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Long-term modification of Arctic lake ecosystems: Reference condition, degradation under toxic impacts and recovery (case study Imandra Lakes, Russia)

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Abstract

In this study, published data on Lake Imandra, north-west Russia, have been synthesised to investigate trends in lake contamination and recovery due to changing inputs of heavy metals and nutrients over time. Records of water chemistry, phytoplankton, zooplankton and fish communities have been used to determine the status of aquatic ecosystem health in three distinct phases of Lake Imandra's recent history. Firstly, background (reference) conditions within the lake have been established to determine lake conditions prior to anthropogenic influences. Secondly, a period of ecosystem degradation due to anthropogenic inputs of toxic metals and nutrients has been described. Finally, evidence of lake recovery due to recent decreases of toxic metals and nutrients has been explored. Pollution of Lake Imandra began in the 1930s, reaching a peak in the 1980s. Increases in heavy metal and nutrient inputs transformed the typical Arctic ecosystem. During the contamination phase, there was a decrease in Arctic species and in biodiversity. During the last 10 years, pollution has decreased and the lake has been recolonised by Arctic water species. Ecosystem recovery is indicated by a change of predominant species, an increase in the individual mass of organisms and an increase in the biodiversity index of plankton communities. In accordance with Odum's ecosystem succession theory, this paper demonstrates that the ecosystem has transformed to a more stable condition with new defining parameters. This illustrates that the recovery of Arctic ecosystems towards pre-industrial reference conditions after a reduction in anthropogenic stresses occur, although a complete return to background conditions may not be achievable. Having determined the status of current ecosystem health within Lake Imandra, the effect of global warming on the recovery process is discussed. Climate warming in Arctic regions is likely to move the ecosystem towards a predominance of eurybiontic species in the community structure. These organisms have the ability to tolerate a wider range of

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environmental conditions than typical Arctic inhabitants and will gain advantages in development. This indicates that the full recovery of Arctic ecosystems in a warming climate may not be possible.

Keywords: Arctic; Aquatic ecosystem; Toxic impacts; Modification; Reference condition; Degradation; Recovery

Introduction

Understanding the impact of anthropogenic contamination on aquatic ecosystems, and also their subsequent recovery as a result of decreasing anthropogenic stress, is important for successful environmental management. The recovery of aquatic ecosystems due to decreasing anthropogenic inputs, including toxic pollutants, has been well-documented in the scientific literature (e.g. Gunn et al. 1995; Jeppesen et al. 2005; Cairns 2005; Harris et al. 2006; Hobbs 2007; Palmer et al. 2007). The impact of anthropogenic pollution on aquatic environments varies greatly across different climatic regions. Arctic ecosystems are particularly vulnerable due to cold water, simplified food webs, low biodiversity, rapid transfer of material through trophic levels, and the stenobiontic character of aquatic species. This can result in the rapid migration of pollutants throughout food webs causing severe damage to ecosystems. However, knowledge about the recovery of these ecosystems is limited and further information is required on whether the recovery to natural characteristics is possible, which reference conditions should be taken for the purposes of ecosystem recovery and whether climate change will affect the recovery process. The changes that are currently occurring in aquatic ecosystems often have analogues in the past that can be used to predict different scenarios of future change. Harris et al. (2006) emphasised the significance of studying recovery mechanisms in order to predict the ecosystem state.

This study of Lake Imandra provides a unique opportunity to explore the anthropogenic modification of an Arctic ecosystem. Lake Imandra is situated within the Arctic Circle in the Kola Peninsula, Russia. The lake has an area of 880 km^2 with a catchment area of 12,300 km^2, the maximum depth is 67 m and mean depth is of 13 m. The lake has a complex shoreline and consist of three main basins connected by narrow passages (Fig. 1). It has been subject to greater levels of pollution than many Arctic lakes. Industrial development of copper- and nickel-rich apatite–nephelinite and iron deposits in the catchment area of Lake Imandra began in the 1930s. Considerable industrial expansion in the early 1900s resulted in the building of large industrial enterprises in the lake catchment. Large amounts of pollutants entered the lake between 1940 and 1990; the catchment area was also was polluted by airborne contaminants. The main pollutants were heavy metals (predominantly nickel and copper), sulphates, chlorides and nutrients. The main pollution occurred in the northern part of the lake (Bol'shaya Imandra). Since 1990, as a result of the economic crisis in Russia, anthropogenic pressure on the lake has decreased. The recent recovery of the economy has occurred simultaneously with technological modernisation and tighter controls of pollutant emissions into the lake and the atmosphere.

The exploitation of mineral resources in Arctic regions has increased in recent years which makes the case of Lake Imandra important. It is a useful case study in environment management, particularly the avoidance of negative impacts of Arctic mineral resources exploitation.

This study aims to clarify the main changes in the lake's ecosystem over time (the change from background conditions through a period of degradation to a recent recovery) using retrospective analysis of ecosystem dynamics.

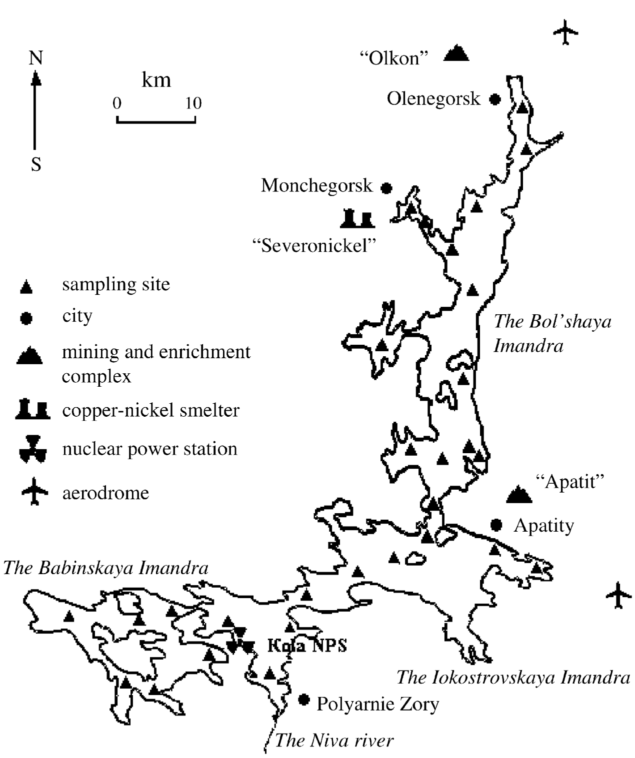

Fig. 1. A map of Lake Imandra showing the distribution of sampling sites, cities and industrial enterprises ('Olkon' complex specialises in iron ore, 'Apatit' complex specialises in apatite–nepheline ore).

A considerable body of work exists on the lake and this paper is based on an analytical review of published studies: Vereshagin (1930), Krogius (1931), Rikhter (1934), Poretskij et al. (1934), Voronikhin (1935), Krokhin and Semenovich (1940), Berg and Pravdin (1948), Galkin et al. (1966), Petrovskaya (1966), Dol'nik and Stalmakova (1975), Den'gina (1980), Moiseenko and Yakovlev (1990), Yakovlev (1998), Iliyashuk (2002a, b), Moiseenko (1999), Moiseenko et al. (1999), Moiseenko et al. (2002), Sharov (2002), Vandish (2000, 2002). Although much information is available, there has been no continuous long-term monitoring of the lake and so this paper is based on discontinuous information. Detailed descriptions of methods of investigation have been given in the literature cited above and so is not repeated here. In this review attention is focused on the main parameters of water chemistry and indicators of phytoplankton, zooplankton, benthos and fish condition, which reflect ecosystem changes during periods of degradation and recovery in relation to reference condition. This work is focused mainly on the condition of the Bol'shaya Imandra basin.

Reference condition

Knowledge of a reference condition (the ecological condition found at undisturbed or minimally disturbed sites) is important when trying to manage anthropogenic stress. The background conditions of the lake prior to industrialisation provide a benchmark for water quality and ecosystem recovery. An estimation of the reference condition of Lake Imandra has been made based on spatial and temporal analysis of lake ecosystem parameters. In order to determine realistic reference conditions, datasets for Lake Imandra published since the 1930s have been used (Vereshagin 1930; Rikhter 1934; Poretskij et al. 1934; Voronikhin 1935; Krokhin and Semenovich 1940), along with measurements on similar lakes in the region that are remote from the influence of industrial activity (Moiseenko and Yakovlev 1990). Hydrochemical lake parameters were close to natural parameters shown in Table 1. Water inhabitants were typical Arctic cold-water species. Table 2 gives the dominant species present and Table 3 shows the main indicators of phytoplankton, zooplankton and benthos condition.

Water chemistry

The climatic and geophysical conditions of the Arctic (predominance of atmospheric precipitation, low temperatures, thin layer of soil and impoverished plant life, slow chemical weathering processes and slow element cycling) form clear, oligotrophic waters (sum of ions 20–30 mg/l). Prior to the 1930s, Lake Imandra was a

Table 1. Water chemistry (basic ions, nutrients and trace elements) of the Bol'shaya Imandra during the period under review

Variable	1930 and earlier (Reference condition)	1983–1992 years		1998–2003 years	
		$M \pm m$	Min–max	$M \pm m$	Min–max
pH	6.4–7.2	7.31±0.35	6.30–8.16	7.37±0.17	6.88–7.70
Σ ions (mg/l)	20–30	82.7±27.8	13.3–176	81.3±9.1	67.0–112
Ca (mg/l)	1.6–4.0	4.80±1,83	1.31–15.0	3.73±0.20	3.43–4.14
Mg (mg/l)	0.5–1.3	1.61±0.74	0.46–5.11	1.18±0.12	0.97–1.44
Na (mg/l)	–	20.7±8.7	2.6–62.0	17.5±3.2	13.6–30.0
K (mg/l)	–	3.18±1.58	0.42–10.9	2.41±0.37	1.64–3.10
HCO_3 (mg/l)	13–18	21.1±8.6	2.4–70.5	22.3±2.9	12.7–27.2
SO_4 (mg/l)	1–3	30.4±10.1	4.0–64.5	28.8±5.8	22.7–50.8
Cl (mg/l)	1.4–1.8	8.1±5.1	1.0–48.0	5.4±1.1	4.3–10.3
PO_4 (μgP/l)	0–8	21±32	2–154	2±3	0–12
P_{tot} (μg/l)	2–10	26±29	2–176	26±14	13–68
NH_4 (μgN/l)	0–20	41±19	26–75	23±20	2–76
NO_3 (μgN/l)	0–35	102±243	1–1271	19±38	1–158
N_{tot} (μg/l)	10–100	436±386	164–1925	195±84	106–402
Si (mg/l)	0.3–0.6	1.05±1.11	0.02–5.15	0.42±0.33	0.10–1.45
COD (mgO/l)	3–6	3.3±1.1	0.9–9.9	2.9±0.4	2.3–3.8
Ni (μg/l)	$\leqslant 1$	43±32	4–150	11±5	7–27
Cu (μg/l)	$\leqslant 1$	14±22	0–165	6±2	4–10
Sr (μg/l)	$\leqslant 30$	57±22	15–149	58±11	32–72
Al (μg/l)	$\leqslant 10$	58±109	0–540	55±34	14–145
Fe (μg/l)	$\leqslant 15$	34±67	0–645	34±14	9–60
Mn (μg/l)	$\leqslant 5$	12±12	0–75	13±8	6–35
Zn (μg/l)	$\leqslant 2$	16±18	1–113	3±3	1–15

Table 2. Dominating complexes of community structure of the Arctic Lake Imandra during the key periods of ecosystem modification

Communities	Periods		
	Reference condition	Intensive pollution and degradation	Decreasing pollution and revitalisation
Phytoplankton	*Aulacoseira distans*, *A. italica*, *A. islandica*, *Asterionella formosa*, *Cyclotella comta*, *Tabellaria fenestrata*, *Dinobryon divergens*, *Anabaena* sp.[a]	*Aulacoseira islandica, Rhizosolenia*, *Eudorina elegans*, *Pandorina morum*, *Cryptomonas* sp., *Stephanodiscus* sp., *Asterionella formosa*, *Sphaerocyctis schroeteri*[b]	*Cryptomonas* sp., *Stephanodiscus* sp., *A. islandica*, *Cyclotella* sp., *Synedra* sp., *Asterionella formosa*, *Pandorina morum*, *Anabaena* sp., *Limnothrix planctonica*, *Tabellaria fenestpate*[i]
Zooplankton	*Kellicottia logispina, Conochilus unicornis*, *C. hippocerpis*, *Bosmina obtusirostris*, *Daphnia longispina v. hyalina*, *Cyclops scutifer*, *Eudiaptomus gracilis*[c]	*Synchaeta pectinata*, *Keratella cochlearis*, *Polyarthra* sp., *Bipalpus hudsoni*, *Filinia* sp., *Bosmina obtusirostris, Mesocyclops* sp., *Cyclops* sp., *Eudiaptomus* sp.[d]	*Asplanhna priodonta*, *Polyarthra vulgaris*, *Bipalpus hudsoni*, *Keratella cochlearis*, *K. quadrata*, *Kellicottia longispina*, *Notholca* sp., *Bosmina obtusirostris*[i]
Macrozoobenthos	*Trissocladius paratatricus*, *Tanytarsus* spp., *Procladius* spp., *Monoporeia affinis*, *Limnodrilus hoffmeisteri*[e]	*Limnodrilus hoffmeisteri*, *Euiliodrilus hammoniensis Tubifex tubifex*, *Chironomus* spp.[f]	*Monoporeia affinis*, *Tubifex tubifex*, *Chironomus* spp., *Tanytarsus* spp.[g]
Ichthyofauna	*Salmo trutta* (L.), *Coregonus lavaretus* (L.), *C. albula* (L.), *Salvelinus alpinus* (L.) *Thymlus thymallus* (L.)[h]	*C. lavaretus C. albila*, *Esox lucius*, *Perca fluviatilis*, *Phoxinus phoxinus* (L.)[f]	*Coregonus lavaretus*, *C. albula*, *Esox lucius*, *Perca fluviatilis*, *Osmerus eperlanus* (L.)[i]

The table is compiled from the following dates:
[a]1930: Voronikhin (1935), Poretskij et al. (1934).
[b]1981–1987: Moiseenko and Yakovlev (1990), Sharov (2002).
[c]1930: Krokhin and Semenovich (1940).
[d]1981–1987: Moiseenko and Yakovlev (1990), Vandish (2000, 2002).
[e]1930: Krogius (1931), Krokhin and Semenovich (1940).
[f]1978–1996: Moiseenko and Yakovlev (1990), Yakovlev (1998).
[g]1998: Iliyashuk (2002b).
[h]1930: Krogius (1931).
[i]Unpublished data (2003).

typical oligothrophic lake with hydrocarbonate – sodium salt contents, low concentrations of suspended material (0.7–1.0 mg/l) and microelements ($<1\,\mu g/l$). Total phosphorus was less than $2\,\mu g/l$, and phosphates produced in the period of vegetation growth were completely utilised in production processes (Table 1). Water transparency was about 8 m. The lake water was characterised by high oxygen saturation (to the bottom) due to the inflow of water from mountain ice-free rivers and there was no stratification owing to wind turbulence during the ice-free period.

Phytoplankton

In the pre-industrial period of the lake's history, Voronikhin (1935) noticed the abundance of desmids and diatoms, particularly of the genera *Cosmarium* and *Staurastrum*. The most widespread species of plankton were: *Ceratium hirundinella*, *Quadrigula closterioides*, *Spondylosim planum*, *Stichogloea olivacea*, *Botryococcus braunii*, *Coelosphaerium naegelianium*, *Anabaena lemmermannii*; and of periphyton were *Oocystis grandum*, *Spirogyra* sp., *Mougeotia* sp., *Chlorococcus turgidus*, *Chlorococcus minutus*, *Nostoc* sp. Poretskij et al. (1934) investigated diatoms in three large lakes in the Kola Peninsula, including fossil flora of sediments. They give a list of 412 species and forms of diatoms.

The analysis of algal composition showed significant abundance and diversification of diatoms, with the predominance of northern-alpine forms including *Aulacoseira alpigena*, *Aulacoseira distans*, *Aulacoseira valida*, *Aulacoseira subarctica*, *Cyclotella bodanica* and *Cyclotella schumanii*. In areas of the lake situated at a distance from industrial sources, the phytoplankton was represented mainly by diatomaceous, dinophyceans and Chrysophycea algae. Typically, the dominant species by biomass were *Peridinium cinctum*, *Peridinium aciculiferum*, *Dinobryon bavaricum*, *Aulacoseira italica*, *Tabellaria fenestrata*, *Asterionalla formosa*, *C. hirundinella* and *Rhizosolenia longiseta*.

Table 3. The main indicators of community conditions during the key periods of Bol'shaya Imandra lake ecosystem modification

Indicators	Reference condition	Intensive pollution and degradation	Decreasing pollution and revitalization
Phytoplankton	1930 and earlier[a]	1994–1996 (august)[b]	2003 (august)[h]
Chlorophyll "*a*" (mg/m^3)	0.1–0.5	3.8 ± 0.9	3.6 ± 0.4
Biomass (g/m^3)	0.3–0.5	3.6 ± 0.5	3.4 ± 0.3
Number ($\times 10^6$ cells/l)	0.01–0.3	3.8 ± 0.3	2.2 ± 0.1
The information index of the species (Shannon's Index), H_{bit}	3.0–3.5	2.5 ± 0.2	3.1 ± 0.2
The mean individual weight of alga, (B/N) 10^{-6} g	2.0–2.5	0.9 ± 0.2	1.5 ± 0.2
%B *Stephanodiscus*	0.5	6 ± 1.7	8 ± 0.7
%B *Cryptomonas*	0.3	17 ± 2.5	15 ± 1.3
%B Bluegreen	2–4	9 ± 1.2	11 ± 0.5
%B Green	5.1	21 ± 1.9	20 ± 0.7
Zooplankton	1930 and earlier[c]	1994–1996 (August)[d]	1998–2003 (August)[h]
Biomass (d.w.w. g/m^3)	0.2–1.0	1.71 ± 1.07	1.21 ± 0.91
Number ($\times 10^3$ specimens/m^3)	10–100	271.3 ± 139.2	107.14 ± 99.52
The information index of the species (Shannon's Index), H_{bit}	2.5–3.0	2.3 ± 0.5	2.53 ± 0.36
The mean individual weight of zooplankters (B/N) ($\times 10^{-3}$ g)	0.01–0.02	0.006 ± 0.002	0.012 ± 0.003
% B Rotatoria (in total biomass of zooplankton)	10–20	44 ± 17	20 ± 11
Macrozoobenthos	1930 and earlier[d,e]	1978–1986 (october)[f]	1998 (october)[g]
Biomass (g/m^2)	0.3–0.8	21.3–49.0	12.9 ± 6.7
Number ($\times 10^3$ specimens/m^2)	0.4–0.7	23.9–62.7	6.2 ± 5.4
The information index of the species (Shannon's Index), H_{bit}	3.5	1–2.2	1.1

The table is compiled from the following dates:
[a]Voronikhin (1935), Poretskij et al. (1934).
[b]Moiseenko and Yakovlev (1990), Sharov (2002).
[c]Krokhin and Semenovich (1940).
[d]Moiseenko and Yakovlev (1990), Vandish (2000, 2002).
[e]Krogius (1931), Krokhin and Semenovich (1940).
[f]Moiseenko and Yakovlev (1990), Yakovlev (1998).
[g]1998: Iliyashuk (2002b).
[h]Unpublished data (2003).

The biomass of phytoplankton in typical lakes of the Kola Peninsula and others sub-Arctic lakes is low, with average values during the vegetation period generally not exceeding 1 g/m^3, with the number of cells less than 150,000 cells/l (Rodhe 1948; Lund 1962; Sharov 2002). In Lake Imandra, the structure of biomass pre-industrialisation was represented by diatoms and chrysophyceans (Table 2). Average values of chlorophyll '*a*' reflecting production processes during the period of open water, would have been about 0.2–0.3 mg/m^3 by analogy with other lakes of the northern Kola Peninsula. Using Shannon's Index (H') to characterise community species diversity in August (period of vegetation growth) lakes similar to Lake Imandra, but unpolluted, have an H' score of 3.5 (Table 3).

Zooplankton

Early research by Rylov (1916) showed that the zooplankton structure of Lake Imandra included copepods (*Cyclops scutifer*, *Megacyclops viridis*, *Eudiaptomus gracilis*, *Heterocope appendiculata*) and cladocera (*Holopedium gibberum*, *Daphnia longiremis*, *Bosmina obtusirostris*, *Polyphemus pediculus*, *Bythotrephes longimanus*, *Leptodora kindtii*). Krokhin and Semenovich (1940) found 44 zooplankton species (4 protozoa, 19 rotifera, 6 copepoda, 15 cladocera). Dominant organisms were *Kellicottia longispina*, *Polyarthra platyptera*, *Keratella cochlearis*, *Asplanchna priodonta*, *Conochilus unicornis*, *B. obtusirostris*, *C. scutifer* and *Diaptomus gracilis*.

There is no data on the pre-industrial number and biomass of species for the Bol'shaya Imandra basin, although it is possible to estimate background conditions by analogy with non-polluted areas of the lake (south-western parts of Babinskaya Imandra) and other lakes of the Kola Peninsula. Using analogous sites, the number and biomass is likely to have varied between 10,000 and 100,000 specimens/m^3 and 0.2 and 1.0 d.w.w. g/m^3, respectively, Shannon's Index of species diversity is likely to have been between 2.5 and 3.0 (see Table 3).

The mean individual weight of zooplankton (biomass (B)/number (N)) is $0.01–0.02\times10^{-3}$ g, indicating the predominance in the community of the species valuable in terms of feeding: cladocera (*B. obtusirostris*, *Daphnia cristata*, *H. gibberum*) and copepoda (*Mesocyclops leuckarti*, *E. gracilis*) (Vandish 2002).

Macrozoobenthos

The macrobenthos in the profundal zone of the lake consisted of more than 70 species and forms of invertebrates, dominated (by number of species and frequency of occurrence) by midge larvae (*Chironomidae*), bivalves (*Euglesa* spp.) and crustacea (*Monoporeia affinis*, *M. Relicta*), which are distributed in many lakes of Fennoscandia (Gerd 1949; Sarkka et al. 1990). Oligochaeta were represented by the *Lumbriculidae*, *Naididae* and *Tubificidae* families (Table 2). It is rather difficult to reconstruct natural parameters of zoobenthos using analogies with non-polluted areas, because of the influence of the ecosystem character on species content and structure of zoobenthos communities. *M. affinis* predominates in non-polluted areas. For the pre-industrial period, average values of biomass of zoobenthos were not more than 1.1–1.4 g/m^2 (Krogius 1931, Krokhin and Semenovich 1940) corresponding to an α-oligotrophic lake. Shannon's Index of species diversity was 3.5 (Table 3).

Fish

Fish catches showed that the lake was a typical whitefish-loach lake with the presence of trout. Sixteen species of fish were recorded. In catches of 1945–1960 species of freshwater–Arctic complex predominated, their contents in catches was: 20% *Coregonus lavaretus* (L.) and 50% *Coregonus albula* (L.), 7% *Salvelinus alpinus* (L.) and 3% *Salmo trutta trutta* (L.) (Galkin et al. 1966). Estimated fish productivity of the lake was 4 kg/ha.

Lake ecosystem degradation

Anthropogenic pressure on the lake began in the 1940s, reached a peak in the 1980s and caused changes in all structural components of the ecosystem. Data is available for the period 1983–1992, when the effects of pollution were most evident, and processes of degradation reached a maximum. During this period there were detailed investigations of water chemistry, zoobenthos and fish population. Investigations of phytoplankton and zooplankton communities started in 1994, when the pollution level had begun to decrease. The lake was subject to pollution by a number of contaminants including heavy metals, nutrients, sulphates and chlorides. This led to complex changes in the ecosystem with interaction of processes often in different directions, for example toxic pollution and eutrophication. Fig. 2 shows the influx of nickel in the lake and concentration in water giving an indication of toxic pressure.

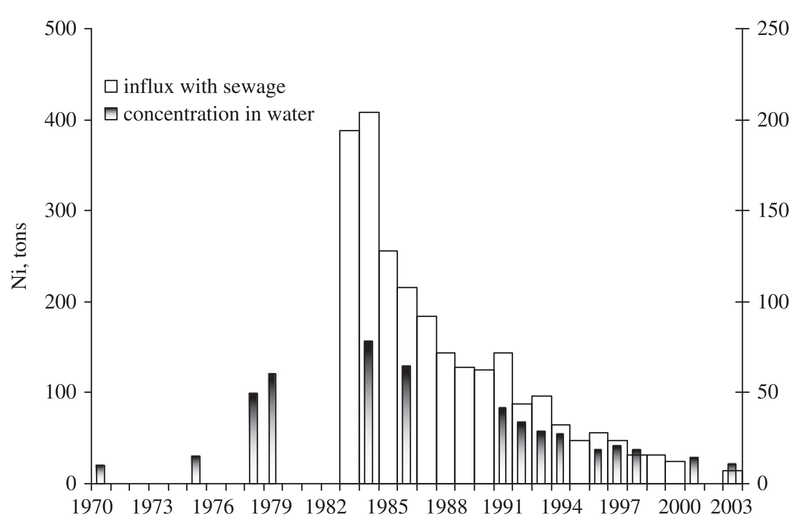

Fig. 2. Dynamics of nickel influx (tons) into Lake Imandra and nickel concentration in water (μg/l) of the Bol'shaya Imandra.

Water chemistry

During the period of intensive anthropogenic loading, water chemistry changed from background conditions, especially at sites of pollution. Table 2 indicates that water transparency decreased, salt concentration increased, and pH became more alkaline. As a result of salt inputs the ionic composition of waters changed. Technogenic sulfates dominated over hydrocarbonates. The influence of pollution from the metallurgical industries resulted in increased concentrations of heavy metals, particularly nickel, copper and zinc in the water column.

Due to the low levels of organic matter, surface waters of sub-Arctic lakes have low complexing abilities and, accordingly, low buffer capacity with respect to pollutant metals. Consequently, most of the technogenically introduced metals migrated in the most toxic ionic form (Moiseenko 1999).

The anthropogenic influx of nitrogen and phosphorus to Lake Imandra, largely from domestic sewage, caused an increase of total contents of these elements and their mineral forms compared with reference conditions. According to the classification of Vollenweider (1979), most of the lake corresponded to mesotrophic status in terms of phosphorus content. Monche and Belaya bays (which receive domestic sewage) and other warmed bays became eutrophic. The accumulation of organic matter near the sediment–water interface contributed to the formation of anoxic conditions, which increased stress for vulnerable lake species. The depletion of oxygen at the bottom of the lake in winter and spring would have been likely to increase metal cycling and the formation of abnormally high concentrations of metals Mn, Fe, Cd, Cu, Hg, Mo, Ni, Pb, Co, and Zn (Moiseenko 1999).

Table 1 shows that water chemistry and living conditions of aquatic species altered from natural background conditions, and water toxicity caused changes in aquatic communities.

Phytoplankton

The transformation of the structure of phytoplankton communities resulted in poorer communities; phytoplankton species contents and community structure changed (Table 2), increasing the range of maximal and minimal values of number of species and biomass of phytoplankton. In the areas of high metal concentrations, the transformation of phytoplankton community structure showed an increase in diversity of green algae, and the number of species in samples reach 17 (compared with a typical background of 5).

There was a mass expansion of green algae (e.g. *Scenedesmus* sp., *Pandorina* sp.) and some diatoms. *Scenedesmus* predominated in terms of number of species. The amount of blue-green algae was significantly higher in Bol'shaya Imandra than in other part of lake and desmids were absent. In terms of diatoms, the role of *Tabellaria* species decreased. Particularly, intensive algal development was noticed where drainage from the metallurgy industries and from household waste entered the lake. Phytoplankton biomass increased across the lake, in some parts it reached more than $20\,g/m^3$ (Sharov 2002). This can be explained by the abundance of nutrients in household waste waters. As toxic pollution increased, smaller algal forms prevailed which is reflected in the mean individual weight of algae (Table 3). The increasing biomass of chlorophyceans and cryptomomads may also be an indicator of lake eutrophication.

Zooplankton

In the period of intensive metal pollution of the lake, the adult species of Cladocera were largely absent (e.g. *L. kindtii*, *P. pediculus*). *Collotheca* sp., *Conochilus* sp. and *H. gibberum* were also absent (Table 2). Widely spread eurybiontic species *A. priodonta*, *K. cochlearis*, *K. quadrata*, *Notholca* sp., *B. obtusirostris* dominated. *L. kindtii*, *P. pediculus*, *D. cristata*, *E. gracilis*, *H. appendiculata* were found in isolated cases. Being most adapted to the influence of industrial sewage, rotifers began to dominate the zooplankton structure. *A. priodonta* accounted for 40–50% of the total number of species (Yakovlev 1998; Vandish 2000). The domination of rotifer individuals in polluted waters was noted by MacIsaac et al. (1987).

Large numbers and biomass of zooplankton were recorded in 1994 (260.62 thousand specimens/m^3, 1.13 d.w.w. g/m^3) and in 1996 (309.6 thousand specimens/m^3, 1.49 d.w.w. g/m^3), respectively. Shannon's Index varied between 1.5 and 2.3 (Table 3) and the mean individual weight of zooplankton (expressed as biomass/number) decreased ($0.006 \pm 0.002 \times 10^{-3}$ g, Table 3). This illustrates the predominance in the community of small-size organisms with small individual mass (mainly rotifers). During this period, in areas of intensive technogenic influence, total number, biomass and production of zooplankton increased compared to control areas and areas of lesser pollution. These community changes were due to high contents of nutrients that caused eutrophication.

Zoobenthos

The most significant degradation of macrozoobenthos communities occurred from the late 1970s to the early 1980s when biodiversity decreased and biomass increased. In areas of copper and nickel pollution, *Chironomus* spp., *Procladius* spp., Dytiscidae and

Table 4. The characteristic of whitefish diseases (% from number of the surveyed individuals) from Lake Imandra in various years of research

The main symptoms of fish diseases	1981 ($n = 788$)	1986 ($n = 721$)	1991 ($n = 453$)	1996 ($n = 462$)	2003 ($n = 235$)
Nephrocalcitosis	52	47	45	14	–
Fibroelastosis	48	53	55	48	39
Lipoid degeneration of a liver and a cirrhosis	100	89	78	48	39
Anomalies of a structure gonads	34	27	8	–	–
Scoliosis and osteoporosis	6	4	2	–	–

Nematoda (biomass up to 20.0 g/m^2) were predominant. In areas of mining industry and intensive euthrophication *Chironomus*, *Tubifex tubifex*, *Limnodrilus hoffmeisteri* and *Procladius* spp. were predominant (Table 2). Yakovlev (1998) recorded an abnormally high zoobenthos biomass in 1978–1979: up to 58 g/m^2 near the copper–nickel industry and up to 300 g/m^2 in areas of high eutrophication. In general, up to 1980, biomass values in the polluted part of lake Imandra increased in 20 times, illustrating the intensive processes of eutrophication alongside toxic pollution.

The stability of midges (mainly of genera *Chironomus* and *Procladius*) and Nematoda to the impact of heavy metals has been noticed by some authors (Nalepa 1987; Yakovlev 1998; Iliyashuk 2002b). These organisms prosper when there is an increase in sediment organic matter accompanied by a decrease in competitors, whose populations decline due to toxic pollution. Some oligochaetes develop actively in conditions of accumulation of sediment organic matter (Milbrink 1983), but with low metal concentration in water and sediments.

Biodiversity, as measured by the Shannon index, was low in areas of high pollution (0.5–1.0H'). Numbers of some Chiromonidae and mollusc species more sensitive to pollution decreased more than two times in relation to the total number of benthic invertebrate species. The presence of two glacial relict species, *Monoporea relicta* and *M. affinis*, had been previously recorded in Lake Imandra, but during the period of degredation, only *M. affinis* was found, the more vulnerable *M. relicta* being absent.

Fish

During the period of maximum water pollution, the number of typical Arctic fish, salmon trout and Arctic char, decreased sharply; stenobiontic, cryophilic and oxiphilic fish such as these are vulnerable to toxic conditions (Heath 1995). Fish productivity decreased to less than 1 kg/ha despite the fact that there was no fishery in operation during this period. Whitefish and perch dominated fish catches. There was a large increase in minnow numbers, and the number of smelt also increased.

Incidences of mass whitefish disease were recorded in the 1970s. These included change of the integument colour (de-pigmentation), anal inflammation, tousling of scales, oedema gills and appearance of anaemia rim, destructive changes of liver (increase of size, change colour and friability) and kidneys (colour, granulation, thickening of renal and presence of nephritic calculi), and anomalies in gonad texture. The main types of fish diseases are given in Table 4.

Rarer whitefish diseases were also recorded, such as nephrocalsitosis (kidney stones). The frequency of case rate (percent of those surveyed) was closely related to nickel concentration in water and its accumulation in the kidney (Moiseenko and Kudrjavzeva 2001). The salt content of the lake water altered during this period, with total content increased three to five times, and this may have stimulated the emergence of this endemic disease. Productive areas of benthic communities developed which attracted whitefish due to the high biomass of zoobenthos. By migrating to these food-rich areas, fish were exposed to the effects of heavy metals. Symptoms of disease and impairment of functioning were apparent in fish caught in polluted areas of the lake (Moiseenko and Yakovlev 1990). Criteria to determine fish condition (by physiological indicators of toxification) are important to estimate toxic effects and are used often in assessment of ecosystem health as integral parameter (Adam and Ryon 1994). The fish diseases of Imandra lakes indicated the critical condition of ecosystem health in Lake Imandra during this period.

Main recover tendencies

Since the mid-1980s, toxic metal pollution has decreased significantly (Fig. 2). Evidence of water quality improvement and community revitalisation can be seen in Tables 1–3.

Water chemistry

The decrease of heavy metal inputs to the lake have been reflected in decreased water concentrations of nickel, copper and zinc (Table 1), and consequent

Table 5. Frequency of occurrence (F_{am}, %) in samples, the maximal values of amount (max N_{am}) and biomass (max B_{am}) of *Monoporea affinis* in Imandra lake; n, total of sample, n_{am}, amount of sample in which it is found *M. affinis* (n_{am})

Years	Depth, *m*, mean (min–max)	F_{am} (%)	max N_{am} (ind m^{-2})	max B_{am} (g/m^2)	*n*	n_{am}
1930	–	62.0	–	–	–	–
1968	–(3.5–49.0)	35.9	1087	1.21	46	–
1978–1986	13.4 (6.2–24.0)	26.3	1125	1.17	19	5
1987–1996	12.2 (6.2–22.0)	50.0	2961	10.28	12	6
1998	14.8 (6.5–30.0)	45.2	22,350	27.20	42	19

Crossed out, the data are absent. Citing Iliyashuk (2002a).

decrease in the toxic properties of the lake water. In the area of high pollution, the nickel content in 1983–1992 was about 150 μg/l, and in 2003 it was no more than 27 μg/l. The concentration of total phosphorus has not reduced, although the concentration of bioavailable phosphates has decreased, suggesting more active utilisation in the foodchain. The lake catchment is exposed to airborne pollution, which causes sulphate influx into the lake. During the period of intensive pollution, the silicon content was higher than the natural background, and in the recovery period, concentrations are decreasing due to eutrophication and diatom growth.

Phytoplankton

Results of an investigation in August 2003 showed an insignificant decrease in phytoplankton abundance in comparison with similar periods in 1994–1996 (Table 3). Values of biomass fluctuated from 2.3 to 7.9 g/m^3. A high algal biomass was accompanied by chlorophyll '*a*' concentrations that were rather high for northern regions (from 2.8 to 4.5 mg/m^3). At the same time, there was a more uniform distribution of algae by lake area in comparison with period of maximum pollution. There is still an abundance of species of *Cryptomonas, Stephanodiscus* and of *Aulacoseira islandica* (Table 2) and the diatom *T. fenestrata* was present in plankton samples of Lake Imandra during the recovery period.

Zooplankton

Between 1998 and 2003, the number of species in the zooplankton community decreased (up to 70,000 specimens/m^3), although biomass did not show such a marked decrease (to 0.74 and 0.87 d.w.w. g/m^3, respectively). Analysis of a recent zooplankton species list showed that the proportion of rotifers (typical pollution indicators) decreased and that of Cladocera (*B. obtusirostris*) and Copepods (*Cyclops* sp., *Mesocyclops leucrarti*) increased. Being important in respect of feeding, Cladocera (e.g. *H. gibberum*, *Daphnia* sp., *L. kindtii*) were recolonising, having been absent during the period of most intensive pollution. The crustaceans *L. kindtii*, *P. pediculus*, *E. gracilis* and *H. appendiculata*, sensitive to pollution, were present in small numbers. The rotifers *Bipalpus hudsoni*, *K. longispina*, *Notholca* sp. predominated (Table 2).

In mid-August 2003, the total number and biomass of zooplankton was lower than previous years (35.8 thousand specimens/m^3 and 0.63 d.w.w. g/m^3, respectively). The mean individual weight of zooplankton had increased reflecting the increasing number of zooplankton organisms valuable with respect to feeding. Shannon's Index also increased in the recovery period (Table 3).

Zoobenthos

Data gathered in 1998 showed that, in the polluted areas of Bol'shaya Imandra, condition for benthic organisms were still extreme (Iliyashuk 2002a). Lake sediments accumulated large amounts of metals and organic matter during the pollution period, which is why benthic invertebrate communities are slower to recover. Benthic communities are still characterised by low values of biodiversity (Shannon's Index is 0.95–1.05). Here still quantitative development is determined by the oligochaete species *T. tubifex* and *L. hoffmeisteri* (Iliyashuk 2002a).

During recent years maximum values in the abundance of the amphipod *M. affinis* in the profundal zone has increased sharply (Table 5). Table 5 shows that in 1998, both the maximum values for the amount (individuals/m^2) and biomass (g/m^2) have increased markedly over previous years. The increasing impact of *M. affinis* on the Chiromonidae family fauna is evidence of the euthrophication process. These results correspond to data from Marzolf (1965). However, the decreasing anthropogenic pressure on Lake Imandra led to estimates of decreasing *M. affinis* abundance in the current decade (Iliyashuk 2002b).

Fish

Numbers of the most vulnerable species (Arctic char and trout) are still low in the lake. There are high numbers of minnows in the shallows. During the period of economic crisis in Russia, fish were poached in increasing numbers, which together with eutrophication may have affected the structure of fish communities, in particular, on white fish population as their forage base increased. It is difficult to determine the relative importance of different factors affecting the fish population and structure because it is not possible to quantify the number of fish taken from the lake.

Cases of fish disease have become rarer as a result of decreasing toxicity of the lake water (Table 4). Physiological conditions of the fish have improved and ecosystem health has improved. Of the fish caught during the recovery period, none exhibited the diseases nephrocalcitoses and scoliosis. Cases of other whitefish diseases were found which probably reflect the prolonged influence of low doses of contaminants.

General discussion and conclusion

Anthropogenic modifications of an Arctic water ecosystem

The ecological conditions in Lake Imandra before, during and after receiving large inputs of anthropogenic pollutants have been well documented. By reviewing and collating data from past studies, a unique dataset has been compiled documenting the anthropogenic modifications of an Arctic water ecosystem.

Prior to the 1930s, the lake was an oligotrophic ultrafresh reservoir with low concentrations of nutrients, suspended matter and microelements. The aquatic community consisted of cryophilic stenobiontic species, which are characteristic of the Arctic. For more than 60 years, the lake was polluted with toxic metals from industrial sources and by the influx of nutrients into the ecosystems from domestic sewage. During this period of anthropogenic pollution, the hydrochemical regime was transformed; concentrations of sulphates and suspended matter increased, heavy metals and organic matter from the water column were deposited into the lake sediment where they accumulated. The combined impact of domestic sewage and industrial waste meant that the lake was exposed to processes of euthrophication and toxification.

During the period of degradation, the lake ecosystem was transformed from stable reference conditions into a new developing phase. The number of typical Arctic species, many of which were vulnerable to contamination, decreased and number of eurybiontic species increased rapidly due to the high concentrations of nutrients and absence of competition with typical Arctic species. The disappearance of a number of species led to a simplification of the community structure and of the trophic foodchain. In the zoobenthos community, filter feeders and scrapers were replaced by detritophages, pulverisers, pelophages and predators. The zooplankton structure saw a replacement of large Cladocera and Calanoida with smaller fine-particle filter feeders (Bosmina), the former being unable to filter large food elements under conditions of increased water turbidity. The role of predatory pelagic brown trout and Arctic char in the trophic structure of the ecosystem was reduced (Moiseenko 1999). The mean individual weight of plankton organism decreased during the period of high pollution, indicating the predominance of small forms (r-strategists), which provided more rapid cycling of biomass and utilisation of additional energy.

In benthic communities under conditions of toxic pollution there were rising numbers of metal pollution-resistant midges (Chironomidae family). In some areas, where euthrophication was predominant, the mean individual weight of benthic organism decreased due to the mass growth of Tubificidae.

This is characteristic of an unstable ecosystem under stress (Odum 1985). Since the 1990s Lake Imandra has shown signs of recovery due to a decrease in toxic pollution. The lake has seen recolonisation by inhabitants of Arctic waters along with changing of dominant complexes, increasing mean individual weight of organisms in communities, and increasing Shannon's Index of species diversity.

Accumulated biogenic elements are involved in biological cycling in the ecosystem, shown by: levels of total phosphorus exceeding mineral forms (P_{tot}/PO_4^{2-} was 2.6 in 1983 and 8.7 in 2003); increasing of number of predators in the structure of zooplankton and benthos; and increasing numbers of fish. Decreasing concentrations of bioavailable forms of nutrients and silicon content suggest that they are being utilised by diatom algae, which in the period of recovery are predominant, and present in greater numbers than previously. Maximum and mean values of biomasses and chlorophyll '*a*' content during the recovery period have not changed.

The above changes are consistent with the character of recovery in the American Great Lakes. For example, in Lake Ontario the phosphorus concentration decreased steadily from 1968 to 1985. However, production of phytoplankton and chlorophyll '*a*' remained unchanged until the beginning of the 1980. Since then indications of oligotrophication of the lake have become apparent. Further monitoring data show that by the year 2000 the phosphorus concentration had decreased to 6 μg/l. The lag in response of phytoplankton numbers is explained by a change in the structure of phytoplankton, and by an increase of cryptomonads supporting its biomass (Greate Lakes Ecosystem Report 2000, 2001).

Benthic communities are less responsive to ecosystem recovery and their biodiversity is low. Such inhabitants as the amphipod *M. affinis* have recovered well in conditions of decreasing toxic pressure and favourable food conditions. The chemical and biological indicators of ecosystem health suggest that the ecosystem condition is improving.

Unfortunately, published data on fish production in the recovery period are not available. However, anecdotal evidence suggests that the unauthorised catching of fish has increased. At the same time the biomass of zooplankton is decreasing, which can be explained by two factors: (i) an increase in the predominance of predatory forms in these zooplankton communities; and (ii) the increase in number of fish due to reduced pressure on the population.

Attributes of ecosystem revitalisation shown in this work conform to the stages of ecosystem succession as described by Odum (1985): from developing stage to more stable (climax) modification, but distinguishable from its natural structure.

A return to reference conditions?

This paper has used available data to determine a set of reference conditions for an Arctic lake (Tables 1–3). It has also shown that Lake Imandra has passed through a degradation phase and entered into a recovery period. This poses a number of questions as to how the recovery phase might continue. Firstly, is it possible to return to reference conditions with a further decrease, or cessation, in anthropogenic loads? Secondly, if it is possible to return to reference conditions, what is the timescale for such a recovery? Finally, how might climate change and global warming affect the recovery process?

Taking the first two questions, the residence time of water in Lake Imandra is 2 years. If toxic pollution of the water ceased altogether, it is estimated that the water column would be free from anthropogenic industrial sources of metals over a 5- to 6-year period. A movement of the lake towards an oligotrophic state would progress slowly.

Analysis of literature data on ecosystem recovery focusing on re-oligotrophication shows that, after the input of nutrients has ceased, the recovery does not depend only on the velocity of phosphorus removal from the ecosystem. For example, Lake Erie in North America has a similar water exchange period to that Lake Imandra (2.6 years). Studies show that Lake Erie returned to a mesotrophic–oligotrophic state 16 years after the phosphorus loading ceased (Greate Lakes Ecosystem Report 2000, 2001), i.e. much slower than two to three complete water exchanges.

Our investigations show that biogenic elements involved in biological cycling are removed from the ecosystem slowly and support bioproductivity formed in the period of pollution. This agrees with literature data (Jeppesen et al. 2005; Falk et al. 2006; Palmer et al. 1997). The species structure of communities differed from pre-pollution 'natural' ones, exhibiting the following features: (i) a number of species typical of the natural state are not recovered or are represented by a single specimen; and (ii) species dominance is changing in communities, for example, species that are present in low numbers in natural conditions are multiplying in high quantities.

A return to reference conditions for Arctic ecosystems after a period of heavy anthropogenic stress (toxins and nutrients) is likely to be a lengthy process and possibly unattainable, because an aquatic ecosystem with new parameters attains stability. The indicators of ecosystem modifications described during degradation and in the current period of recovery characterise decreasing pollution and agree with the regularities stages of ecosystem successions (after Odum): from a natural background via a developing phase to a more stable, mature (climax) modification. It has the following basic features: a higher role of upper blocks of the ecosystem's trophic structure; successful utilisation of mineral forms of biogenic elements; a growth in the proportion of K-strategists and in predatory forms. Therefore, the term '*recovery of ecosystems*' cannot be used in this case pertaining to a return to the natural state.

The impact of climate change on ecosystem recovery

To predict the impact of climate warming on ecosystem recovery is rather theoretical. It is probable that increasing water temperature as a result of global warming will make a return to reference condition unattainable (Harris et al. 2006). Temperature influences the following ecosystem functions: (1) rate of carbon fixation, (2) rate of nutrient increase/decrease, (3) rate of detritus processing and storage, (4) rate of suspended solid trapping, and (5) nutrient trapping and storage (Cairns 2005). Accumulated nutrients will be more actively utilised in trophic nets, as in warmer conditions communities will move towards the predominant development of eurybiontic species. The influx of biogenic elements from the catchment is likely to increase with rising temperature and it will provide increasing productivity for pollution-resistant species. In warm, more eutrophic conditions, there will be increasing bioaccumulation of metals from the water column by plankton communities, and active release from sediments in anoxic conditions, which will cause oppression of stenobiontic species, more vulnerable to toxic pollution and typical for Arctic regions.

At the present time, macrophytes communities are very poor in the lake, but with increasing temperatures their development will create more favourable conditions for perch and pike. Rising temperatures are likely to increase uptake of metals into fish, increasing the toxic effect. Climate warming is unlikely to favour fish species such as Arctic char and trout, although other species such as whitefish, perch, minnow and smelt may benefit from advantageous ecosystem changes. For example, higher bioproductivity of amphipods in warmer temperatures will create more favourable conditions for feeding and growth of whitefish and lead to an increase in numbers. The lake is likely to fluctuate from mesotrophic to eutrophic in some areas, which is observed in present conditions of biogenic pollution.

Studies reviewed in this paper show the changing effects of man's impact on an Arctic water ecosystem under varying conditions of anthropogenic pollution. In the future, it is likely that the human population, and so the anthropogenic pressure, in Arctic regions will increase.

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