the natural cycle of the water chemistry. Trace element concentrations in Lake Thingvallavatn shore springs water especially in Varmagjá and Eldvík, within the geothermal groundwater zone show small variations though levels are slightly elevated in comparison with springs outside the geothermal flow zone such as Markagjá. From an ecotoxicological point of view, arsenic seems to be the only constituent of geothermal effluents that may be of concern to the ecology of Lake Thingvallavatn. Arsenic concentrations at Eldvík were slightly above the recommended 5.0 μg/l in Canadian guidelines limit for protection of aquatic life, by 0.97μg/l. A spring water temperature range of 19.7-26˚C vis-à-vis the 49-100˚C temperature of separator water, shows efficient cooling during the flow towards Lake Thingvallavatn. The blending of geothermal inflow with lake water at the springs is further enhanced by wind driven currents very common in the lake.
- As chemical effluents are diluted by groundwater on the way to the lake, and further dispersed by wind driven currents in the lake, any effects on Lake Thingvallavatn biota are expected to be highly localized. The same principle is true for the heated discharge from the plant.
- In general, the temporal variation in trace element concentrations in biological samples was small. Elevated Pb levels in some biological samples from Vatnskot (not influenced by geothemal inflow) could be from the dissolution of lead balls on fishing gear/nets.
- There is potential for contamination of fresh underground water by geothermal waste at Grámelur, shown by variations in the temperature profile of experimental drillhole NL-08.

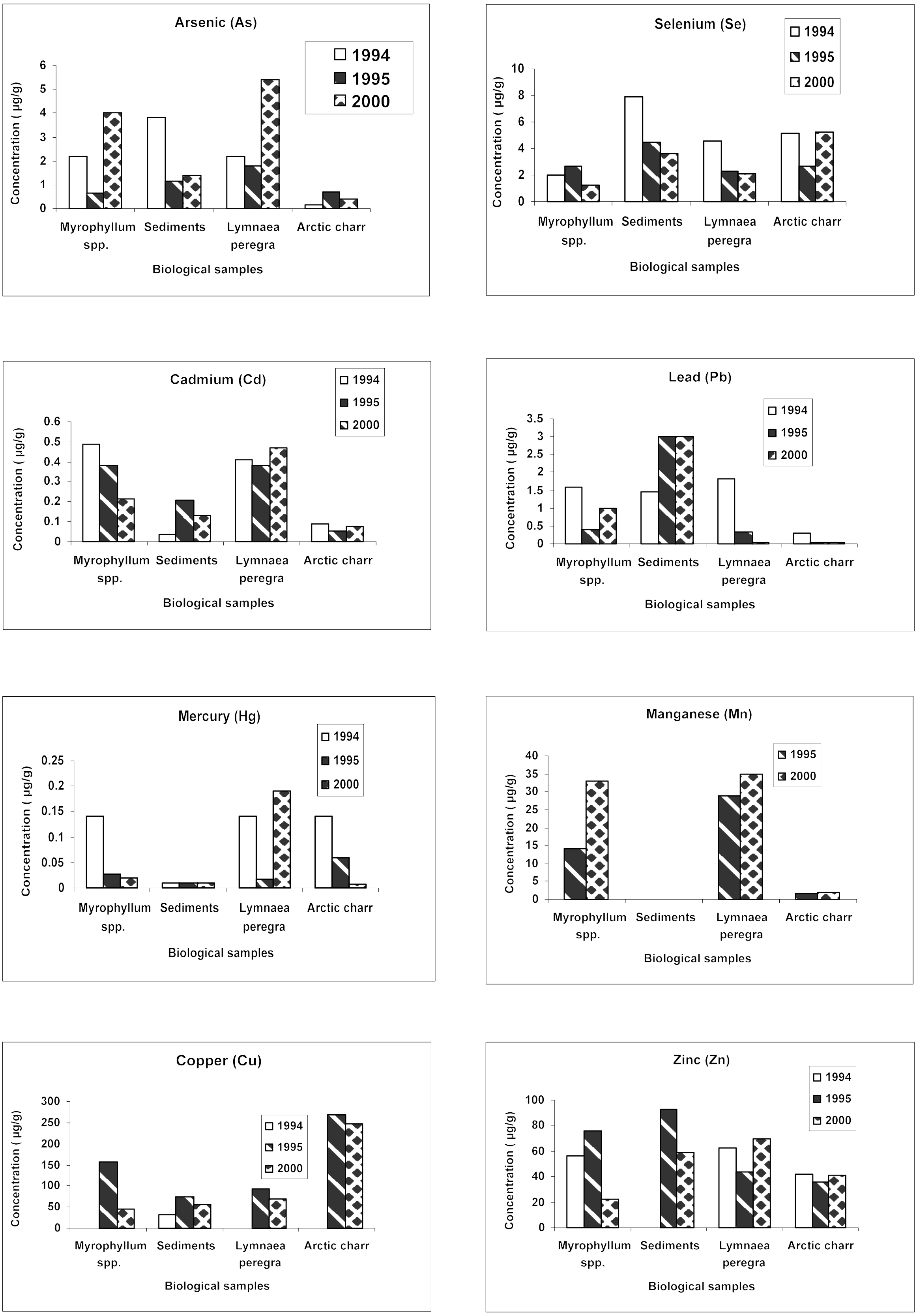

FIGURE 13: Temporal variations in trace elements concentration in biological samples at Thorsteinsvík

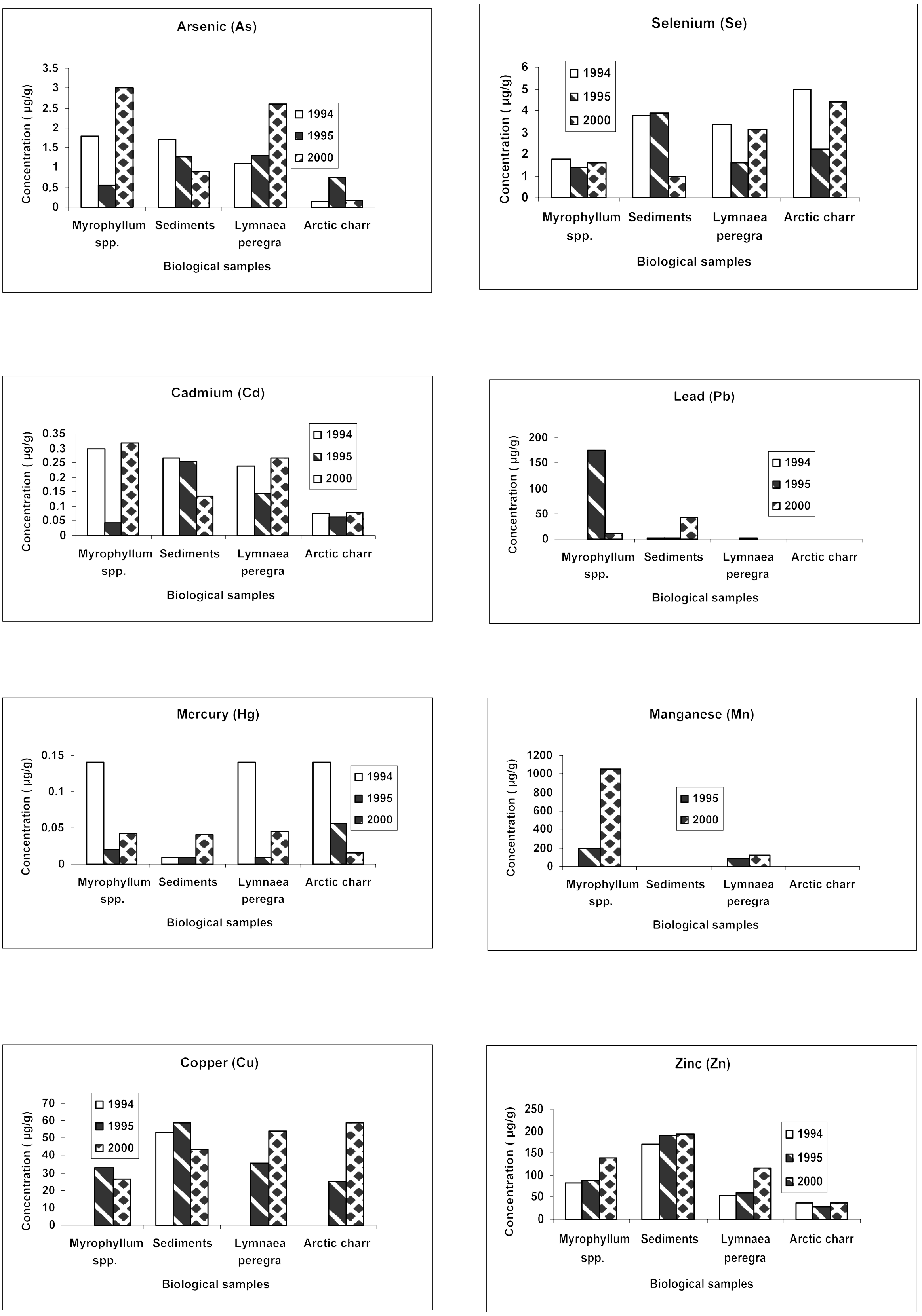

FIGURE 14: Temporal variations in trace elements concentration in biological samples at Vatnskot

6.2 Recommendations

- As part of an environmentally sound practice trace elements should be monitored regularly as long as geothermal wastewater is released into shallow drillholes that connect to the groundwater system or into the nearby Nesjavellir stream.
- Monitoring of trace elements in Lake Thingvallavatn shoreline springs should be incorporated into the monitoring programme.
- Assessment is needed of the concentration level of selenium in geothermal wastewater and Lake Thingvallavatn shoreline springs to authenticate its source in the biological samples.
- Deep reinjection of the discharged geothermal wastewater is recommended.
- Cooling water should be recycled to reduce the amount of heated discharge released into the Nesjavellir stream.

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APPENDIX I: Analytical data for the Nesjavellir geothermal power plant and the Lake Thingvallavatn samples

TABLE 1: Average concentration of trace elements in separator water (Nesjavellir geothermal power plant) and Lake Thingvallavatn shoreline springs in 2000, in µg/l

	Separator water	Varmagjá	Eldvík	Markagjá	Grámelur pumping station
Phosphorus, PO_4 -P	<5.00	71.3	74.4	37.7	43.1
Ammonia, NH_4 -N	#	<0.8*	<0.8*	<0.8*	<0.8*
Nitrogen, NO_3 -N	#	49*	67*	90*	56*
Zinc, Zn	1.94	<0.200	0.636	1.61	28.4
Lead, Pb	<0.0300	<0.0300	0.146	0.046	0.054
Cadmium, Cd	<0.0050	<0.0050	<0.0050	<0.0050	<0.0050
Copper, Cu	<0.100	0.56	1.02	0.567	1.6
Manganese, Mn	3.8	0.08	0.49	0.29	0.33
Nickel, Ni	0.123	0.138	0.166	0.149	0.561
Mercury, Hg	<0.0022	<0.0022	<0.0022	<0.0022	<0.0022
Arsenic, As	20.9	0.709	5.97	0.035	3.82
Chromium, Cr	0.031	0.479	0.46	0.287	0.31

* Analysis by Marine Research Institute # Not detected by method used at MRI

TABLE 2: Average concentration of major constituent elements in separator water and Lake Thingvallavatn shoreline springs in 2000, in mg/l

	Separator water	Varmagjá	Eldvík	Markagjá	Grámelur pumping station
Silica (SiO_2)	807	49	64	14	40
Sodium (Na)	140	21.1	27.9	8.85	12.6
Potassium (K)	29	3.31	3.81	0.906	1.5
Calcium (Ca)	0.2	14.8	14.1	5.72	8.1
Magnesium (Mg)	0.005	6.7	6.93	3.9	4.7
Sulphate (SO_4^{2-})	7.9	27.8	26.4	4.28	12.2
Aluminium (Al)	1.67	0.00154	0.349	0.0149	0.291

TABLE 3: Winter and summer water temperature of Lake Thingvallavatn shoreline monitoring springs (1990-2000)

	1990	1991	1992	1993	1994	1995	1996	1997	1998	1999	2000
Varmagjá	19.8/24.3	21.7/23.1	24.7	21.9/21.9	21.0/23.1	21.5/21.8		23.1/23.1	21.7/22.6	20.3	21.8/25.1
Eldvík	20.6/23.9	21.0/22.9	25.0	21.9/21.9	21.0/23.1	21.5/21.9		23.0	21.7/22.9	20.6	22.2/25.4
Markagjá	22.0/24.3	21.1/22.4	24.0	21.9/21.9	21.0/23.1	21.1/21.8		22.6/22.9	21.3/22.4	25.1	21.6
Markatangi	23.8	23.6	24.1	21.9/21.9	22.6		21.6/23.3	23.4	23.4	25.2	
Sigguvík	20.8/24.3	23	26.0	21.9/21.9	21.0/23.1		21.6/23.3	23.1	21.8	22.0	22.5/25.6
Stapavík	23.8/24.3	23.2	24.2	21.9/21.9	22.9		21.8/23.4	23.2	23.2	25.2	25.6
Nesjalækur	22.4/24.2	22.6	24.2	20.0/21.9	21.0/23.1		20.6/23.2	22.6/23.2	19.7/22.4	25.3	21.3
Grámelur			25.0	21.9			21.7/23.3	23.2/23.4	21.8/23.5		22.6/25.5

Blank cells means temperature readings not taken

TABLE 4: Average concentrations and standard deviation (lower figure) of trace elements in liver of small benthivorous charr (wet weight samples) (Snorrason and Jónsson, 1995, 1996 and 2000)

	Dry weight (%)	As (ng/g) n=4	Se (µg/g) n=4	Zn (µg/g) n=4	Cu (µg/g) n=4	Cd (ng/g) n=4	Pb (ng/g) n=4	Hg (ng/g) n=9	Mn (µg/g) n=4
Vatnskot '94	22.21	153	4.96	37.3		76.8	≤300	26.0	
	0.04	7	0.34	2.1		5.0		2.6	
Thorsteinsvík '94	21.67	143	5.16	42.2		88.7	≤300	18.1	
	0.01	17	0.06	2.4		2.9		5.5	
Vatnskot '96									
Group 1 (3.7 yrs)		<750	2.71	31.6	32.2	<65	40	93.6*	1.82
			0.24	1.3	0.2		6	43	0.16
Group 2 (6.7 yrs)		<750	1.78	24.3	17.6	65	116	19.3	1.74
			0.06	0.7	0.1	27	46	2.6	0.11
Thorsteinsvík '96									
Group 1 (5.6 yrs)		506	2.88	37.4	170.5	42.1	20	106.6*	1.88
		14	0.07	0.7	1.9	9.6	5	23.0	0.05
Group 2 (9.1 yrs)		926	2.48	33.9	364.9	61.4	44	12.0	1.63
		113	0.09	0.2	1.9	3.6	7		0.03
Vatnskot 2000									
Group 1 (8.9 yrs)		200	3.9	35.4	84	108	< 20	18	3.55
		30	0.1	1.4	2	1		1	0.15
Group 2 (3.8 yrs)		150	4.9	40.0	34	49	< 20	13	1.94
								1	
Thorsteinsvík 2000									
Group 1 (5.6 yrs)		360	5.0	39.0	296	91	<20	7	1.54
		60	0.3	1.6	5	4		1	0.07
Group 2 (4.0 yrs)		390	5.2	43.7	244	59	<20	6	2.35
		50	0.1	1.1	2	1		1	0.04
Group 3 (2.9 yrs)		370	5.4	40.8	200	74	60	7	1.71
		30	0.3	1.7	3	8	1	1	0.08

* Sample contamination or faulty measurements

TABLE 5: Average concentrations and standard deviation (lower figure) of trace elements in aquatic plant, gastropod snail and sediments (Snorrason and Jónsson, 1995, 1996 and 2000)

	Hg (µg/g) n=4	**As (µg/g) n=4**	**Se (µg/g) n=4**	**Zn (µg/g) n=4**	**Cu (µg/g) n=4**	**Cd (µg/g) n=4**	**Pb (µg/g) n=4**	**Mn (µg/g) n=4**
Aquatic plant - *Myrophyllum alterniflorum*								
Vatnskot '94	<0.14	1.8 0.1	1.8 0.2	81.3 1.6		0.30 0.02	0.7 0.2	
Thorsteinsvík '94	<0.14	2.2 0.2	2.0 0.3	56.2 7.3		0.49 0.04	1.6 0.3	
Vatnskot '95	0.0201 0.0022	0.558 0.036	1.39 0.04	87.0 1.4	32.7 0.04	0.0452 0.0156	175	190.8 4.8
Thorsteinsvík '95	0.0283 0.0054	0.644 0.075	2.63 0.23	75.7 0.3	157.9 11.6	0.3785 0.0281	0.38 0.06	14.2 0.4
Vatnskot '00 (n=3)	0.042 0.001	3.0 0.1	1.61 0.01	140 16	26.8 3.8	0.320 1	10.8 0.2	105512 6
Thorsteinsvík '00 (n=3)	0.021 0.001	4.0 0.4	1.20 0.02	22 1	45.3 8.9	0.216 14	1.1 0.1	33 5
Gastropod snail - *Lymnaea peregra*								
Vatnskot '94	<0.14	1.1 0.2	3.4 0.3	53.9 3.1		0.24 0.01	≤0.3	
Thorsteinsvík '94	<0.14	2.2 0.2	4.6 0.2	62.4 2.1		0.41 0.03	1.8 0.3	
Vatnskot '95	0.0095 1.0	1.3 0.2	1.58 0.12	60.5 1.7	35.4 1.2	0.144 0.030	1.22 0.13	85.3 3.3
Thorsteinsvík '95	0.0170 4.0	1.8 0.3	2.27 0.08	43.4 1.6	92.3 4.5	0.378 0.028	0.34 0.18	28.9 0.8
Vatnskot '00 (n=3)	0.05	2.6 0.1	3.13 0.07	116 4	54 1	0.265 5	0.8 0.1	118 1
Thorsteinsvík '00 (n=3)	0.191	5.4 0.1	2.10 0.02	70 1	682	0.468 5	<0.03	35 2
Lake sediments								
Vatnskot '94	0.01	1.7	3.8	170	53.2	0.268	1.82	
Thorsteinsvík '94	0.01	3.8	7.9	<0.01*	32.1	0.04	1.45	
Vatnskot '95	0.01	1.3	3.9	190.66	58.56	0.255	3.07	
Thorsteinsvík '95	0.01	1.13	4.5	92.66	74.26	0.208	2.99	
Vatnskot '00	0.04	0.9	1.0	193	43.3	0.136	42	
Thorsteinsvík '00	0.01	1.4	3.6	59	55.1	0.133	3	

* = Faulty measurements

TABLE 6: Flow rate of wastewater from the Nesjavellir geothermal plant

Type of discharge	**Flow rate (kg/s)**		**Discharge temperature (˚C)**	
	2000	**2001**	**2000**	**2001**
Unused brine	41.39	48-143	100	100
Used brine	116.2	115	54.83	46
Condensate	119.9	126-140	65.47	46-68
Cooling water	489.1	343-1776	65.8	49-69