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and fish from lakes in Northern and Arctic regions of
Norway.**



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Sammendrag / Summary

This report presents the levels and distribution of contaminants in lake sediments and fish in Northern and the Arctic Norwegian islands, Spitsbergen and Bear Island. Samples were analysed for the following groups of contaminants: heavy metals, persistent organic pollutants (POPs: PCBs and organochlorine pesticides) and PAHs. The study serves as a baseline investigation of the present situation, as well as providing reference values for future studies. The chosen contaminants analysed, as well as the methods used for field sampling and analysis, were in accordance with AMAP recommendations. The results are presented graphically and discussed.

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Sammendrag og Konklusjon

Arctic Monitoring and Assessment Programme (AMAP) ble etablert i 1991 på den første Arctic Ministerial Conference i Rovaniemi, Finland, som en del av Arctic Environmental Protection Strategy (AEPS). Hovedformålet med AMAP er kontinuerlig overvåkning av nivåene av antropogen forurensing i alle miljøkomponenter, inkludert mennesker, i Arktiske områder og vurdere effektene av forurensingen på økosystemet.

Programmet er delt inn i fem underprogram: atmosfære, terrestrisk, ferskvann, marin og human helse. I AMAP's underprogram for ferskvann, er fisk og sediment i innsjøer valgt som undersøkelsesparametre. Programmet inkluderer også en liste over tungmetaller og persistente organiske forurensninger (POPs) som skal prioriteres i målingene.

Denne undersøkelsen er utført i henhold til AMAP's anbefalinger for metoder på feltinnsamling, analyser og parametre. Innsamlingen ble utført i 1993, 1994 og 1995. En oversikt over innsjøer og stasjonsnummer, deres lokalisering og morfologiske karakteristikk, samt hvilke prøver som ble samlet inn og i hvilket år, er gitt i Appendix 1. Tungmetaller ble analysert i sedimenter fra 60 innsjøer. I tillegg er datamateriale for tungmetaller i innsjøsedimenter fra undersøkelser i 1992 (Rognerud *et al.* 1993) inkludert, dette gir totalt 92 undersøkte innsjøer. Konsentrasjonene av POPs ble analysert i sedimenter fra 22 innsjøer.

Rapporten viser nivåer og fordeling av miljøgifter i innsjøsedimenter og fisk på fastlandet i Nord-Norge, og på de arktiske norske øyene, Spitsbergen og Bjørnøya. Resultatene er presentert grafisk og diskutert. Målet med undersøkelsen er å kartlegge den nåværende situasjonen med hensyn til kontaminering, samt fremskaffe oversikt over referanseverdier for framtidige studier.

De originale dataene fra miljøgiftanalysene er gitt i to separate appendix rapporter. Appendix A presenterer dataene fra sedimentanalysene og Appendix B analysene av fisken.

Feltarbeid, data-analyser og rapportering er finansiert av Statens Forurensingstilsyn (SFT), Fylkeskommunen i Finnmark, Landsdelsutvalget for Namdalen og Nord-Norge samt Akvaplan-niva's instituttprogram. Denne rapporten er en del av Norges bidrag til AMAP.

Tungmetaller i sedimenter

Konsentrasjoner

Nivåer av de fleste tungmetallene i de øvre sedimentsjiktene i innsjøer i Nord-Norge er generelt lavere enn de som er funnet i sørlige deler av landet (unntatt for Ni konsentrasjonene). Høyeste nivå av Cd i innsjøsedimentene ble funnet i Holmevatn i Nordland (1.46 µg/g). Høyeste konsentrasjonen av Hg ble funnet i Vikvatnet (0.43 µg/g), og høyeste konsentrasjonen av Pb og Ni ble funnet i Norlivatn, Nordland (131 µg/g) og Bjørnevatn i Finnmark (372 µg/g). Konsentrasjonen av Ni i sedimentet øker med økende breddegrad. Rognerud & Fjeld (1990) viste at gjennomsnittskonsentrasjonen i den sørøstlige delen av Norge er 15-20 mg/kg tørrvekt. I denne studien var median-konsentrasjonene av Ni 25-58 mg/kg tørrvekt i overflatesjiktene. For elementene Al, Cu, Fe, Zn, Ti og Se er det ikke utført analyser i nasjonale undersøkelser.

På de arktiske øyene, er gjennomsnittskonsentrasjonene av Hg, Pb og Cd i overflatesedimenter generelt lavere enn nivåene funnet på fastlandet i Nord-Norge.

Anrikningsfaktorer (K_f)

Overflatesedimentet i innsjøer på fastlandet i Nord-Norge er anrikt med tungmetallene Hg, Pb, Sb og Cd. De sentrale delene av det undersøkte området er mindre anrikt sammenlignet med de sørlige og nordlige delene. Slike konklusjoner kan man ikke trekke for de andre elementene. Anrikningsfaktoren, K_f, for henholdsvis Hg og Pb viser en 1.5-10 og 1.5- >10 ganger økning fra 2-3 cm sjiktet til 0-1 cm sjiktet, noe som indikerer en høy variasjon mellom innsjøer når det gjelder deponering og sedimentering i området gjennom de siste 10 årene. En slik trend vises ikke for de andre metallene i denne undersøkelsen.

Medianverdier for anrikningsfaktorer for metaller, basert på verdier fra alle prøvetatte innsjøer på Spitsbergen og Bjørnøya er beregnet. Dataene viser at de øverste sedimentsjiktene i innsjøer på Spitsbergen er anrikt med tungmetallene Hg, Pb, Se, Ti og Zn. På Bjørnøya er antimon (Sb) det eneste elementet som viser anrikning i overflatesedimentet.

Den geografiske fordelingen av de fleste metallene i innsjøene på fastlandet indikerer spredningstrender. For elementene Al, Cd, Hg, Pb, Sb og Zn er konsentrasjonene høyest langs kysten av Nordland og Troms. Steinnes *et al.* (1994) fant indikasjoner på lignende geografiske mønster på nivåene av Pb, Sb og Se i mose. Det er blitt vist at den signifikante tilførselen av Se i Norge kommer fra det marine miljøet (Steinnes *et al.* 1994). Nivåene av As, Cd, Hg, Pb, Sb og Zn i moser er relatert til atmosfærisk langtransport (Berg *et al.* 1995).

Forhøyede konsentrasjoner av Hg, Ni og Pb ble målt i Sør-Varanger, og området ble klassifisert til å være moderat til sterkt påvirket. Forurensingen er relatert til lokale utslippskilder på Kola halvøya, hovedsakelig nikkellindustrien (Steinnes *et al.* 1994; Rognerud *et al.* 1993).

Den geografiske spredningen av Hg i sediment virker ikke å være relatert til spredningsmønsteret i moser. Steinnes & Andersson (1991) og Steinnes (1995) fant en fordobling i konsentrasjonen av Hg i moser fra innlandet i Troms og Finnmark mot den nordlige kysten av Finnmark. En mulig forklaring på dette var den økende bidrag av tørr deponering av Hg⁰, under forhold hvor forflytning til atmosfæren ble redusert. I denne undersøkelsen ser man ingen signifikante mønster fra innlandet mot kysten. De høyeste konsentrasjonene i overflatesediment ble funnet langs kysten av Nordland og Troms og i Sør-Varanger (0.1-0.2 µg/g, sammenlignet med konsentrasjoner hovedsakelig under 0.1 µg/g i Finnmark). Noen innsjøer i disse områdene hadde også høye anrikningsfaktorer for Hg (fra markert til sterkt påvirket). De høyere nivåene i innsjøsedimentene langs kysten ser ut til å henge sammen med større nedbørsmengder.

Anrikningsfaktorene for Pb var høy i mange innsjøer i Lofoten-Vesterålen og Senja, så vel som i kommunene Kautokeino og Sør-Varanger. To innsjøer hadde høye anrikningsfaktorer for flere metaller, nemlig Ø. Kaperdalvatn (68) på Senja og Ryggdalsvatn (75) i Vesterålen. Disse innsjøene var moderat til meget sterkt påvirket av Al, Cd, Cu, Fe, Hg, Pb, Sb, Se og Zn. Metallkonsentrasjonene i overflatesedimentet var bare litt høyere enn i prøver fra nærliggende innsjøer, men lave konsentrasjoner av de fleste metallene i referansesedimentet førte til høyere anrikningsfaktorer (beregnet fra forholdet mellom konsentrasjon i toppsjiktet og konsentrasjonen i referansesjiktet).

Storvatnet nært Alta, hadde også forhøyede konsentrasjoner av Ni, Cu og Pb. Årsaken til dette er trolig tidligere gruvedrift i nedbørsfeltet til dette vannet.

Kvikksølv i fisk

Hg ble analysert i fisk fra Ellasjøen på Bjørnøya, 8 innsjøer og Kautokeinoelva i Finnmark og Finnsnesvatnet i Troms.

Konsentrasjonsnivåene

De høyeste konsentrasjonene av Hg i denne undersøkelsen ble funnet i gjedde fra Kautokeinoelva, i abbor fra Gavdujavri, i sik fra Ravdujavri, og i røye fra Ellasjøen.

Predatorfisk, som gjedde og stor abbor, hadde de høyeste Hg konsentrasjonene. Konsentrasjonen i gjedde økte med kroppsstørrelsen for fisk som var lengre enn 50 cm. Det samme gjaldt for abbor over 20-25 cm. For gjedde og abbor var det en signifikant korrelasjon ($p < 0.05$) mellom lengden på fisken og konsentrasjonen av Hg i muskelvevet. Når det gjaldt røye i denne studien, var ikke Hg konsentrasjonen signifikant korrelert med lengden på fisken. Konsentrasjonen av Hg i sik var lavere enn i de andre artene og var omtrent lik i alle innsjøene det ble tatt prøver fra.

En av grunnene til forskjellen i Hg nivået mellom fiskearter er at abbor og gjedde generelt har andre næringsvalg enn stor røye og sik. Gjedde og abbor, spesielt de største individene, er predatorer, som vanligvis spiser fisk som byttedyr. Mindre størrelser av røye og sik spiser på et lavere trofisk nivå, med zooplankton eller bunndyr som hovedføde.

For sik var det ingen signifikant korrelasjon mellom kroppslengde og kvikksølvnivå i fiske-muskel. Men det viste seg at Hg konsentrasjonen i gytende sik fra S. Galdinjavri var signifikant forskjellig fra ikke-gytende individer ($p = 0.0001$). Det var også en signifikant lengdeforskjell mellom gytende og ikke-gytende fisk. Disse resultatene indikerer at i denne innsjøen er Hg konsentrasjonen i sik primært relatert til lengde, og ikke til graden av modenhet hos fiskene. Ved sammenligning av sik fra alle de undersøkte innsjøene, viser statistiske analyser at det ikke var signifikant forskjell i Hg konsentrasjonen mellom hunn- og hannfisk. Det var heller ingen signifikant lengdeforskjell mellom kjønnene.

Kvikksølv i sediment og kvikksølv i fisk

Man fant indikasjoner på en positiv relasjon mellom Hg konsentrasjon i abbor og i sediment. På grunn av et begrenset datamateriale må denne observasjonen imidlertid behandles med forsiktighet.

Undersøkelsen indikerer at det ikke finnes en positiv korrelasjon mellom konsentrasjonen av Hg i sediment og konsentrasjonen av Hg i sik. Det finnes to typer sik i Finnmark, en som hovedsakelig utnytter bentiske organismer som føde, mens den andre ernærer seg på plankton i den pelagiske sonen. Bunndyretende sik blir større enn pelagisk sik. En slik forskjell mellom sikpopulasjonene kan forklare at fisk fra Ravdujavri hadde høyere verdier av Hg konsentrasjoner. Det er kjent fra litteraturen at forurensningsnivået i fisk øker med byttets trofiske nivå (Kidd *et al.* 1995).

Kvikksølv og selen

Faktumet at den laveste Hg konsentrasjonen ble funnet i fisk fra innsjøen med det høyeste nivået av Hg i sedimentet, kan delvis forklares av Se konsentrasjonen i sedimentet. En høy konsentrasjon av Se ser ut til å redusere opptaket av Hg i biota (Shindler *et al.* 1995; Rognerud & Fjeld 1990). Se kan fungere som en antagonist og oppta det samme bindingspunkt som kvikksølv. Rognerud & Fjeld (1990) fant også en negativ korrelasjon mellom Se konsentrasjon i sediment og i fisk fra samme innsjøer. I denne undersøkelsen ble Se analysert i sedi-

ment fra to av innsjøene hvor det også ble foretatt analyser av sik (Tabell 10). Selv om datagrunnlaget er lite, kan man se indikasjoner på en lignende trend. Innsjøen som inneholder fisk med den høyeste konsentrasjonev av Hg har den laveste konsentrasjon av Se i sedimentet, mens sedimentet i innsjøen med den laveste Hg konsentrasjonen i sik har de høyeste nivåene av Se.

I henhold til de siste retningslinjene fra EU, har Norge etablert nye retningslinjer for Hg nivå i fiskeprodukter (Sosial og Helsedepartementet 1995). I følge disse skal ikke konsentrasjonen av Hg i fiskeprodukter som benyttes til konsum overstige 0.5 mg/kg våtvekt. For gjedde og fettrike arter har man definert en høyere max grense på 1.0 mg/kg våtvekt.

All røye og sik, samt mindre abbor og gjedde i denne undersøkelsen inneholder Hg konsentrasjoner under 0.3 mg/kg våtvekt. Den største gjedda (i Kautokeinoelva) og den største abborren (i Gavdujavri) hadde konsentrasjoner på henholdsvis 0.52 og 0.47 mg/kg.

POPs i sediment

PCB og HCBz

Resultatene fra en tidligere undersøkelse viser at PCB konsentrasjonene i overflatesediment fra innsjøer på fastlandet i Nord-Norge er generelt lave. Den høyeste konsentrasjonen av Σ_7 PCB i overflatesediment ble målt i Ellasjøen på Bjørnøya. Innsjøsedimentet i Ellasjøen hadde en Σ_7 PCB konsentrasjon på 32.7 ng/g tørrvekt. Konsentrasjonene av Σ_7 PCB i sedimenter fra de andre undersøkte innsjøene, var alle under 15 ng/g tørrvekt. De høyeste Σ_7 PCB nivåene i Nordland ble funnet i Storvatnet (14 ng/g tørrvekt) og Holmevatn (11 ng/g tørrvekt). I Troms ble den høyeste konsentrasjonen funnet i Skøvatnet (10 ng/g tørrvekt), og i Finnmark i Andrevann (11 ng/g tørrvekt).

HCBz konsentrasjonene i overflatesediment fra innsjøer både i Nordland, Troms og de arktiske øyene var relativt lave. Sammenlignet med konsentrasjonene i andre områder, er nivået i innsjøer i innlandet i Finnmark signifikant høyere.

Pesticider

Nivåene av pesticider i overflatesediment var generelt lav.

De høyeste registrerte konsentrasjonene av γ -HCH (lindan), fra 0.56 til 0.7 ng/g tørrvekt, ble funnet i Haukesjøen, Rabbvatnet og Ravdujavri i Finnmark.

Forhøyede konsentrasjoner av Σ DDT og p,p'-DDE i overflatesediment ble funnet i Rabbvatnet og Ropelvatn i Sør-Varanger, Finnmark (3 - 3.4 ng/g tørrvekt) og i Ellasjøen på Bjørnøya.

POPs i fisk

PCBs.

Ekstremt høye PCB konsentrasjoner, hundre ganger høyere enn nest høyeste i denne undersøkelsen (på våtvektsbasis), ble funnet i muskelvev fra røye på Bjørnøya. Konsentrasjonen av Σ_7 PCB ble målt til 1292 ng/g våtvekt, og 50 000 ng/g ekstraherbart lipid. På grunn av de ekstreme verdiene ble prøven reanalysert flere ganger for kvalitetskontroll, resultatet var de samme ekstremt høye verdiene. Nivået er flere ganger høyere enn det verste tilfellet som er rapportert i arktiske strøk i Nord-Amerika. Selv om denne konsentrasjonen er basert på et be-

grenset materiale (en fisk), og trengs verifisering, er nivået så høyt at det må beregnes som alarmerende.

I prøver fra røye på Spitsbergen, varierte Σ_7 PCB fra 2.4 til 34.4 ng/g våtvekt og 64 til 5182 ng/g ekstraherbart lipid. Konsentrasjoner over 1000 ng/g ekstraherbart lipid ble kun målt i røye fra Linnèvann og Kongressvatn.

Det ble målt lave konsentrasjoner i sik, alle under 1 ng/g våtvekt og mellom 45 og 200 ng/g ekstraherbart lipid. Konsentrasjonene av Σ_7 PCB var høyere i abbor enn i sik. Høyeste registrerte verdi var 6 ng/g våtvekt og gjennomsnittsverdiene for abbor fra to innsjøer var hhv. 2 og 3 ng/g våtvekt og 25 til 700 ng/g lipid.

Røye fra Spitsbergen

Røye fra innsjøer på Spitsbergen viser noen karakteristiske vekstmønstre. Spesielt stasjonære populasjoner har sen vekst og lang levetid. En av effektene av fiskens vekst på forurensningsnivået er vekstfortynning (Hammar *et al.* 1993). Store, langsomt-voksende fisk akkumulerer generelt høyere konsentrasjoner av organoklorider enn mindre, hurtigvoksende fisk (Schindler *et al.* 1995). Den eldste fisken i vårt materiale, 22 år gammel på den tiden den ble fanget, inneholdt en svært høy Σ_7 PCB konsentrasjon, 34 ng/g våtvekt.

I Linnèvann og Kongressvann ble svært små fisk samlet inn, 9 år gammel og kun 15 g i vekt. Sammenslåtte prøver av disse "dvergformene" inneholdt svært lite lipider (henholdsvis 0.45 og 0.98 %), sammenlignet med større individer fra de samme innsjøene (mellom 3.5 og 4.5 % ekstraherbart lipid). Konsentrasjonen av Σ_7 PCB i dvergfiskene var ca. 20 ng/g våtvekt og mellom 1500 og 5000 ng/g lipid. Statistiske analyser viste at det var en signifikant negativ korrelasjon mellom PCB konsentrasjonen i røye fra Spitsbergen og deres lipidinnhold, vekt og vekst ($p < 0.0001$; $R = \text{ca } -0.8$).

HCBz i fisk

HCBz konsentrasjonene i fisk fra innsjøer på fastlandet i Nord-Norge var lave. Konsentrasjonene i fisk fra innsjøer på de arktiske øyene var høyere enn i andre områder. Den høyeste konsentrasjonen av HCBz ble funnet i røye fra Rikardvannet på Spitsbergen (6.8 ng/g våtvekt, 322 ng/g ekstraherbart lipid).

HCH i fisk

Konsentrasjonene av γ -HCH på våtvektsbasis var relativt lave i de analyserte artene. Abbor, røye og sik hadde verdier fra 0.05 til 0.18 ng/g våtvekt og mellom 4 og 22 ng/g ekstraherbart lipid. Fordi γ -HCH er mer polar enn for eksempel de mer hydrofobe PCBer, har γ -HCH en mindre tendens til bioakkumulering i fisk (Lockhart *et al.* 1992). γ -HCH konsentrasjonene i biota fra fjerne områder er typisk lavere enn konsentrasjonene av PCB eller DDT.

Nivået av α -HCH var ikke målbar i de fleste prøvene, fordi konsentrasjonene var under 0.06 ng/g våtvekt. Konsentrasjonene av α -HCH som var målbare i prøver av sik og abbor, var mindre enn 0.10 ng/g våtvekt. I røye fra Bjørnøya, var α -HCH konsentrasjonen 0.40 ng/g våtvekt.

DDT i fisk

DDT er et insekticid som er svært giftig, akkumuleres i biota, og dermed anrikes oppover i næringskjeden. Bruken av DDT er sterkt redusert i vestlige land siden 1970-tallet.

Konsentrasjonene av Σ DDT i abbor og sik fra Finnmark var under 1 ng/g våtvekt, med gjennomsnittlige konsentrasjoner mellom 0.17 og 0.6 ng/g våtvekt, og mindre enn 90 ng/g lipid. *p,p'*-DDE utgjorde den største delen av DDT komponentene og deres metabolitter. Konsentrasjonen av *p,p'*-DDE representerte 80 % av Σ DDT konsentrasjonen i abbor og sik. Konsentrasjonen av Σ DDT i røye fra Bjørnøya var alarmerende høy (76.4 ng/g våtvekt, 2 634 ng/g lipid), flere ganger høyere enn i andre fiskeprøver. Denne ekstremt høye konsentrasjonen ble bekreftet av reanalyser. Her utgjorde *p,p'*-DDE omkring 97 % av Σ DDT konsentrasjonen.

Klordaner i fisk

Sum av klordan-relaterte forbindelser (klordaner) var under 0.5 ng/g våtvekt for alle fiskeprøvene fra fastlandet. Gjennomsnittskonsentrasjonene var mellom 0.035 og 0.2 ng/g våtvekt og under 25 ng/g lipid. Konsentrasjonen for klordaner i røye på Bjørnøya var flere ganger høyere (4.1 ng/g våtvekt og 142 ng/g lipid).

De enkelte forbindelsene som bidro mest til konsentrasjonen av klordaner var *trans*- og *cis*-nonaklor, etterfulgt av oxyklordan og *cis*-klordan. Klordan, heptaklor og heptaklor epoksid fant man ikke i prøvene. I røye fra Bjørnøya var det hovedsakelig oxyklordaner og *trans*-nonaklor som utgjorde klordanene.

Andre pesticider i fisk

Deldrin kunne ikke påvises i alle prøvene av sik og abbor, men der hvor dieldrin var påvisbar, var konsentrasjonen i de to artene i samme område, fra 0.02 til 0.07 ng/g våtvekt. Konsentrasjonen av dieldrin i røye fra Bjørnøya var flere ganger høyere enn i de andre fiskeprøvene (0.43 ng/g våtvekt), men her kan det ha vært forstyrrelser i analysene.

Pesticidene aldrin, endrin og trifluarin var ikke målbare i noen av de analyserte prøvene. Endosulfan ble kvantifisert i alle prøvene, med konsentrasjoner fra 0.003 til 0.03 ng/g våtvekt.

PAH i sedimenter

PAH konsentrasjonene var generelt lave. På fastlandet ble de høyeste Σ PAH nivåene i overflatesediment funnet i Nordland (4 til 7 μ g/g tørrvekt). I Troms og Finnmark var Σ PAH konsentrasjonene lave, mellom 0.1-1.0 μ g/g tørrvekt. Konsentrasjonene av Σ PAH i innsjøene på Spitsbergen og Bjørnøya var litt høyere (henholdsvis 0.9 og 1.2 μ g/g tørrvekt) enn i Troms og Finnmark.

Konsentrasjonene av all individuelle forbindelser var høyest i sedimenter fra Nordland. Konsentrasjonene av nafatalen var lavere i prøvene fra Finnmark og Sør-Varanger enn i Troms, Nordland, Bjørnøya og Spitsbergen.

1. Summary and Conclusions

The Arctic Monitoring and Assessment programme (AMAP) was established in 1991 at the first Arctic Ministerial Conference in Rovaniemi, Finland, as a part of the Arctic Environmental Protection strategy (AEPS). The main task of AMAP is the continuous monitoring of the levels of anthropogenic pollutants in all components of the Arctic environment including humans, and to assess the effects of these on the ecosystem.

The programme is divided in five sub-programmes: atmosphere, terrestrial, fresh water, marine and human health.

According to the AMAP sub-programme for freshwater, the elements to be studied in the monitoring of pollution in remote areas are lake sediments and fish. The programme also includes a listing of heavy metals and persistent organic pollutants (POPs) which should be given priority for monitoring.

The study is carried out according to the AMAP recommendations on methods for field sampling, analysis and parameters. Sampling was carried out in 1993, 1994 and 1995.

An overview over the station numbers and lakes included in this investigation, their location and morphological characteristics, as well as the kind of samples collected and the year of sampling, is given in Appendix 1. Heavy metals were analysed in sediment from 60 lakes. In addition, data material on heavy metals in lake sediments from investigations in 1992 (Rognerud *et al.* 1993) are included, giving a total number of 92 investigated lakes. Concentrations of POPs were analysed in sediments from 22 lakes.

This main report presents the levels and distribution of contaminants in lake sediments and fish on the Northern Norwegian mainland, as well as the Arctic Norwegian islands, Spitsbergen and Bear Island. The results are presented graphically and discussed. The aim of the study is to present a baseline investigation of the present situation in Northern Norway. The report also provides reference values for future studies. In addition, the study aims to augment the existing database on contaminants in the Arctic regions of Norway.

The original data from the laboratory analyses for contaminants are given in two separate appendix reports. One for Appendix report A presents the data from sediment analyses and one for Appendix report B for the fish analyses.

Field work, data analysis and reporting is financed by the Norwegian Pollution Authority (SFT), Finnmark Regional Council, the Regional Authority for Namdalen and Northern Norway, as well as Akvaplan-niva's Institute Programme. This report forms part of Norway's contribution to AMAP.

Heavy metals in sediments

Concentrations

The levels in the upper sediment layer for most heavy metals studied in lakes in the northern part of Norway are generally lower than those found in the southern parts of the country (except for Ni concentrations). Highest levels of Cd in lake sediments in the study were found in Holmevatn in Nordland County (1.46 µg/g). Highest concentrations of Hg were found in Vikvatnet (0.43 µg/g), and highest concentrations of Pb and Ni were found in Nordlivatn, Nordland County (131 µg/g) and Bjørnevatn in Finnmark County (372 µg/g) respectively.

Concentrations of Ni show an increase with latitude. Rognerud & Fjeld (1990) showed the mean concentrations in the south-eastern part of Norway to be 15 - 20 mg/kg dry weight. In this study, the median concentrations of Ni in the top sediment were found to be from 25 - 58 mg/kg dry weight. No analyses of the elements Al, Cu, Fe, Sb, Zn, Ti and Se, have been carried out in national surveys which can be compared with the results of this study.

Average concentrations of Hg, Pb and Cd in the upper layers of lake sediment studied on the Arctic islands are generally lower than the levels found on the mainland of Northern Norway.

Enrichment factors (K_f)

The upper sediment layers in lakes on the Northern Norwegian mainland are enriched with the heavy metals Hg, Pb, Sb, and Cd. The central areas appear to be less enriched compared to the southern and northern parts of the study area. However, such conclusions cannot be drawn for the other elements in this study. The K_f factors for Hg and Pb respectively show an 1.5 - 10 and 1.5 - >10 fold increase from the 2 - 3 cm layer to the 0 - 1 cm layer indicating a high variation between lakes in deposition and sedimentation in the region over the past decades. The other metals in this study do not show such a general trend.

Median enrichment factors (K_f), for metals, based on values from all lakes sampled in the different areas on Spitsbergen and Bear Island are calculated. The data shows the upper sediment layers in lakes on Spitsbergen to be enriched by the heavy metals Hg, Pb, Se, Ti and Zn. On Bear Island, antimony (Sb) is the only element which shows enrichment of the upper sediment layer.

The geographical distribution of the levels of most metals in lakes on the mainland indicate some spatial trends. For the elements Al, Cd, Hg, Pb, Sb, Se, and Zn, the concentrations were highest along the coasts of Nordland and Troms Counties. Steinnes *et al.* (1994) also found indications of similar geographical patterns of Pb, Sb and Se levels in mosses. It has been demonstrated that a significant input of Se in Norway originates from the marine environment (Steinnes *et al.* 1994). Levels of As, Cd, Hg, Pb, Sb, and Zn in mosses were found to be determined mainly by long-range atmospheric transport (Berg *et al.* 1995).

Concentrations of the elements Hg, Ni and Pb were elevated in Sør-Varanger, and the area was classified as being moderately to very severely affected. This contamination is related to local emission sources on the Kola Peninsula, mainly the nickel industry (Steinnes *et al.* 1994; Rognerud *et al.* 1993)

The distribution of Hg in sediment does not appear to be related to the distribution pattern in mosses. Steinnes & Andersson (1991) and Steinnes (1995) found that concentrations of Hg in mosses doubled from the inland of Troms and Finnmark towards the northern coast of Finnmark. A possible explanation proposed for this was the increasing significance of dry deposition of Hg^0 , under conditions where re-emission to the atmosphere is reduced. In this study the levels of Hg in the sediments do not show any significant gradient from the inland towards the coast. The highest concentrations in surface sediment were found along the coasts of Nordland and Troms and in Sør-Varanger (0.1 - 0.2 $\mu g/g$, compared to concentrations mainly below 0.1 $\mu g/g$ in Finnmark). Some lakes in these areas also had elevated enrichment factors for Hg (strongly to severely affected). The higher levels in lake sediments along the coast appear to be related to higher precipitation.

Enrichment factors for Pb were high in many lakes on the Lofoten-Vesterålen and Senja Islands, as well as in the municipalities of Kautokeino and Sør-Varanger.

Two lakes had elevated enrichment factors for several metals, these being Lakes Ø. Kaperdalvatn (68) at Senja and Ryggedalsvatn (75) in Vesterålen. These lakes were moderately to very severely affected by Al, Cd, Cu, Fe, Hg, Pb, Sb, Se and Zn. In the top sediment, most metal concentrations were only slightly higher than in samples from surrounding lakes. Low concentrations of most metals in the reference sediment layer lead to high enrichment factors (calculated from the ratio between the concentration in the top layer and the concentration in the reference layer).

Lake Storvatn close to the town of Alta, also had elevated concentrations of Ni, Cu and Pb. This is most likely due to previous mining activities in the catchment area of this lake.

Mercury in fish

Hg was analysed in fish from Lake Ellasjøen on Bear Island, 8 lakes and the Kautokeino River in Finnmark County and Lake Finnsnesvatn in Troms County.

Concentration levels

Highest concentrations of Hg in this study were found in pike from the Kautokeino River, in perch from Lake Gavdujavri, in whitefish from Lake Ravdujavri, and in Arctic char from Ellasjøen.

Predatory fish, such as pike and large perch, had the highest concentrations of Hg. The concentration in pike appeared to increase with body size, for fish larger than about 50 cm in length. The same applies to perch above approximately 20-25 cm. In the case of pike and perch, there was a significant correlation ($p < 0.05$) between fish length and the concentration of Hg in muscle tissue. For Arctic char in this study, the Hg concentration was not significantly correlated with the length of the fish. The concentrations of Hg in whitefish were lower than in the other species and were approximately similar in all the lakes sampled.

One of the reasons for the difference in Hg levels between fish species is that perch and pike generally have different feeding habits from most Arctic char and whitefish. Pike and perch, especially the larger individuals, are predatory fish, feeding mainly on prey fish. Most Arctic char and whitefish feed on a lower trophic level, with zooplankton or bottom-dwelling animals being the main food sources.

For whitefish, there was no general significant correlation between body length and mercury levels in the fish muscle. However, Hg concentrations in spawning whitefish from the Lake S. Galdinjavri were found to be significantly different from non-spawning individuals ($p = 0.0001$). There was also a significant length difference between spawning and non-spawning fish. These results indicate that in this lake, the Hg concentration in the whitefish muscle tissue was primarily related to the length, rather than the state of maturity of the individuals.

Comparing whitefish from all the lakes investigated, statistical analysis showed that there were no significant differences in Hg concentrations between male and female whitefish. There was neither a significant length difference between male/female).

Mercury in sediment versus mercury in fish

There were indications of a positive relationship between Hg concentration in perch and in the sediment. However, due to the scarcity of data available, this observation should be treated with some caution.

This study indicates that there is no positive correlation between the concentration of Hg in sediment and the average Hg concentration in whitefish. In general, two types of whitefish are found in Finnmark, one exploiting mainly benthic organisms as prey while the other mainly feeds on plankton in the pelagic zone. The bottom-feeding whitefish tend to grow to a larger body size than the pelagic-feeding type. The higher Hg concentration in Ravdujavri fish could be explained by different feeding habits between the whitefish populations. It is known from literature that the contamination levels in fish increase with the trophic level of their prey (Kidd *et al.* 1995).

Mercury and selenium

The fact that the lowest Hg concentrations were found in fish sampled from the lake with the highest levels of Hg in sediment, may partly be explained by the Se concentrations in the sediment. High concentrations of Se appear to reduce the uptake of Hg by biota (Schindler *et al.* 1995; Rognerud & Fjeld 1990). Se may function as an antagonist, occupying the same binding sites as mercury. Rognerud & Fjeld (1990) also found a negative correlation between the concentration of Se in the sediment and that in fish from the same lake. In this investigation, we analysed Se concentrations in sediments from two of the lakes which had whitefish (Table 10). Although based on scarce data, a similar trend is indicated. The lake containing fish with the highest concentrations of Hg has the lowest concentrations of Se in the sediment, while the sediment in the lake with the lowest Hg concentration in the whitefish has the highest levels of Se.

In accordance with the latest EU guidelines, Norway has established new guidelines for Hg levels in fish products (Social and Health Department 1995). Thus the average concentration of Hg in fish products used for consumption should not exceed 0.5 mg/kg wet weight. For pike and lipid-rich species, a higher maximum limit of 1.0 mg/kg wet weight is defined.

All char and whitefish, as well as the small to medium size perch and pike in this investigation, contained Hg concentrations below 0.3 mg/kg wet weight. The largest pike (in the Kautokeino River) and the largest perch (in Gavdujavri) had concentrations of 0.52 mg/kg and 0.47 mg/kg respectively.

POPs in sediments

PCBs and HCBz

The results from the previous study show that the PCB concentrations in surface sediment from lakes on the mainland of Northern Norway are generally relatively low. The concentrations of Σ_7 PCB in sediments of investigated lakes were all below 15 ng/g dry wt. The highest Σ_7 PCB levels in Nordland County were found in Lakes Storvatnet (14 ng/g dry wt) and Holmevatn (11 ng/g dry wt). In Troms County, the highest concentration was found in Skøvatnet (10 ng/g dry wt), and in Finnmark County in Andrevann (11 ng/g dry wt).

The highest concentration of Σ_7 PCB in the surface sediment was recorded on Bear Island in the Arctic region. Lake sediments in Ellasjøen had a Σ_7 PCB level of 32.7 ng/g dry wt.

The HCBz concentrations in surface sediment from lakes both in Nordland and Troms Counties and on the Arctic islands were also relatively low. The levels in lakes in the inland part of Finnmark County are significantly elevated compared to concentrations in other areas.

Pesticides

Levels of pesticides in surface sediments were generally low.

The highest detectable concentrations of γ -HCH (lindane), from 0.56 - 0.7 ng/g dry weight, were found in Haukesjøen, Rabbvatnet and Ravdujavri in Finnmark County.

Elevated concentrations of Σ DDT and p,p'-DDE in surface sediment were found in Lakes Rabbvatnet and L. Ropelvatn in Sør-Varanger, Finnmark County (3 - 3.4 ng/g dry weight) and in Ellasjøen on Bear Island.

POPs in fish

PCB levels

Extremely high PCB concentrations, one hundred times higher than the second highest concentration in this study (on wet weight basis), were detected in muscle tissue from Arctic char from Bear Island. The total concentration of the seven Dutch PCBs (Σ_7 PCB) was found to be 1292 ng/g on a wet weight basis, and on a lipid basis 50 000 ng/g extractable lipid. Due to the extreme value, this sample was reanalysed several times by the NILU laboratory for quality control, resulting in the same extremely high level. This level is several times higher than the worst case reported from the American Arctic. Even if this concentration is based on a limited material (one fish), and needs to be verified, this level is so high that it must be classified as alarming.

In other samples of Arctic char from Spitsbergen, the Σ_7 PCB ranged from 2.4 to 34.4 ng/g w.w. and from 64 to 5182 ng/g extractable lipid. Concentrations of over 1000 ng/g extractable lipid were only measured in Arctic char from Lakes Linnèvvann and Kongressvatn.

In whitefish, low concentrations were measured, all below 1 ng/g on a wet weight basis and between 45 and 200 ng/g extractable lipid. In perch, concentrations of Σ_7 PCB were higher than in whitefish, up to 6 ng/g w.w., with averages for the two lakes of 2 and 3 ng/g w.w. and 25 to 700 ng/g lipid.

Arctic char from Spitsbergen

Arctic char from lakes on Spitsbergen show some characteristic growth patterns. Land-locked populations in particular have a slow growth rate and a high longevity. One of the effects of growth of the fish on contamination levels is growth dilution (Hammar *et al.* 1993). Large, slow growing fish generally accumulate higher concentrations of organochlorines than smaller, faster-growing fish do (Schindler *et al.* 1995). The oldest fish in our material was 22 years old at the time of capture, and contained very high Σ_7 PCB concentrations of 34 ng/g w.w. and 4% lipid.

In Linnèvvann and Kongressvannet, very small fish were collected, which were 9 years old and weighed only 15 g. The pooled samples of these 'dwarf forms' contained very little lipid (0.45 and 0.98% respectively), compared to larger specimens from the same lakes (between

3.5 and 4.5% extractable lipid). The concentration of Σ_7 PCBs in these dwarf fish was around 20 ng/g w.w. and from 1500 to over 5000 ng/g lipid. Statistical analysis showed that there was a significant negative correlation between the PCB concentrations in Arctic char from Spitsbergen and their lipid content, weight and growth ($p < 0.0001$; $R = \text{circa } -0.8$).

HCBz in fish

The HCBz concentrations in fish from lakes on the mainland of Northern Norway were all at the lower end of the scale. However, the levels in fish in lakes on the Arctic islands were elevated compared to concentrations in other areas. The highest concentrations of HCBz were found in Arctic char from Rickardvannet on Spitsbergen (6.8 ng/g wet weight, 322 ng/g, extractable lipid).

HCH in fish

The concentration of γ -HCH on a wet weight basis was consistently relatively low in the species analysed, with both perch, Arctic char and whitefish containing around from 0.05 ng/g w.w to 0.18 ng/g w.w and between 4 and 22 ng/g extracted lipid. As γ -HCH is a more polar compound than for example the more hydrophobic PCBs, γ -HCH has a lower tendency to bio-accumulate in fish (Lockhart *et al.* 1992). The γ -HCH concentrations in biota from remote areas are typically lower than concentrations of PCBs or DDTs.

The levels of α -HCH were not quantifiable in most samples, as concentrations were below 0.06 ng/g w.w. Concentrations of α -HCH which could be quantified in whitefish and perch samples were less than 0.10 ng/g w.w. In Arctic char from Bear Island, the concentration of α -HCH was 0.40 ng/g w.w.

DDTs in fish

DDT is an insecticide which is very toxic, and accumulates in biota, thereby becoming magnified in the food chain. The use in western countries has been greatly reduced since the 1970's

All concentrations of Σ DDT in perch and whitefish from Finnmark were below 1 ng/g w.w., with average concentrations between 0.17 and 0.6 ng/g w.w., and below 90 ng/g on a lipid basis. Of the DDT components and their metabolites, the largest proportion was made up of p,p'-DDE. The concentration of p,p'-DDE comprised approximately 80% of the Σ DDT concentration in perch and whitefish. The concentration of Σ DDT in Arctic char from Bear Island was alarmingly high (76.4 ng/g w.w., 2 634 ng/g lipid), being two orders of magnitude higher than in the other fish. This extremely high concentration was also confirmed by reanalyses. Here p,p'-DDE made up about 97% of the Σ DDT concentration.

Chlordanes in fish

The total concentrations of chlordane-related compounds (Σ chlordanes) were below 0.5 ng/g w.w. for all samples from lakes on the mainland. The average concentrations were between 0.035 and 0.2 ng/g w.w. and below 25 ng/g lipid. Concentrations in perch and whitefish were in the same range. However, the concentration of Σ chlordanes in Arctic char from Bear Island was more than a order of magnitude higher (4.1 ng/g w.w. and 142 ng/g lipid).

The individual compounds which contributed most to the concentrations of Σ chlordanes were *trans*- and *cis*-nonachlor, followed by oxychlordane and *cis*-chlordane. Chlordane, heptachlor

and heptachlor epoxide were not detectable in the samples. In Arctic char from Bear Island, oxychlordanes and *trans*-nonachlor were the major contributors to Σ chlordanes.

Other pesticides in fish

Dieldrin could not be detected in all of the whitefish and perch samples, but where dieldrin was detectable, the concentrations in the two species were in the same range, ranging from 0.02 to 0.07 ng/g w.w. However, in most samples, the recorded value deviated more than 20% from the theoretical value, due to interference or instrument noise. The analysed concentration of dieldrin in Arctic char from Bear Island was several orders of magnitude higher than the other fish samples (0.43 ng/g w.w.) but some interference may have been present.

The pesticides aldrin, endrin and trifluralin were not quantifiable in any of the samples analysed. Endosulfan was quantified in all samples, with concentrations ranging from 0.003 to 0.03 ng/g w.w.

PAH in sediments

The PAH concentrations were generally low. On the Norwegian mainland, the highest Σ PAH levels in surface sediment were found in Nordland (4 to 7 μ g/g dry wt.). In Troms, Finnmark and Sør-Varanger, the Σ PAH concentrations were low, between 0.1-1.0 μ g/g dry wt. Concentrations of Σ PAH in the lakes on Spitsbergen and Bear Island were somewhat higher (0.9 and 1.2 μ g/g dry wt. respectively) than the low levels found in Troms and Finnmark.

Concentrations of all individual compounds were highest in sediments from Nordland. The concentrations of naphthalenes were lower in samples from Finnmark and Sør-Varanger than in Troms, Nordland, Bear Island and Spitsbergen.

2. Introduction

The Arctic Monitoring and Assessment programme (AMAP) was established in 1991 as a part of the Arctic Environmental Protection strategy (AEPS) at the first Arctic Ministerial Conference in Rovaniemi, Finland. The main task of AMAP is the continuous monitoring of the levels of anthropogenic pollutants in all components of the Arctic environment including humans, and to assess the effects of these on the ecosystem.

The programme is divided in five sub-programmes; atmosphere, terrestrial, fresh water, marine and human health.

According to the AMAP sub-programme for freshwater, the elements to be studied in the monitoring of pollution in remote areas are lake sediments and fish. The programme also includes a listing of heavy metals and persistent organic pollutants (POPs) which should be given priority for monitoring. The POPs analysed are a range of polychlorinated biphenyls (PCBs) congeners as well as organochlorine pesticides. Also PAHs were analysed.

The study is carried out according to the AMAP recommendations on methods for field sampling, analysis and parameters. Sampling was carried out in 1993, 1994 and 1995.

This main report presents the levels and distribution of contaminants in lake sediments and fish on the Northern Norwegian mainland, as well as the Arctic Norwegian islands Spitsbergen and Bear Island. The results are presented graphically and discussed. The aim of the study is to present a baseline investigation of the present situation in Northern Norway. The report also provides reference values for future studies. In addition, the study aims to augment the existing database on contaminants in the Arctic regions of Norway. Therefore, the emphasis of this report is on presenting levels and geographical distributions of contamination levels in sediment and fish from Northern Norway, rather than discussing the processes behind these levels.

The original data from the laboratory analyses for contaminants are given in two separate appendix reports. Appendix report A presents the data from sediment analyses ("Heavy metals, POPs and PAH in sediment from lakes in Northern and Arctic regions of Norway. Analytical results from laboratory reports.") and Appendix report B contains the fish analyses ("Mercury, POPs and PAH in fish from lakes in Northern and Arctic regions of Norway. Analytical results from laboratory reports.").

Field work, data analysis and reporting were financed by the Norwegian Pollution Authority (SFT), Finnmark Regional Council, the Regional Authority for Namdalen and Northern Norway, as well as Akvaplan-niva's Institute Programme. This report forms part of Norway's contribution to AMAP.

3. Material and methods

3.1 Selection criteria

The lakes were selected after evaluation of existing regional data on geochemistry, topography/precipitation, general water quality, pollution status, deposition of long-range atmospheric pollution, and distance from sources of atmospheric pollution. Earlier sampling stations were also considered in the design of the geographical distribution.

3.1.1 Geochemistry

The Geological Survey of Norway (NGU) are in possession of extensive data concerning the amounts of heavy metals in geological material in the North (Bergstrøm *et al.* 1986, Ottesen *et al.* in prep). The sampling stations were selected to represent the natural variations.

3.1.2 Atmospheric pollution

The distribution of long-range atmospheric pollutants in Norway has been mapped, based on monitoring of heavy metals and persistent organic pollutants (POPs) in ambient air at meteorological stations as well as concentrations of heavy metals in mosses (Steinnes *et al.* 1994, Oehme *et al.* 1995). The distribution gradients of these contaminants have been taken into account in the selection of lakes.

3.1.3 Water quality

Water quality influences the behaviour of pollutants, in particular the mobility of heavy metals and their toxic impact on organisms within the ecosystem. Published data on water quality in lakes in Norway (Henriksen *et al.* 1987) have been considered when selecting the lakes to be monitored. Of particular interest are areas with similar atmospheric depositions, but differing geochemistry and water quality.

3.1.4 Precipitation pattern and lake morphology

Precipitation (wet deposition) plays an important role in scavenging most pollutants from the atmosphere. Because of this, lakes in areas with high precipitation rates along the coast, as well as those in areas of low precipitation in the inland are included in the sampling programme. The lakes sampled should be deep enough for the existence of an accumulation area. Accumulation areas do not occur at specific depths, but their location and distribution varies with the area and morphology of the lake, as well as the 'effective wind-exposure'. Topographical maps were therefore used to select lakes which were expected to have an accumulation-area. Before sampling, the bottom morphology of the lakes concerned were documented by echo-sounder. During the sampling it was verified whether a lake contained an appropriate accumulation area and could be accepted for the programme.

3.1.5 State of pollution in the study-area

Most of the selected lakes are remote headwater lakes in areas with no local pollution. However, the lakes selected from the north-eastern part of the study area (Sør-Varanger) have previously been shown to be polluted by emissions from smelter industry close to the Russian-Norwegian border (Rognerud *et al.* 1993). These lakes were monitored in order to follow local contamination trends.

3.2 Sediments

3.2.1 Sampling locations

An overview over the station numbers and lakes included in this investigation is given in Figure 1 and Table 1. Their geographical position and morphological characteristics, as well as the kind of samples collected and the year of sampling, are given in Appendix 1. Heavy metals were analysed in sediment from 60 lakes. In addition, data material on heavy metals in lake sediments from investigations in 1992 (Rognerud *et al.* 1993) are included, giving a total number of 92 lakes investigated for heavy metals. Concentrations of POPs were analysed in sediments from 22 lakes.

Table 1. Lakes and station numbers included in this investigation.

No.	Lake name	No.	Lake name	No.	Lake name
FINNMARK				TROMS	
1	Storvatnet	32	Vuorasjav`ri	62	Svarthammervatnet
2	Hestekovatnet	33	Avzejav`ri	63	Y.Kårviksvatnet
3	Storvatnet	34	Lavvujav`ri	64	Storvatnet
4	Doggejav`ri	35	Guotkojav`ri	65	Josvatnet
5	Ø. Saltvatn	36	S.Galdinjav`ri	66	Langfjordvatn
6	Hamnevatn	37	Ravdujav`ri	67	Tårnvatn
7	L. Havvatn	38	Gavdujav`ri	68	Ø. Kaperdalsvatn
8	Lafjordvatnet	39	Røyvatn	69	Skøvatnet
9	Kaifjordvatnet	40	St.Ingasjarvi	NORDLAND	
10	Skogsfjordvatnet	41	Vouddajav`ri	70	Langvatnet
11	Gussajav`ri	42	Vuostamusjav`ri	71	Skallavatnet
12	Cullujav`ri	43	Gædgesuolojav`ri	72	Holmevatn
13	Kjæsvatnet	44	Korpvatn	73	Vikvatnet
14	Vasavatnet	45	Gardsjøen	74	Storvatnet
15	Syltevikvatnet	46	Hundvatn	75	Ryggedalsvatnet
16	Oardujav`ri	47	Ødevatn	76	Storevatnet (Bleikvt.)
17	Hergevatn	48	Ellevatn	77	Ågevatn
96	Magistervatn	49	Følvatn	78	Grønåsvatn
18	Store Kløftvatnet	50	St. Spurvvatn	79	Markvatn
19	Diergejav`ri	51	St. Sametjern	80	Storvikvatnet
20	Gålgutjav`ri	52	Coalbmejav`ri	81	Valnesvatnet
21	Maskejav`ri	53	L.Ropelvvatn	82	Nordlivatn
22	Baisjav`ri	54	Vegvatnet	83	Kilevatn
23	Suolujav`ri	55	Gravsjøen	84	Tennvatn
24	Lævvajav`ri	56	Langvatnet	SPITSBERGEN	
25	Stourrajav`ri	57	Rabbvatnet	87	Revvatn
26	Stoppuluobbal	97	Haukesjøen	85	Istjøna
27	Doulbmajav`ri	58	Holmevatnet	86	Vann 210
28	Oksevatnet	59	Vaggatem	90	Barentsvann
29	Kibergvatnet	60	Bjørnevatn	88	Kongressvann
30	Vuovddejav`ri	61	Midtre Pumpehusvatn	89	Linnèvann
31	Cuolbmajav`ri			93	Rickardvannet
				94	Hornsundet
				95	Diesetvannet
				BEAR ISLAND	
				91	Ellasjøen

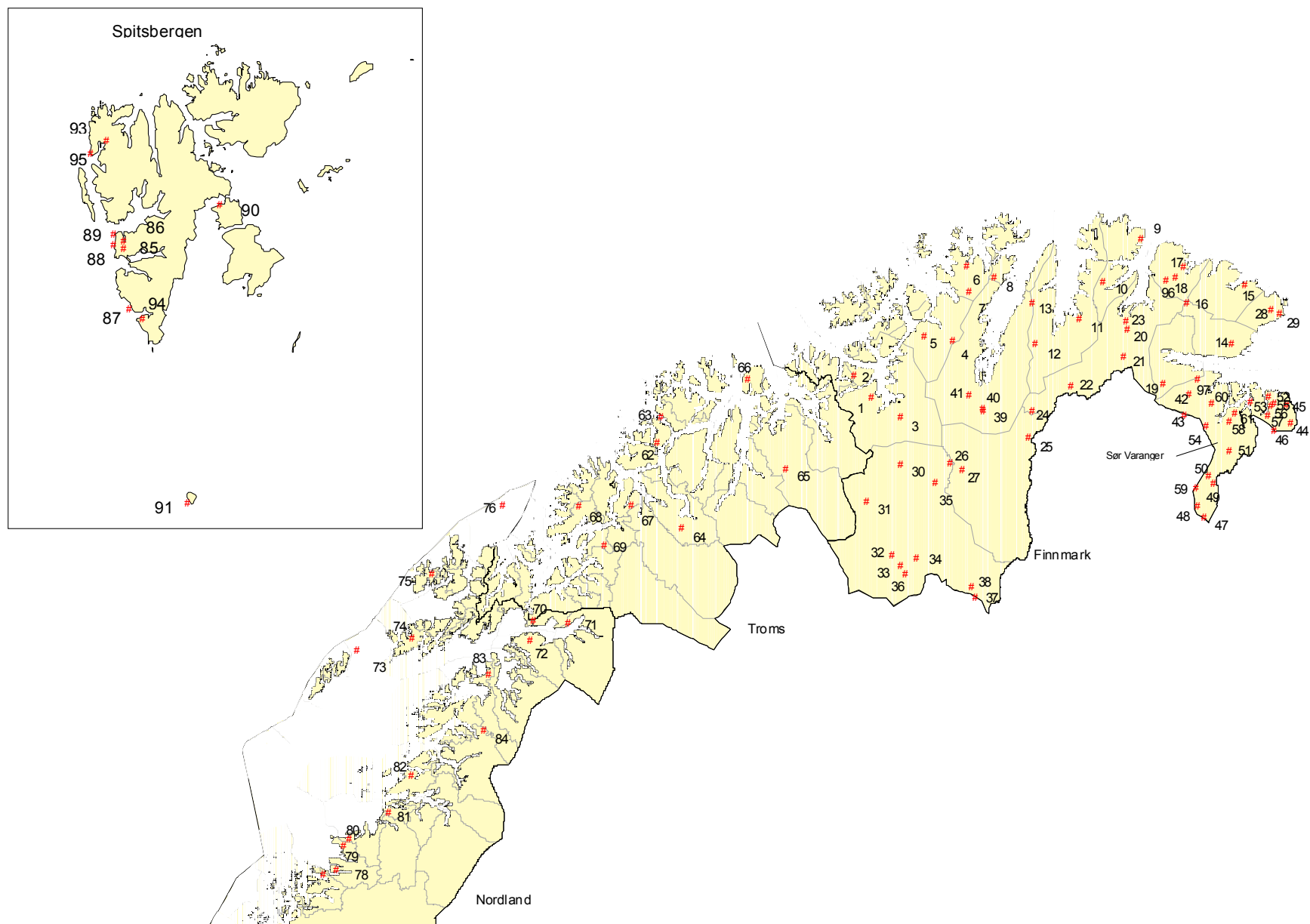


Figure 1. Lakes and station numbers included in this investigation.

3.2.2 Sample collection and treatment

Sediment samples were collected from a total of 94 lakes in the Counties of Nordland, Troms and Finnmark, as well as at Bear Island and Spitsbergen. Samples were collected during the summer, from 1992 to 1995. The sediment samples were taken from the deepest point in the lake, using a sediment corer (Skogheim 1979). The sampler is equipped with exchangeable plexiglas tubes (44 mm internal diameter), within which sediment cores may be transported in an undisturbed condition to the laboratory for further treatment. Sediment cores were sectioned into 1 cm thick slices at, or near the sampling site, then put into acid-treated polyethylene tubs (metals) or acid-treated and ignited glass jars (POPs), and frozen at -18 °C.

For metal-analyses, the upper 3 slices of each lake were analysed, together with the deepest, "reference" slice, from about 20 cm depth. The latter is defined as the "reference layer", likely to have been deposited in pre-industrial times. The levels of metals in this reference layer are therefore representative of background conditions.

To obtain sufficient sediment material for analysis of POPs, samples from the same sediment depth were combined from several cores. In deep Arctic lakes with accumulation areas, several slices from the same core depth can be combined provided that the cores are sampled from the accumulation zone (as defined in Håkanson & Jansson 1983). Several studies confirm that analyses of separate samples give similar results if the cores are collected from the accumulation area in a lake (Coker & Nichol 1975, Rognerud & Fjeld 1993).

3.2.3 Contaminant analyses

The sediment samples were then freeze-dried and analysed for loss of ignition, metals (Fe, Zn, Ti, As, Hg, Ni, Cu), and polycyclic aromatic and organochlorine compounds (PAH, PCBs, HCBz, DDTs, HCHs, and other pesticides). All concentrations in sediment are given as dry weights. In the case of concentrations below the detection limit, the value of the detection limit was used in the calculations and presentations. As a result of this conservative treatment of the low concentrations, the values presented represent maximum possible concentrations for the sample in question.

Sediment dry weight was determined after drying the sediment at 105 °C at the NIVA laboratory in Oslo. Subsequently, the samples were combusted at 550 °C, and the loss of ignition (LOI) was calculated from the weight loss. Statistical tests have shown that organic carbon (OC), together with water from crystallised iron-hydroxide, are included in the values for LOI. Organic carbon content was therefore estimated from the loss of ignition and the percentage of iron in the sediment (FeSE), using the formula $OC = (LOI - 0.48 \text{ FeSE})/1.9$ (Ranem & Hongve 1980).

Metal content in sediments were analysed at the Institute of Energy Technology (IFE) in Norway and at Svensk Grundämnesanalys AB (SG) in Sweden. Freeze-dried sediment was dissolved in nitric acid (0.5-1 g HNO₃ per 10 ml) under pressure (120 °C). Hg and Se were analysed by neutron activating analyses (NAA), while all other elements were analysed using an atom adsorption spectrophotometer (AAS).

PCBs, chlorinated pesticides and PAH in sediments were analysed at the Norwegian Institute for Water Research (NIVA) and at the Norwegian Institute for Air Research (NILU). Both laboratories have Norwegian Accreditation status, and conduct the analyses under the European Quality Assurance Protocol EN 45001 (CEN/CENELEC 1989). The two laboratories were, however, using different methods.

At the NIVA laboratory, the material was freeze dried and homogenised, after which PCB-53 was added as an internal standard. The sample was twice extracted with a mixture of cyclohexane and acetone, using high energy ultrasonic disintegration (475 W). Samples were centrifuged and the centrifugate /organic layer was evaporated and dried, for lipid weight determination. The samples were then dissolved with dichlormethane and further purified with gel permeation chromatography followed by sulphuric acid treatment. Single components were identified and quantified by a gas chromatograph (GC) with a 60 m capillary column and an electron capturing detector (ECD). This method was used for the sediment samples collected in 1993 and '94. Those were from Nordland and Troms Counties, as well as 3 lakes on Spitsbergen and Bear Island. The detection limit was at 0.1 or 0.5 ng/g dry weight.

The NILU laboratory analysed the sediment samples from lakes in Finnmark, collected in 1995. NILU analysed more PCB congeners and persistent organic pesticides than NIVA (Table 2). Some of the compounds could be analysed below 0.010 ng/g dry weight. The methods used (method NILU-O-2 and 3) were as follows: 5-30 g wet sediment was filtered to remove water. The filtered sediment was weighed, internal standards (C_{13} -marked isotopes) were added and soxhlet extracted in a cleaned cellulose case with acetone.

For treatment for PCB and pesticide analyses, the sample was extracted again with cyclohexane. After the extraction the filter and case were dried. Both extracts were mixed, sodiumsulphate was added and the sample was reduced to 1 ml. The sample was split in two. One fraction was treated with acid (concentrated sulphuric acid) to analyse acid stable pesticides. The other fraction was saponified (lye, ethanol and distilled water) for analyses of acid labile pesticides. Each of the two organic fractions was reduced to 500 ml and eluted over sodium sulphate and silica with diethylether/hexane. The sample volumes were reduced again, recovery standard was added and analyses were carried out by high resolution gas chromatography coupled with mass spectrometry (HRGC/MS).

PCBs and DDTs were identified and quantified by two separate GC analyses. The gas chromatograph coupled with a high resolution mass spectrometer in electron-impact mode (EI). Chlorinated pesticides were quantified by two separate GC analyses with low resolution mass spectrometry and negative ionisation chemical ionisation (NICI). The identification of the compounds with high and low resolution mass spectrometry was carried out in selected ion monitoring (SIM), by registration of M and M+2 ions for each substance.

For treatment for PAH-analyses, the sample was further extracted with toluene. After extraction both filter and case were dried to decide the dry weight of the sample. The extracts were mixed, concentrated and transferred to cyclohexane and liquid-extracted with dimethylformamide/cyclohexane. In addition, the end extract was cleaned by HPLC with a deactivated silica column. The HPLC-extract volume was reduced, recovery standard was added, and the sample was analysed by gas chromatography and low resolution mass spectrometry (GC/MS) with electron impact mode (EI). One main fragment ion was registered for each PAH-compound.

Table 2. Contaminants analysed by the NILU and NIVA laboratories, in sediment from Northern Norway.

PCBs	(IUPAC-no.)		Pesticides		PAHs	
	NILU	NIVA	NILU	NIVA	NILU	NIVA
TriCB	18		α -HCH	5-CB α -HCH	Naphthalene	x
	28	28	β -HCH		2-Methylnaphthalene	x
	31		γ -HCH	γ -HCH	1-Methylnaphthalene	x
TetCB	47		Chlordane		Biphenyl	x
	52	52	Heptachlor		2, 6-Dimethylnaphthalene	x
	60		Heptachlor epoxide		Acenaphthylene	x
	66		Oxychlordane	OCs	Acenaphthene	x
PenCB	74		trans-Chlordane		2,3,5-Trimethylnaphthalene	x
	99		cis-Chlordane		Dibenzofuran	
	101	101	trans-Nonachlor		Fluorene	x
	105	105	cis-Nonachlor		Dibenzothiophene	
	114		Dieldrin		Phenathrene	x
HexCB	118	118	Aldrin		Anthracene	x
	123		Endrin		2-Methylphenathrene	
	128		Trifluralin		2-Methylanthracene	
	138	138	a-Endosulfan		1-Methylphenathrene	x
	149		o,p'-DDE		Fluorantene	x
	153	153	p,p'-DDE	p,p'-DDE	Pyrene	x
	156	156	o,p'-DDD		Benzo(a)fluorene	
HeptCB	157		p,p'-DDD	p,p'-DDD	Retene	
	167		o,p'-DDT		Benzo(b)fluorene	
	170		p,p'-DDT	p,p'-DDT	Benzo(g,h,i)fluorantene	
	180	180	Sum DDT		Cyclopenta(c,d)pyrene	
	187				Benz(a)anthracene	x
	189				Chrysene/triphenylene	x
	209	209			Benzo(b,j,k)fluorantene	x
HCBz					Benzo(a)fluorantene	x
					Benzo(e)pyrene	x
					Benzo(a)pyrene	x
					Perylene	x
					Ind. (1,2,3 cd)pyrene	x
					Dibenz. (a,c/a,h) anthracene	x
					Benzo(ghi)perylene	x
					Antatrene	
					Coronene	

3.2.4 Enrichment factors and pollution classes

The enrichment factor (K_f), or contamination factor expresses the relationship between the concentration of a pollutant in the recent, surficial sediment layer (K) and the natural background concentration in the “reference layer” (K_0) (Håkanson 1984):

$$K_f = K/K_0$$

The K_f factor is a measure of the degree to which a sediment sample is enriched with an element or component. When the concentrations in the reference sediment layer are below the detection limit, and the conservative value of the detection limit is used, the calculated enrichment factors may under-estimate the degree of enrichment.

In the interpretation of the results obtained, it must be emphasised that the enrichment factor (K_f) and concentrations found in this study only reflect the concentrations and degree of en-

richment of the metals analysed. These enrichment factors are not a direct measurement of anthropogenic pollution, but also include the effects of all natural enriching and diluting processes in the sediment. However, the principle of expressing the degree of impact on the basis of K_f values is generally accepted, based on the hypothesis that anthropogenic pollution is often the most important cause of high K_f values, and the fact that the degree of enrichment is the levels to which aquatic organisms are exposed, regardless of cause.

The classification of the degree of enrichment is presented in Table 1. The classification is a modification of the terminology used in Holtan & Rosland (1992).

Table 1. Classification of the degree of enrichment, based on the enrichment factor (K_f).

Degree of enrichment		K_f -factor
1 = Not or slightly enriched		< 1.5
2 = Moderately enriched		1.5 - 3
3 = Markedly enriched		3 - 5
4 = Severely enriched		5 - 10
5 = Extremely enriched		> 10

3.3 Fish

3.3.1 Sampling locations

Fish were collected from lakes on Spitsbergen and Bear Island as well as lakes and a river in the inland parts of Finnmark County and a lake in Troms County. All the lakes sampled in Finnmark are located in the municipality of Kautokeino. Appendix 1 as well as Table 2 and Table 3 present the locations where fish samples were collected.

The lakes in Finnmark and on Bear Island may be considered as being remote lakes, unaffected by local contamination sources. Pike were collected from the Kautokeino River in Finnmark County, close to the settlement of Masi. Being located in the centre of a village, Lake Finnsnesvatn in Troms County is subject to local pollution sources. It is a small and shallow lake which has become relatively eutrophic.

3.3.2 Sample collection and treatment

Fish were collected using gill nets, approximately 20 fish per lake. The fish species used for contaminant analysis were selected according to their importance for both human consumption and the ecosystem. Fish length (fork length: from snout to the end of the middle ray of the tail fin), weight, sex and stage of gonad development were recorded in the laboratory and the age was determined. Age rings in otoliths (ear bones) were generally used for age determination, but annual rings in the opercular bone (gill cover) were used for perch and rings in the shoulder bone (cleithrum) were used for pike.

The four species used for analyses of total mercury were as follows: whitefish (*Coregonus lavaretus* L.), Arctic char (*Salvelinus alpinus* L.), perch (*Perca fluviatilis* L.), and pike (*Esox lucius* L.). A summary of the locations, species and sizes of fish sampled is presented in Table 2. For analyses of Hg in fish muscle, samples were taken from 20 fish from each lake. Individual samples were analysed. At least 10 g (wet weight) of the fish muscle tissue (dorsal axial muscle) was individually stored in plastic bags and frozen until analysis.

Table 2. Locations and fish sampled for analyses of Hg in muscle tissue. Averages are calculated from 20 fish collected from each lake.

Lake	County	Fish species	Length (mm) average	Weight (g) average	Age (years) average
Avzejavri	Finnmark	whitefish	202	128	6.3
S.Galdinjavri	Finnmark	whitefish	222	146	8.0
Guotkojavri	Finnmark	whitefish	250	201	7.3
Lavvujavri	Finnmark	whitefish	219	161	7.8
Ravdujavri	Finnmark	whitefish	349	458	7.9
Cuolbmajavri	Finnmark	Arctic char	261	197	7.7
Ellasjøen	Bear Island	Arctic char	277		9.4
Gavdujavri	Finnmark	perch	220	189	10.4
Vuorasjavri	Finnmark	perch	207	130	7.8
Finnsnesvatn	Troms	perch	199	100	5.4
Kautokeino River	Finnmark	pike	597	1877	8.8

Levels of PCBs, pesticides and PAHs were analysed in muscle tissue from whitefish, perch and Arctic char. An overview of the locations and fish sampled, as well as the lipid content in the samples, is given in Table 3. Samples for analysis of organochlorines and PAHs were taken from the dorsal axial muscle (approximately 10 g wet weight). Specially cleaned equipment and vials were used for sample preparation and storage. Samples were frozen (-18 °C) until analysis.

Some PCB analyses in char from Spitsbergen were carried out in individual samples, but most analyses were carried out in pooled samples of muscle tissue. After the above described biological determinations were muscle samples from two to six fish blended. Pooled samples were taken from fish of the same size, sex and stage of maturity.

Table 3. Data on samples of fish, analysed for POPs and PAHs and the percentage of extractable lipid in the muscle tissue samples.

N = number of fish in pooled sample. * = sex not registered.

Sample code	Lake	Location	Species	N	Sex	Stadium	% extractable lipid
1	Avzejavri	Finnmark	whitefish	3	f	non-spawning	0.19
2	Avzejavri	Finnmark	whitefish	4	m	non-spawning	0.69
3	Avzejavri	Finnmark	whitefish	5	f	spawning	0.80
4	Avzejavri	Finnmark	whitefish	3	m	spawning	0.93
5	Lavvujavri	Finnmark	whitefish	3	f + m	non-spawning	2.61
6	Lavvujavri	Finnmark	whitefish	5	f	spawning	1.17
7	Ravdujavri	Finnmark	whitefish	2	f	non-spawning	0.26
8	Ravdujavri	Finnmark	whitefish	5	f	spawning	0.49
9	Gavdujavri	Finnmark	perch	3	m	non-spawning	0.44
10	Vourasjavri	Finnmark	perch	5	f	non-spawning	1.49
11	Vourasjavri	Finnmark	perch	5	f	spawning	0.80
12	Vourasjavri	Finnmark	perch	2	m	non-spawning	0.56
13	Vourasjavri	Finnmark	perch	4	m	spawning	1.56
14	Ellasjøen	Bear Island	A. char	1	f	spawning	2.90
A	Linnèvann	Spitsbergen	A. char	5	*	non-spawning	4.2
B	Linnèvann	Spitsbergen	A. char	1	*	non-spawning	3.45
C	Linnèvann	Spitsbergen	A. char	1	*	spawning	4.43
D	Rickardvannet	Spitsbergen	A. char	1	*	non-spawning	6.2
E	Rickardvannet	Spitsbergen	A. char	1	*	non-spawning	1.08
F	Rickardvannet	Spitsbergen	A. char	1	*	non-spawning	2.44
G	Rickardvannet	Spitsbergen	A. char	1	*	non-spawning	2.52
H	Linnèvann	Spitsbergen	A. char	6	*	non-spawning	0.45
I	Linnèvann	Spitsbergen	A. char	5	*	non-spawning	0.37
J	Kongressvannet	Spitsbergen	A. char	1	*	non-spawning	3.97
K	Kongressvannet	Spitsbergen	A. char	5	*	non-spawning	1.22
L	Kongressvannet	Spitsbergen	A. char	5	*	non-spawning	0.89
M	Hornsundet	Spitsbergen	A. char	1	*	non-spawning	6.16
N	Diesetvannet	Spitsbergen	A. char	5	*	non-spawning	2.94

3.3.3 Contaminant analysis in fish

Muscle samples were analysed for mercury at the NIVA laboratory in Oslo. Concentrations of total Hg in individual muscle samples were determined using cold vapour atomic adsorption (CVAAS). This method was described for sediment above. The detection limit was 0.2 ng/g wet weight, using 1 g wet weight muscle.

Persistent organic pollutants and PAHs in fish samples were analysed at the NILU laboratory.

The laboratory at NILU (Norwegian Institute for Air Research) analysed PCBs and hexachlorobenzene (HCBz) in the Arctic char from Spitsbergen. The PCB congeners analysed were the 10 selected by AMAP: IUPAC No.'s 28, 31, 52, 101, 105, 118, 138, 153, 156, and 180. The NILU laboratory also analysed persistent organic pesticides, PCBs (26 congeners) and PAHs in fish from Finnmark and Bear Island. An overview of all components analysed in fish is given in Table 4. Gas chromatography and mass spectrometry were used (Internal method No.: NILU-O-2). In addition, extractable fat (extractable lipid) was measured in muscle tissue.

Between 8-50 g fish muscle was homogenised. A sub-sample of this was taken for analysis of lipid content. The sample was dried with sodium sulphate. Internal isotope-marked standards

were added and the sample was eluted with cyclohexane/ethylacetate. After drying, the sample was purified with gel permeation chromatography to remove lipids. The extract was concentrated to 0.5 ml and fractionated on alumina with tert. butylmethylester and *n*-hexane. A recovery standard was added to the extract, which was then analysed by gas chromatography coupled with mass spectrometry (GC/MS).

PCBs and chlororganic pesticides were identified and quantified by the same methods as described above for sediment.

After addition of internal standards, the samples for PAH-analyses were hydrolysed by boiling in KOH and methanol/water. The hydrolysed sample was extracted with cyclohexane. Subsequent the sample was liquid-extracted with dimethylformamide/cyclohexane. Further treatment and analysis of the sample were carried out as described for sediment.

Table 4. Contaminants analysed in muscle tissue from fish.

PCBs	(IUPAC-No.)	Pesticides	PAHs (ng/g)
TriCB	18	α -HCH	Naphthalene
	28	β -HCH	2-Methylnaphthalene
	31	γ -HCH	1-Methylnaphthalene
TetCB	47	Chlordane	Biphenyl
	52	Heptachlor	2, 6-Dimethylnaphthalene
	60	Heptachlor epoxide	Acenaphthylene
	66	Oxychlordane	Acenaphthene
	74	trans-Chlordane	2,3,5-Trimethylnaphthalene
PenCB	99	cis-Chlordane	Dibenzofuran
	101	trans-Nonachlor	Fluorene
	105	cis-Nonachlor	Dibenzotriophene
	114	Dieldrin	Phenanthrene
	118	Aldrin	Anthracene
HexCB	123	Endrin	2-Methylphenanthrene
	128	Trifluralin	2-Methylanthracene
	138	a-Endosulfan	1-Methylphenanthrene
	149	o,p'-DDE	Fluorantene
	153	p,p'-DDE	Pyrene
HeptCB	156	o,p'-DDD	Benzo(a)fluorene
	157	p,p'-DDD	Retene
	167	o,p'-DDT	Benzo(b)fluorene
	170	p,p'-DDT	Benzo(g,h,i)fluorantene
	180	Sum DDT	Cyclopenta(c,d)pyrene
HCBz	187		Benz(a)anthracene
	189		Chrysene/triphenylene
	209		Benzo(b,j,k)fluorantene
			Benzo(a)fluorantene
			Benzo(e)pyrene
			Benzo(a)pyrene
			Perylene
			Ind. (1,2,3 cd)pyrene
			Dibenz. (a,c/a,h) anthracene
			Benzo(ghi)perylene
			Antatrene
			Coronene

In this report, PCBs are referred to as Σ_7 PCB, the sum of the so-called seven Dutch PCBs. These seven congeners are selected to represent the different types of congeners present, and are commonly used in monitoring studies. In our samples, they comprised 50 to 90% of the 26 PCB congeners analysed at NILU. This Σ_7 PCB group is used to compare our results with contamination levels published from other areas.

Levels of organochlorines in muscle tissue are presented on both a wet weight and lipid basis (lipid normalisation, dividing contaminant concentration by concentration extractable lipid). Concentration on wet weight basis are best to indicate the health risks and ecological effects, while concentrations on a lipid basis better indicate the degree of contamination and changes in time (Knutzen & Skei 1990).

3.3.4 Statistical analyses: Weighting of pooled samples

The data matrix for fish consists of contaminant analysis in single samples as well as pooled samples. Pooled samples contain fish of same sex, maturity and of similar age. We expect that there is less variance between the fish in the pooled samples than between those which differ in age and maturity. The value from a pooled sample will be closer to a representative average value than the value from a single fish. By means of weighting we obtain a lower variance than if the fish had been analyzed individually, as each pooled observation is repeated as many times as the weight tells.

To obtain an estimate of the reduction of variance and to determine the weighting which gives results most equivalent to a data set of individually analyzed fish, we used data from an article by Hammar et al. (1993). The data in this article refer to fish collected from Blåsjøen in the central part of Sweden. Statistical analyses on pooled values from three to six individuals, at the same maturity and age, were compared with analyses of individual values. From a theoretical point of view, weighting with N (the number of fish in the pooled sample), or the square root of N, seemed to be the most convenient weighting.

We found that using N for weighting consistently gave a lower variance than the full data set, leading to “decreased” significance levels. Because this decreases the total variance and preserves the total number of observations, the calculated significance levels will be “too good”.

The square root of N is the rate by which the standard deviation of the mean of the pooled samples (or standard error) decreases ($SE = SD/\sqrt{N}$). This is also a compromise between not weighting and weighting with N. Running the data set with $\sqrt{N}$ revealed some weaknesses in the statistical computer programme used, Statistica. The programme rounded the weight ($\sqrt{N}$) to an integer, with no detectable rule of rounding. For the small number of fish in the pooled samples, 3 to 6, $\sqrt{N}$ will be between 1.7 and 2.4. To approach this value we have chosen to weight the values representing a group of individuals with two instead of $\sqrt{N}$.

This double weighting tends to give a slightly higher standard deviation, which leads to more conservative results of analysis. The error in the standard deviation was relatively small for double weighting, except for the parameter fat; where the standard deviation decreased by about one third.

A data matrix of unweighted groups tend to give a slightly lower significance level than for an analysis based on a full data matrix.

It seems that double weighting of the pooled samples is preferable, as this gave significance levels most equal to the full data set.

4. Concentration determination factors in lake sediments

4.1 Sources

4.1.1 Atmospheric sources

Many investigations have shown that Arctic regions, where local emissions are scarce, are no longer pristine. They receive considerable amounts of chemical contaminants from point sources in the south (Barrie 1986; Ottar 1989). Organochlorines, PAHs and heavy metals are known to be transported from temperate latitudes either in the vapour phase or on aerosols. Today, it is widely accepted that contaminants generally reach remote freshwater systems by long distance atmospheric transport (Gregor & Gummer 1989; Barrie *et al.* 1992; Steinnes *et al.* 1994; Mackay & Wania 1995; Oehme *et al.* 1995).

4.1.1.1 Persistent organochlorine compounds

At the Norwegian Arctic islands and the Norwegian mainland, several studies of persistent organochlorine compounds in ambient air have provided evidence of long-range transport of such compounds (Oehme & Manø 1984; Oehme & Ottar 1984; Pacyna & Oehme 1988; Oehme 1991). Transport of such compounds to the Norwegian Arctic is possible from source areas in both North America and Europe (Oehme 1991). It is shown that semi-volatile organochlorines, such as hexachlorocyclohexane (HCH) and hexachlorobenzene (HCBz), in Arctic air are comparable to those in regions closer to the source areas (Pacyna & Oehme 1988). Detectable concentrations of polychlorinated biphenyls (PCBs) and chlordanes have also been determined in ambient air in the Norwegian Arctic (Oehme 1995). The processes of particle deposition of heavy metals and POPs in polar regions are mainly determined by physical factors such as particle size and distribution, surface roughness and wind force (Barrie *et al.* 1992).

The pesticide HCH is a technical mixture of α -, β - and γ -HCH, the use of which has been prohibited in most industrialised countries since the 1970's, but which is still in use in developing countries. However γ -HCH (lindane) is still used in most countries. As γ -HCH is a more polar compound than, for example the more hydrophobic PCBs, γ -HCH has a lower tendency to bio-accumulate in fish (Lockhart *et al.* 1992). The γ -HCH concentrations in biota from remote areas are typically lower than concentrations of PCBs or DDTs. DDT is an extremely toxic insecticide which is accumulated in biota and thereby magnified in the food chain. The use of DDT in western countries has been strongly reduced since the 1970's.

4.1.1.2 Heavy metals

Lead (Pb) is one of the heavy metal elements which is shown to be transported to and widespread within Arctic areas. There are only few regions which are unaffected by lead, as a result of long-range atmospheric transport (Jaworski *et al.* 1987). Earlier investigations have not revealed any large atmospheric point sources of lead in Norway (Rühling *et al.* 1987; Anderson 1989; SFT 1993). Surveys of heavy metals in mosses and lake sediments conclude

that lead deposition is still occurring, and that the main sources of deposited lead in Norway are long range transport (Rognerud & Fjeld 1993; Steinnes *et al.* 1994).

Depositions of cadmium (Cd) in the Arctic are mainly due to long-range atmospheric transport, but are also influenced by local emissions from smelter industry and by waste incineration of nickel-cadmium batteries. Studies of lake sediments show moderate enrichment in the central mountainous areas of southern Norway, as well as in inland parts of Northern Norway (Rognerud & Fjeld 1990). Regional moss investigations and precipitation studies have shown that long-range transport of cadmium has decreased from 1977 to 1985. The main sources of cadmium are mainly outside Norway (Steinnes *et al.* 1994).

Long distance atmospheric transport is suggested as being the primary source of mercury (Hg) in remote areas of Norway (Rognerud & Fjeld 1993). Hg content in air is mainly in gaseous form. Previous studies concluded that it is deposited along a decreasing gradient from the south to the north (Iverfelt & Rohde 1988). Investigations in Norway have shown a high degree of contamination in lakes in southern regions. A few lakes in Sør-Varanger, in Northern Norway, show moderate Hg contamination (Rognerud & Fjeld 1993). In Scandinavia, it has been demonstrated that most atmospheric input of Hg originates from the European continent (Iverfelt & Rohde 1988).

Depositions of Nickel (Ni) and copper (Cu) in Norway are generally of local origin (Steinnes 1994). Some lakes along the Norwegian-Russian border in Sør-Varanger are moderately to significantly contaminated with nickel. The main sources for this pollution are emissions from the smelter industry at the Kola-peninsula (Rognerud & Fjeld 1993, Steinnes *et al.* 1994).

The deposition of the metals antimony (Sb) and arsenic (As) in Norway mainly occurs by means of long range atmospheric transport. Depositions from local emissions of both Sb and As are found in the County of Nordland. The Russian smelters contribute to depositions of As in the eastern parts of Finnmark County (Steinnes *et al.* 1994).

Few data are available in Norway concerning the sources and deposition of the element titanium (Ti). The depositions of zinc (Zn) in northern parts of Norway are mainly caused by long range atmospheric transport (Steinnes *et al.* 1994).

There is no evidence of depositions of iron (Fe) and aluminium (Al) due to long range transport. Studies of mosses show that elevated concentrations of these elements in specific areas can be explained by contributions from local soil materials (windblown dust) or release from soil due to acidification (Steinnes *et al.* 1994). In Nordland County, local Fe and Al deposition occurs due to smelter activity. In Sør-Varanger (East-Finnmark) mining activities contribute to some of the depositions (Steinnes *et al.* 1994).

Research has shown that the increase of heavy metals such as lead, cadmium and mercury in lakes in Scandinavia is a result of long-range atmospheric transport of pollutants from sources in and outside Scandinavia. Depositions follow a decreasing gradient from the south to the north (Rühling & Tyler 1973; Iverfelt & Rohde 1988; Steinnes *et al.* 1989; 1994).

4.1.2 Background concentrations of POPs

Most persistent organic pollutants did not exist in the biosphere prior to their industrial synthesis, which began about 50-70 years ago. Industrially produced organochlorine compounds, such as PCBs and chlorinated pesticides (HCHs, chlordanes, DDTs, etc.) have no natural

background concentrations. None of the compounds analysed in this study has a natural occurrence.

However, some organohalogens do have natural sources. Terrestrial ecosystems mainly produce chlorinated organic compounds, while brominated organic substances are dominant in the aquatic environment. Algae, as well as bacteria and fungi, produce compounds such as halogenated hydrocarbons, phenols, ketones, terpenes and C15 compounds (Kvernheim *et al.* 1992). The recorded amounts of organohalogens (measured as adsorbable organically bound halogen, AOX) in runoff from bogs was high, comparable with heavily polluted rivers in Europe (Asplund *et al.* 1989). It is therefore difficult to estimate true background levels and very little effort has been made to do this (Kvernheim *et al.* 1992).

Polycyclic aromatic hydrocarbons (PAHs) are mainly formed by incomplete combustion of fossil fuels, wood and tobacco, as well as by processes such as incineration of garbage, steel and coal production and gasification. However, some PAHs can be formed naturally from transformation of organic material by micro-organisms. One PAH with high natural background levels is perylene, which can be formed during breakdown of humic materials. Therefore, in this investigation on contamination levels, perylene is excluded from the calculated sum PAH.

4.1.3 Natural background values of heavy metals.

Concentrations of heavy metals in lake sediments are largely related to the natural geochemical content in the catchment area, in addition to deposition from anthropogenic sources. It is therefore important to obtain data on natural background values and variations in heavy metal content in the catchment areas. The geochemistry of the northern part of Norway has been mapped by using overbank sediments as indicators of the geochemistry of the catchment areas. These inorganic sediments, deposited on the river banks during periods of flooding, are considered to be reliable indicators of the geochemistry of the catchment areas (Ottesen *et al.* 1989). This material has been found to be sufficiently representative of the natural geochemical conditions, and the values represent 200-1000 year-old depositions (Ottesen *et al.* 1989). Analyses were conducted on the sediment fraction less than 0.062 mm, which is similar to the dominant particle size in lake sediments in the deposition areas (Håkanson & Jansson 1983). For this study, data for Se, Ni, Pb and Zn were provided by Tore Volden, Norwegian Geological Survey, Trondheim. Regression analysis between concentrations in reference sediments and river bank sediments from 22 localities in same area show acceptable correlation for Se and nickel (respectively -0.52, 0.69), while there was only poor correlation for lead and zinc ($r = 0.25$, $r = 0.11$). The poor correlation for lead and zinc may be due to high content of organic matter in the NGU data set (input by Sigurd Rognerud). Other explanations are the high degree of local variation in geochemical composition in the area, as well as the relatively large distance between the sampling stations in this study and those of NGU. In other studies where the relationship between river bank sediments and lake sediments has been specifically tested, a significant correlation was also found for lead (Rognerud & Fjeld 1993).

4.2 Transport from catchment area to lake

Heavy metals may enter lakes by direct deposition on the water surface, as well as by run-off/drainage from the catchment area. Lake sediments may comprise depositions from natural sources in the catchment area, as well as atmospheric depositions on both the lake surface and the catchment area. The part which is incorporated into lake sediments depends on the characteristics of both the metals and the catchment area.

The transport of heavy metals from the catchment area to a lake usually follows one of two different patterns (Bergkvist *et al.* 1989).

- Transport related to organic acids, mainly humic acids. Acidification and release of minerals in the catchment area have little effect for these compounds. In our investigation, this transport mechanism is relevant for Pb, Hg, Cu, and Sb.
- Transport related to acidification and the subsequent release of the mineral fraction of the catchment area. This transport mechanism may be of relevance for the heavy metals Cd, Ni, Al, and Zn in the eastern part of Finnmark County (Sør-Varanger), which receives contamination from the smelter industry on the Kola Peninsula.

Persistent organic pollutants, such as PCB, PAH, and organochlorine pesticides follow a different and more complicated transport- and deposition pattern, compared to most heavy metals. The most important property which distinguishes them from most metals are that most components are volatile and can evaporate to the atmosphere.

After deposition on the catchment area, most persistent organic pollutants exist, under normal conditions, in one of the following phases: vapour phase, adsorbed to atmospheric particles, adsorbed on the soil surface, adsorbed to particles, associated with humic matter or dissolved in the water phase. The mechanisms for transport and movement between the phases are dependent, amongst others, on the distribution between these reservoirs (Barrie *et al.* 1992).

Gas exchange is dependent on chemical reactions with the soil surface, which includes the solubility of the compounds and reaction with water, as well as absorption mechanisms with the soil, water and ice (Barrie *et al.* 1992).

Substances in the environment will therefore be divided between the particulate and the gas phase. With the exception of PCB-components, which have the capacity to evaporate from the snow when the temperature rises in the spring and summer, the majority of persistent organic pollutants is transported from the catchment area to the lake under snow-melt in the spring/summer and it is demonstrated that most compounds are adsorbed to humic substances (Barrie *et al.* 1992).

Both PAH and organochlorine compounds which reach the lakes can be taken up by organisms. Through natural processes they can deposit in the lake and accumulate in the bottom sediment.

4.3 Geology and geochemistry

Northern Norway is underlain by the northern part of the Precambrian Baltic shield bordered by the Caledonian fold belt to the west. The Precambrian basement is divided into gneisses and supracrustal belts intruded by gabbro, anorthosite, and granite in the northeast. The Caledonides are a series of allochthonous thrust sheets or nappes overthrust to the east. The area has been repeatedly glaciated by continental ice sheets during the Pleistocene. The valleys and

areas with moderate relief are covered by till or glacio-fluvial gravels and sands. Bedrock is well exposed in the west and at higher elevations (Eden and Bjørklund 1994)

The lakes on Svalbard and the one on Bear Island are located mainly on sedimentary rocks (sandstones, shales, and calcareous rocks of Carboniferous, Triassic, Jurassic, and Cretaceous age) or their metamorphic equivalents.

4.4 Vegetation and soil in catchment areas

Atmospheric depositions of most persistent organic pollutants as well as lead, mercury, copper, and antimony are strongly associated with humic substances, and adsorb readily to humic soil particles. Acidification of the catchment areas does not influence the transport essentially as long as it does not alter the decomposition rate of the humic layer (Bergkvist *et al.* 1989). The extent and the thickness of the humic layer in the catchment area determine the capacity of the catchment area to retain these substances. In this study, the proportion of bog and forest areas in all catchment areas were mapped. Generally, the forest area of the catchment areas was highest in the County of Nordland (average 41%). In Finnmark County, the forest area reached an average of 26% of the catchment areas. The lowest percentage of forest in the catchment areas was registered in Troms County (13%). The cover of bogs and mires in the lake catchment areas show opposite trends. In Troms County the bog and mire areas were in average 13% of the catchment areas. In Finnmark and Nordland Counties the areas were respectively 7.5 and 6.3%.

4.5 Processes in lakes

Both persistent organic pollutants and heavy metals exist in particulate, colloidal and dissolved phases in lake water. The majority of pollutants are adsorbed to particles which, in their turn, accumulate in lakes through sedimentation processes.

Heavy metals such as Pb and Hg and persistent organic pollutants are usually bound to organic matter or organic depositions (Förstner 1982; Barrie *et al.* 1992).

The rate of sediment accumulation in Scandinavian lakes in wooded areas is usually between 0.7-1.7 mm/year (Rognerud & Fjeld 1993). Age determinations in lake sediments in the northern part of Norway show that the sedimentation rate for sediment 5 cm or more below the sediment surface is usually less than 1 mm/year (Norton 1986; Rognerud *et al.* 1993). Based on this, it is likely that the top sediment (0-1 cm) has been deposited after the 1980's, while the reference sediment (sediment depth 19-20 cm) represents the pre-industrial era, over 200 years ago.

It has been demonstrated that persistent organic pollutants and heavy metals differ in their mobility and transformation capacities (chemical and biological) in sediment (Hart 1982; Förstner 1982; Carigan & Tessier 1985; Andersson & Borg 1988; Barrie *et al.* 1992). This implies that concentrations of a specific substance at a specific sediment depth are not necessarily directly related to depositions during certain time-periods. The mobility varies with the physical and chemical characteristics of the specific compound, as well as with the physical, chemical and biological processes in the lake water and sediments. These effects should be evaluated for each specific element or compound.

5. Results of sediment investigations

5.1 General

In Norway, several studies of contaminants in sediments have been carried out over the last decade, both on a national and regional basis. Rognerud and Fjeld (1990) carried out a national survey of heavy metals in lake sediments and mercury in fish in the late eighties. Skotvold and Rognerud (1993) carried out a study of heavy metals and POPs in lake sediments in Northern Norway and Spitsbergen. Sediments from lakes on the border area between Norway and Russia have been analysed for heavy metals (Rognerud *et al.* 1993). However, there are only limited data available on contaminants in lake sediments in the areas between the southern and northern part of the Arctic region of the Norwegian mainland, and on the Norwegian Arctic islands.

This study aims to augment the existing database on contaminants in the Arctic regions of Norway, and includes heavy metals and persistent organic pollutants in sediments from a total of 60 lakes in Northern Norway, Spitsbergen and Bear Island. In addition, data material on heavy metals in lake sediments from earlier investigations (Rognerud *et al.* 1993) are included, giving a total number of 92 investigated lakes. The lakes and their primary data are presented in Appendix 1.

5.2 Organic carbon content

Figure 2 below shows the median levels and 85% confidence intervals for organic content (OC) of the cores. The OC levels in lake sediments are significantly lower on the Arctic islands Spitsbergen and Bear Island (2-5%) compared with the lakes on the Northern Norwegian mainland (4-12%). On the Norwegian mainland, the OC content in cores from lakes in Nordland and Finnmark Counties are generally higher than those from lakes in Troms County.

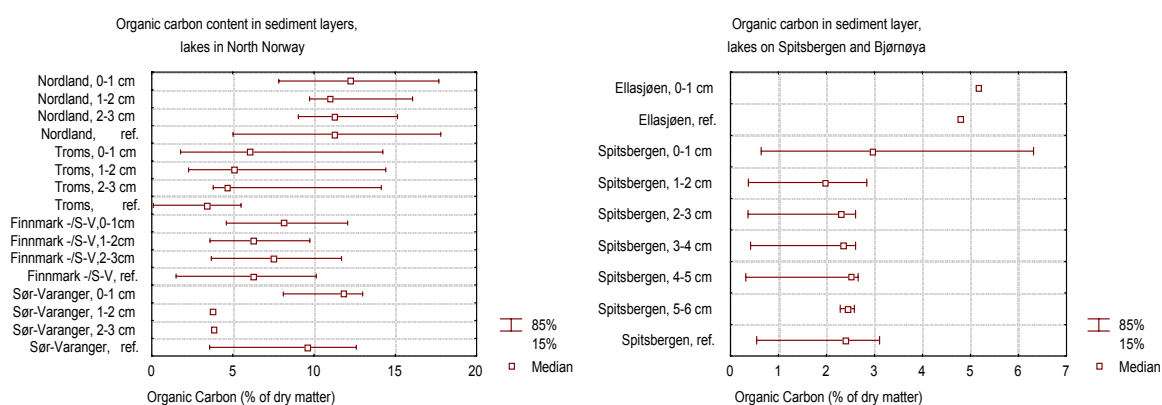


Figure 2. Median levels of organic content (% OC in dry matter) in sediments from lakes on the Northern Norwegian mainland and Norwegian Arctic islands.

5.3 Metal concentrations in sediment

5.3.1 Background concentrations of heavy metals

It would have been expedient to carry out sediment dating on all cores collected in the study, but this goal was unrealistic due to the limited resources available. By carrying out the selective sampling strategy of using cores only from the accumulation zone in the lakes (Håkanson & Jansson 1983), we feel confident that our reference samples are old enough to represent background values, as has been demonstrated in other studies (Rognerud & Fjeld 1993). Therefore, in this investigation reference sediments were defined as being representative of the natural background values in the lakes. Reference sediments were collected from 20-30 cm depth, and are considered to represent the pre-industrial time period, approximately 200 years ago.

5.3.2 Enrichment levels and concentrations of heavy metals

5.3.2.1 Mainland Northern Norway

Figure 3 and Figure 4 show mean concentrations for all the elements studied. For the elements Hg and Se, the data on mean absolute concentrations from the Finnmark area cannot be compared directly with those from the Nordland and Troms area, due to the different analytical methods used. Data from the Finnmark area are, however, comparable to the national study carried out, as the same analytical methods were used (Rognerud & Fjeld 1990).

For the elements which can be compared between the different surveys, the content in the upper sediment layer for most heavy metals studied in the region are generally lower than those found in the southern parts of Norway (except for Ni concentrations). For methodologically comparable elements such as Pb and Cd, Rognerud & Fjeld (1990) found mean concentrations which ranged between 125 - 225 mg/kg dry weight and 0.850 - 0.900 mg/kg dry weight respectively in the south-eastern parts of Norway. In this study, the median concentrations in the upper sediment layer ranged between 17-85 mg/kg dry weight and 0.450 - 0.750 mg/kg dry weight respectively (Figure 3). Concentrations of Ni show an increase with latitude. Rognerud & Fjeld (1990) showed the mean concentrations in the south-eastern part of Norway to be 15 - 20 mg/kg dry weight. In this study, the median concentrations of Ni in the top sediment were found to be from 25 - 58 mg/kg dry weight (Figure 3). For elements such as Al, Cu, Fe, Sb, Zn, Ti and Se, no analyses have been carried out in national surveys with which to compare the results of this study.

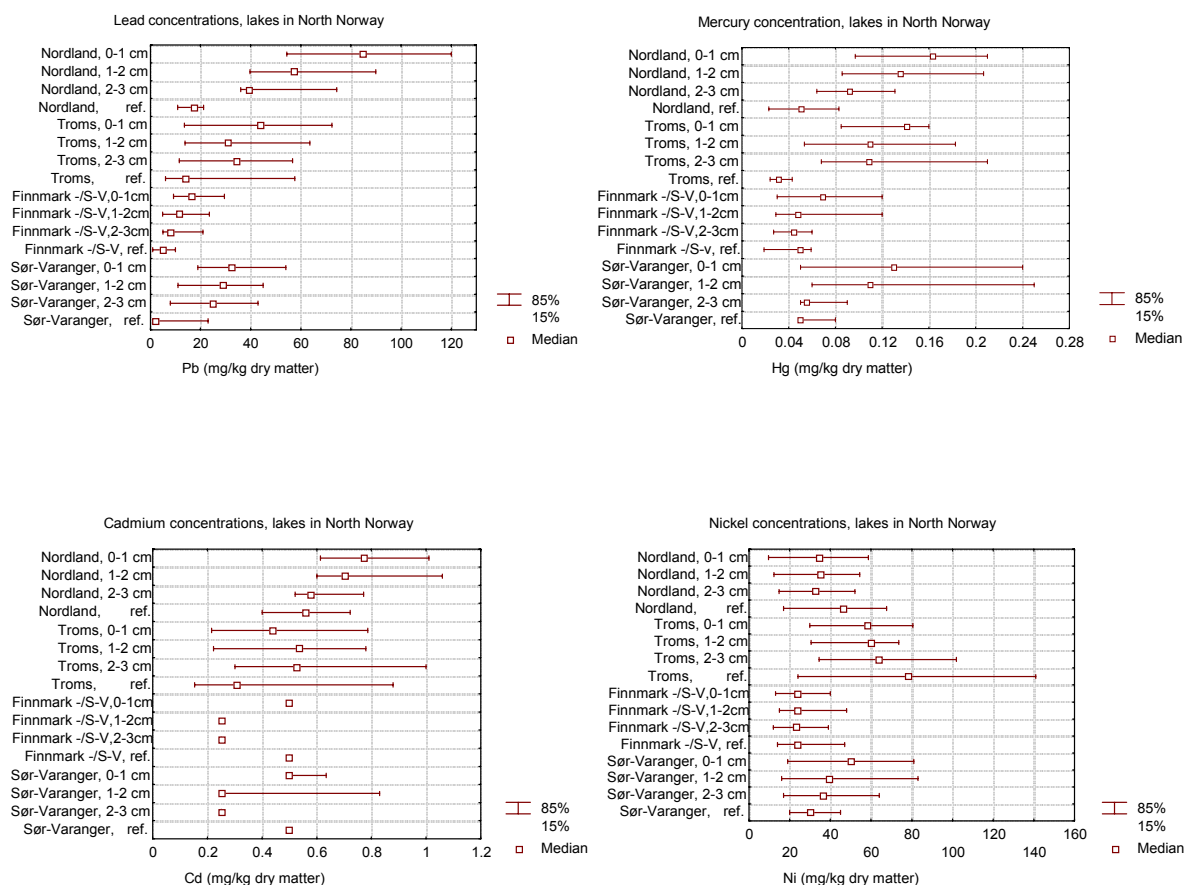


Figure 3. Median concentrations of Hg, Pb, Cd and Ni (mg/kg dry weight) in lake sediments in Northern Norway.

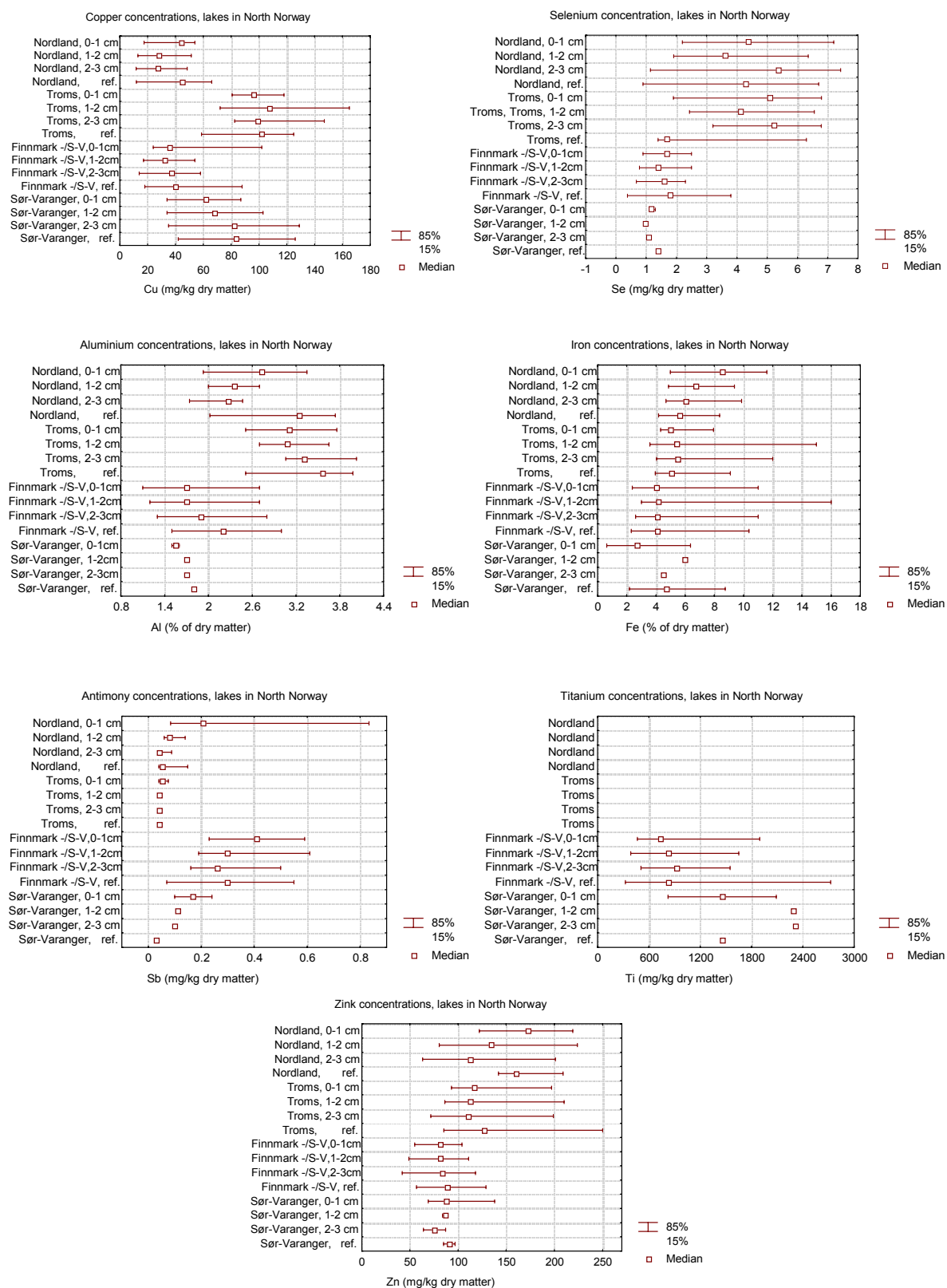


Figure 4. Median concentrations of Cu, Se, Al, Fe, Sb, Ti, and Zn (mg/kg dry weight) in lake sediments in Northern Norway.

Figure 5 shows the median enrichment factors (K_f) for metals, based on values from all lakes from the different areas. It is evident that the upper sediment layers in lakes on the Arctic parts of the Norwegian mainland are enriched with the heavy metals Hg, Pb, Sb, and Cd. The central areas appear to be less enriched compared to the southern and northern parts of the study area. However, such conclusions cannot be drawn for the other elements in this study (Figure 6). The K_f factors for Hg and Pb show an increase from the 2 - 3 cm layer to the 0 - 1 cm layer indicating that deposition of the metals has occurred in the region over the past decades. The other metals in this study do not show such a general trend.

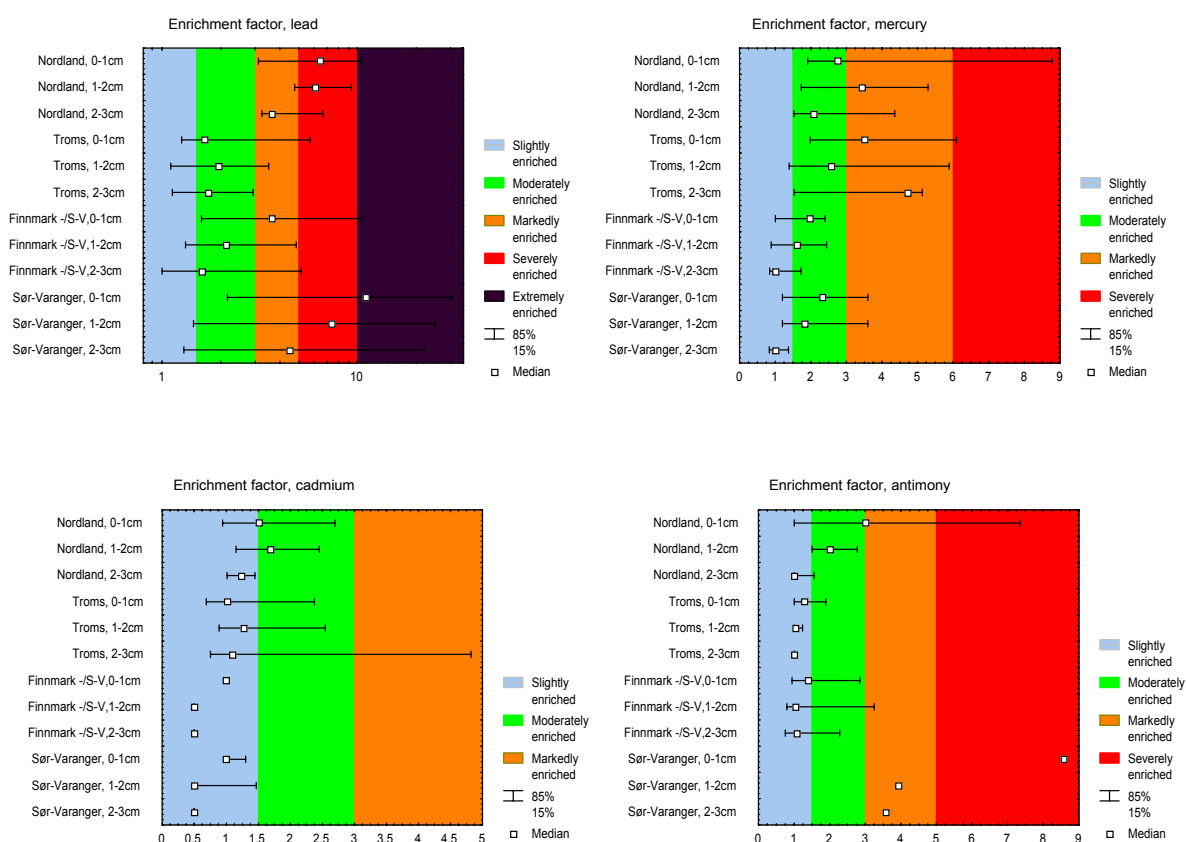


Figure 5. Median enrichment factors for Pb, Hg, Cd and Sb in lakes of Northern Norway.

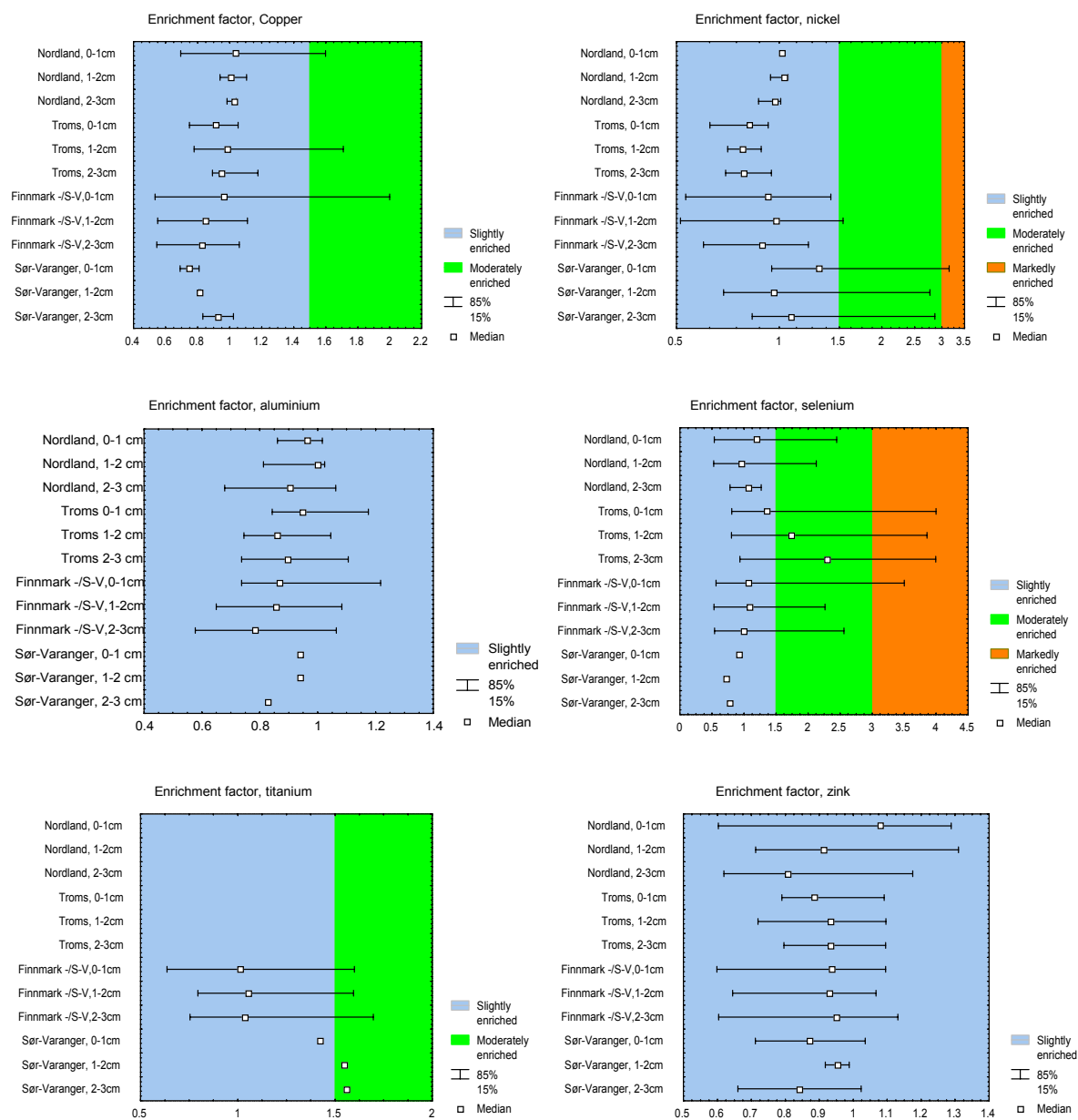


Figure 6. Median enrichment factors for Cu, Ni, Al, Se, Ti and Zn in lakes of Northern Norway.

5.3.2.2 Norwegian Arctic islands

The average concentrations of most heavy metals in the upper layers of lake sediment studied on the Arctic islands are generally lower than those found on the Arctic mainland. For Spitsbergen and Bear Island the median concentrations of Hg were about 0.06 and 0.1 mg/kg, for Pb 25 and 90 mg/kg and for Cd 0.5 and 1.3 mg/kg respectively (Figure 7). On the mainland, the median concentrations of these metals ranged between 0.075 - 0.210 mg/kg, 17 - 85 mg/kg and 0.45 - 0.750 mg/kg respectively (Figure 3). In the lakes on Spitsbergen and Bear Island, the median Ni concentrations were found to be 33 and 25 mg/kg respectively (Figure 8). The median concentrations of Ni in the upper sedimentary layers on the mainland were found to be 25 - 58 mg/kg.

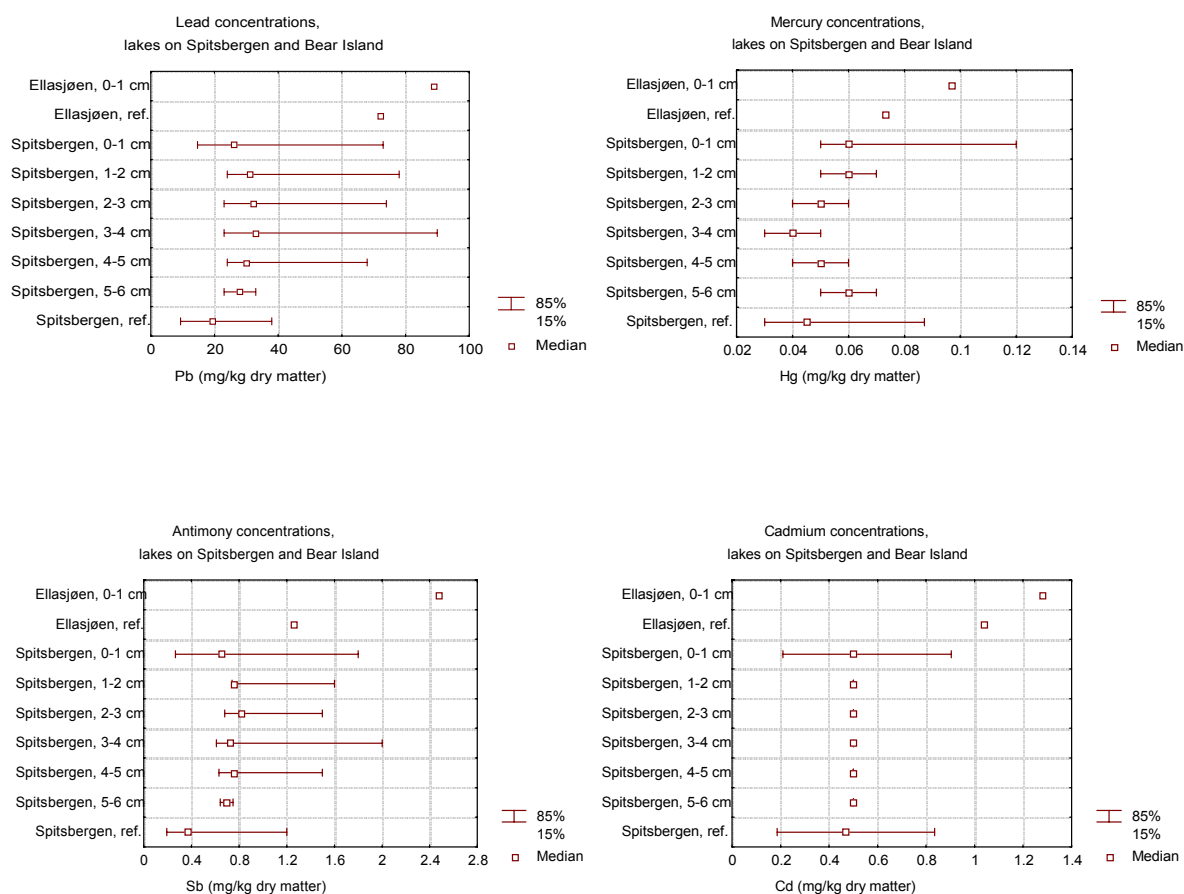


Figure 7. Mean concentrations of Pb, Hg, Sb and Cd (mg/kg dry weight) in lake sediments on the Norwegian Arctic islands.

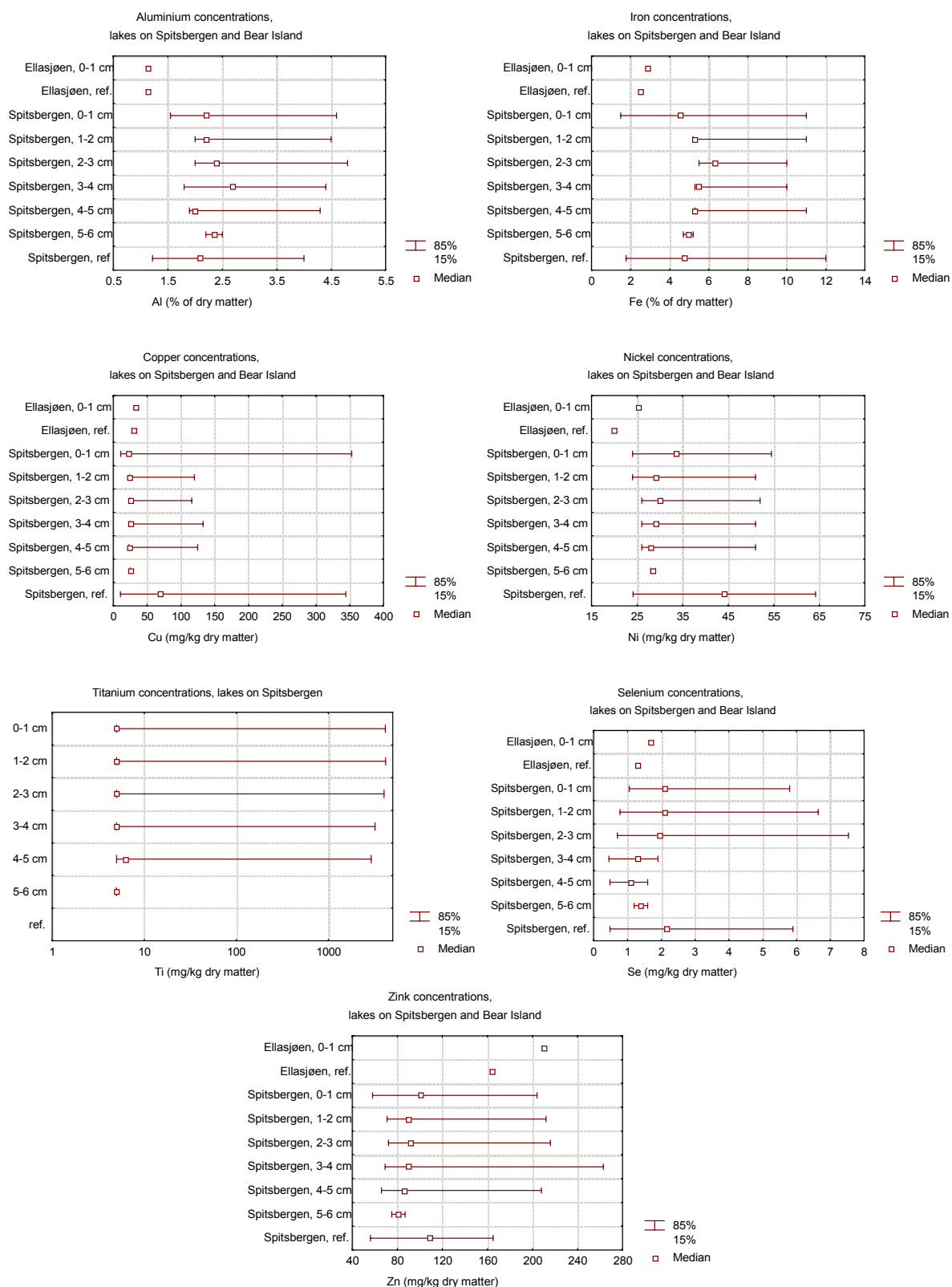


Figure 8. Mean concentrations of Fe, Al, Cu, Ni, Ti, Se and Zn (% of dry matter or mg/kg dry weight) in lake sediments on the Norwegian Arctic islands.

Figure 9 shows median enrichment factors (K_f), for metals, based on values from all lakes sampled in the different areas on Spitsbergen and Bear Island. The figure shows the upper sediment layers in lakes on Spitsbergen to be enriched by the heavy metals Hg, Pb, Se, Ti and Zn. On Bear Island antimone (Sb) is the only element which is enriched in the upper sediment layer.

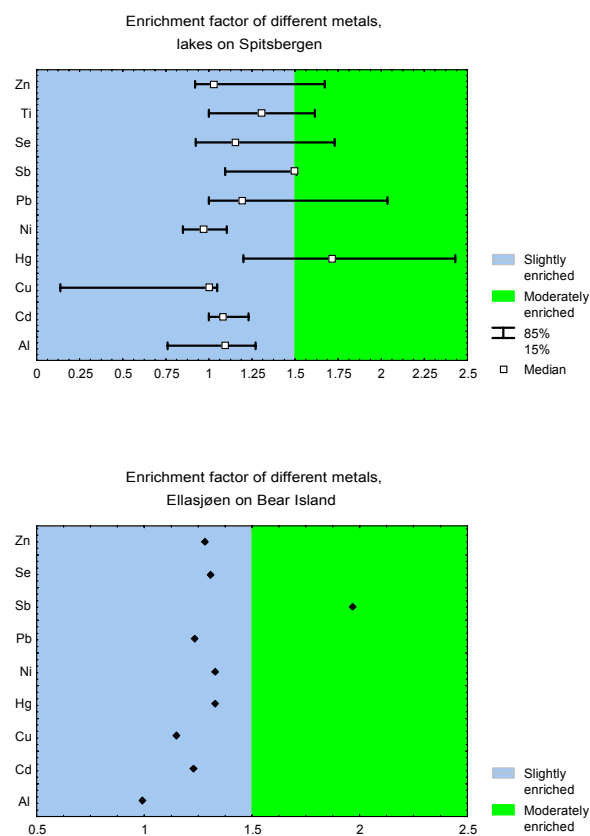


Figure 9. Enrichment factors (K_f) for Zn, Ti, Se, Sb, Pb, Ni, Hg, Cu, Cd, and Al in lakes on the Norwegian Arctic islands.

5.3.3 Geographical distribution of heavy metals in sediment

Figure 10 to Figure 30 present the geographical distribution of concentrations and enrichment factors of heavy metals in surface sediments in lakes on the mainland of Northern Norway and the Norwegian Arctic islands, Spitsbergen and Bear Island. Both concentrations and enrichment factors are presented.

Evaluation of the figures reveals that for a series of compounds, the concentrations were highest along the coasts of Nordland and Troms Counties. This is the case for the following elements: Al, Cd, Hg, Pb, Sb, Se, and Zn. Steinnes *et al.* (1994) also found elevated levels of Pb, Sb and Se in mosses along the coast. For Hg, higher levels were also determined in mosses along the northern coast of Finnmarken County (Steinnes *et al.* 1995). A significant input of Se in Norway originates from the marine environment (Steinnes *et al.* 1994). Levels of Pb, Sb, Cd, As, Zn and Hg in mosses were found to be mainly determined by long-range atmospheric transport (Berg *et al.* 1995). This coastal influenced distribution pattern will be discussed more thoroughly in a following publication (Rognerud *et al.* in prep.).

The distribution patterns of concentrations of Hg and Se should be interpreted with care, as sediments from Nordland and Troms were analysed using different methods than the other sediments.

Some of the elements had elevated concentrations in Sør-Varanger: Hg (moderately to strongly affected), Ni (moderately to strongly affected), Pb (moderately to very severely affected). Not all metals were analysed in this area. This contamination can be related to the local emission sources on the Kola Peninsula, mainly the nickel industry (Steinnes *et al.* 1994; Rognerud *et al.* 1993)

The distribution of mercury in sediment does not appear to be related to the distribution pattern in mosses. Concentrations of Hg in mosses doubled from the inland of Troms and Finnmark Counties towards the northern coast of Finnmark (Steinnes & Andersson 1991; Steinnes *et al.* 1995). A possible explanation presented for this increase in Hg in the terrestrial environment towards the north, is the increasing significance of dry deposition of Hg⁰, under colder conditions where re-emission to the atmospheric is reduced. Our results of Hg in the sediments do not show any significant gradient from the inland towards the coast. The highest concentrations in surface sediment were found along the coasts of Nordland and Troms Counties and in Sør-Varanger (0.2 - 0.45 µg/g, compared to concentrations mainly below 0.1 µg/g in Finnmark County). Some lakes in these areas also had elevated enrichment factors for Hg (strongly to severely affected). The higher levels in lake sediments along the coast appear to be related to higher precipitation, relative to inland areas.

Enrichment factors for Pb were high in many lakes on the Lofoten-Vesterålen and Senja Islands, as well as in the municipalities of Kautokeino and Sør-Varanger.

Two lakes had elevated enrichment factors for many metals, these being Lakes Ø. Kaperdalsvatn (68) at Senja and Ryggedalsvatn (75) in Vesterålen. These lakes were moderately to very severely affected by Al, Cd, Cu, Fe, Hg, Pb, Sb, Se and Zn. In the top sediment, most metal concentrations were only slightly higher than in surrounding lakes. Low concentrations of most metals in the reference sediment layer lead to high enrichment factors (calculated from the ratio between the concentration in the top layer and the concentration in the reference layer). An example is the extremely low reference concentration of Zn in Ryggedalsvatn. Only the concentrations of Cu in the top sediments of both lakes were significantly higher than in surrounding lakes, while the concentrations of Cu in the reference sediments were

low. Concentrations of Cd, Hg, Pb, Sb and Se in the top sediment of Ø. Kaperdalvatn were in the upper range of concentrations in surrounding lakes. The organic carbon content in the top sediment of Ø. Kaperdalvatn was relatively high and, together with the relatively low pH of the water, this may account for the elevated levels in the sediment compared to the low concentrations in the low-organic reference sediment. Other reasons such as local sources are as yet unknown.

Lake 3, Storvatn close to the town of Alta, also had elevated concentrations of heavy metals, such as Ni, Cu and Pb in the top sediment. This is most likely to have originating from previous mining activities in the catchment area of this lake.

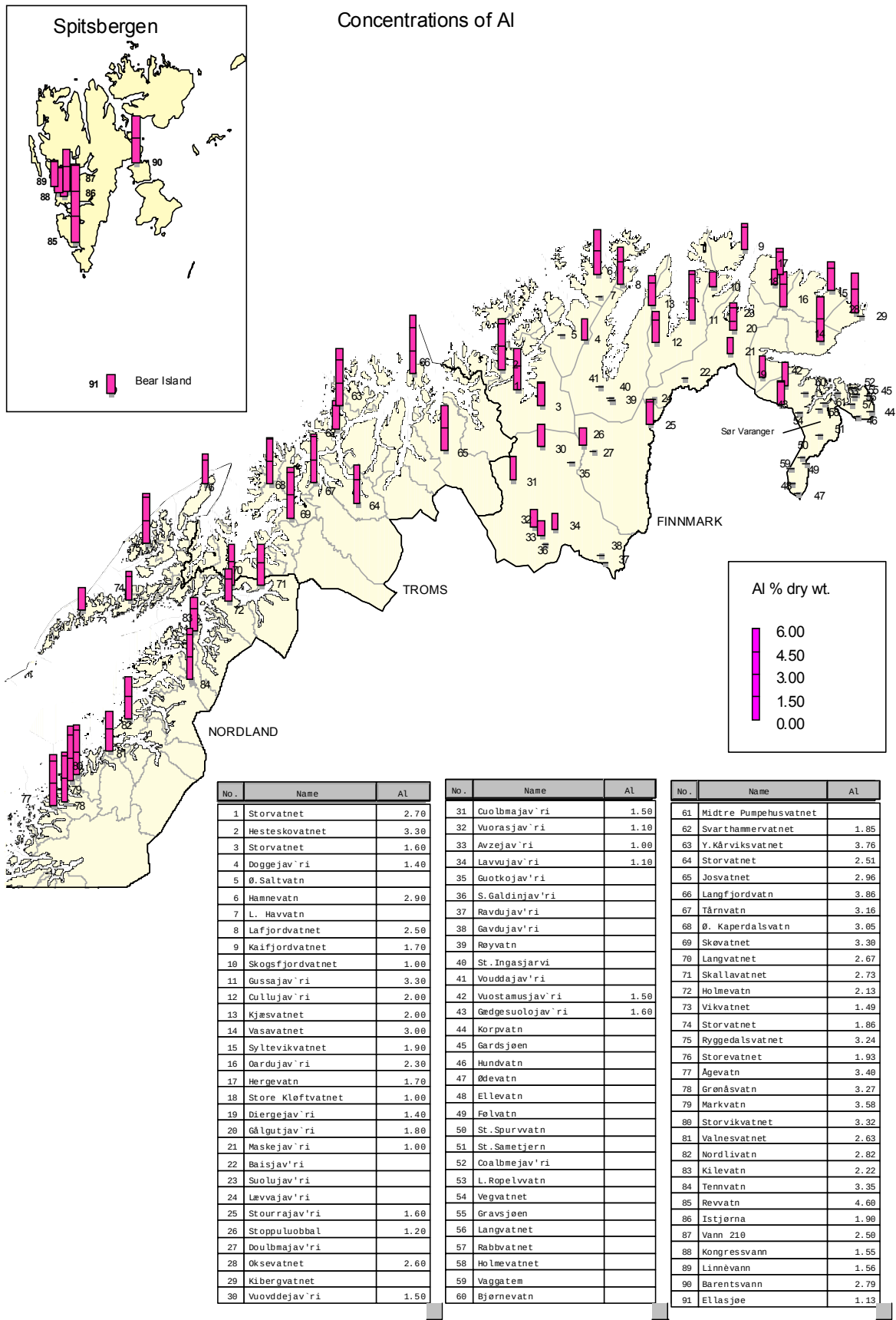


Figure 10. Concentrations of aluminium (% of dry sediment) in surface sediments from lakes on the Northern Norwegian mainland and Norwegian Arctic islands.

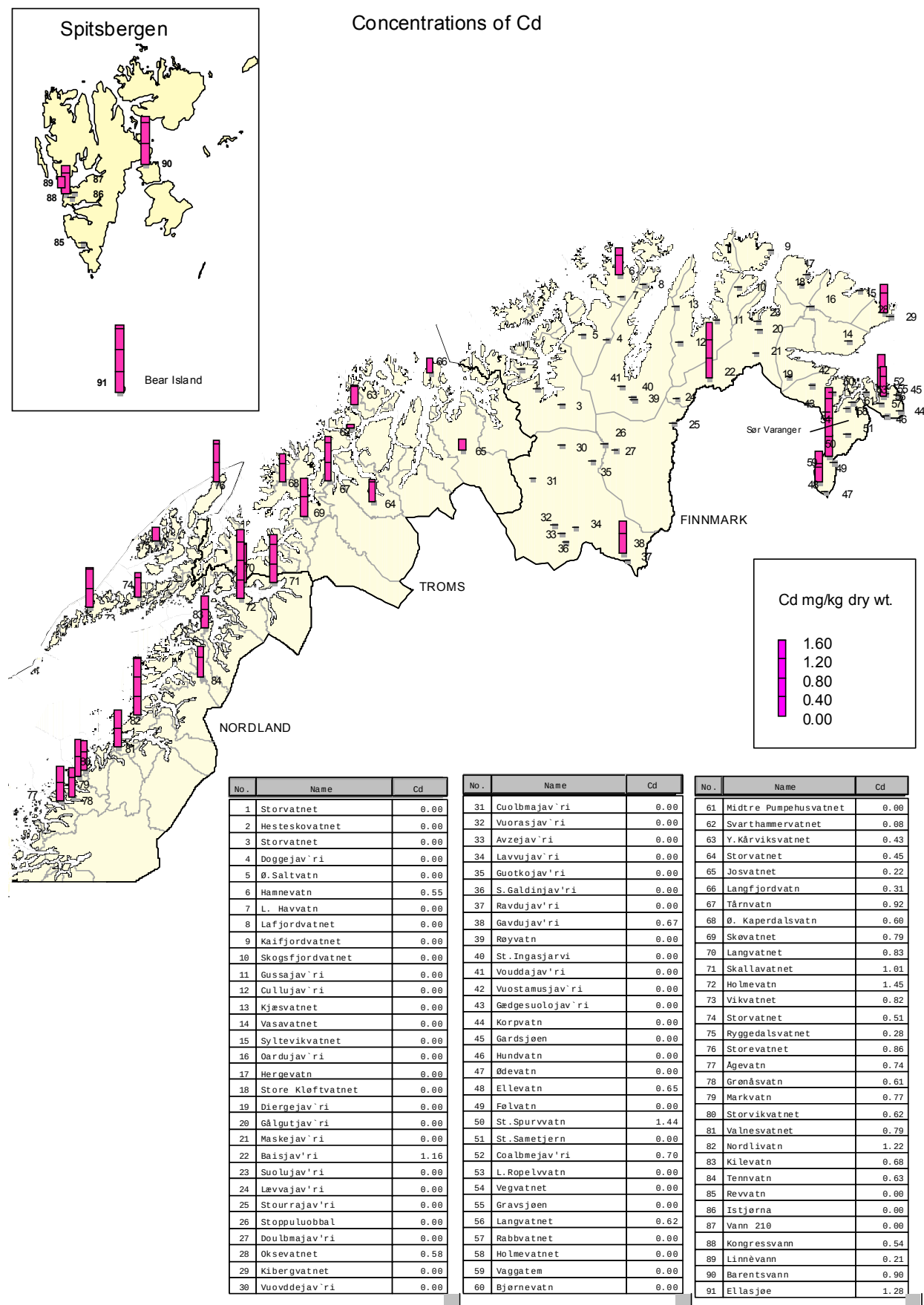


Figure 11. Concentrations of cadmium($\mu\text{g/g}$ dry sediment) in surface sediments from lakes on the Northern Norwegian mainland and Norwegian Arctic islands.

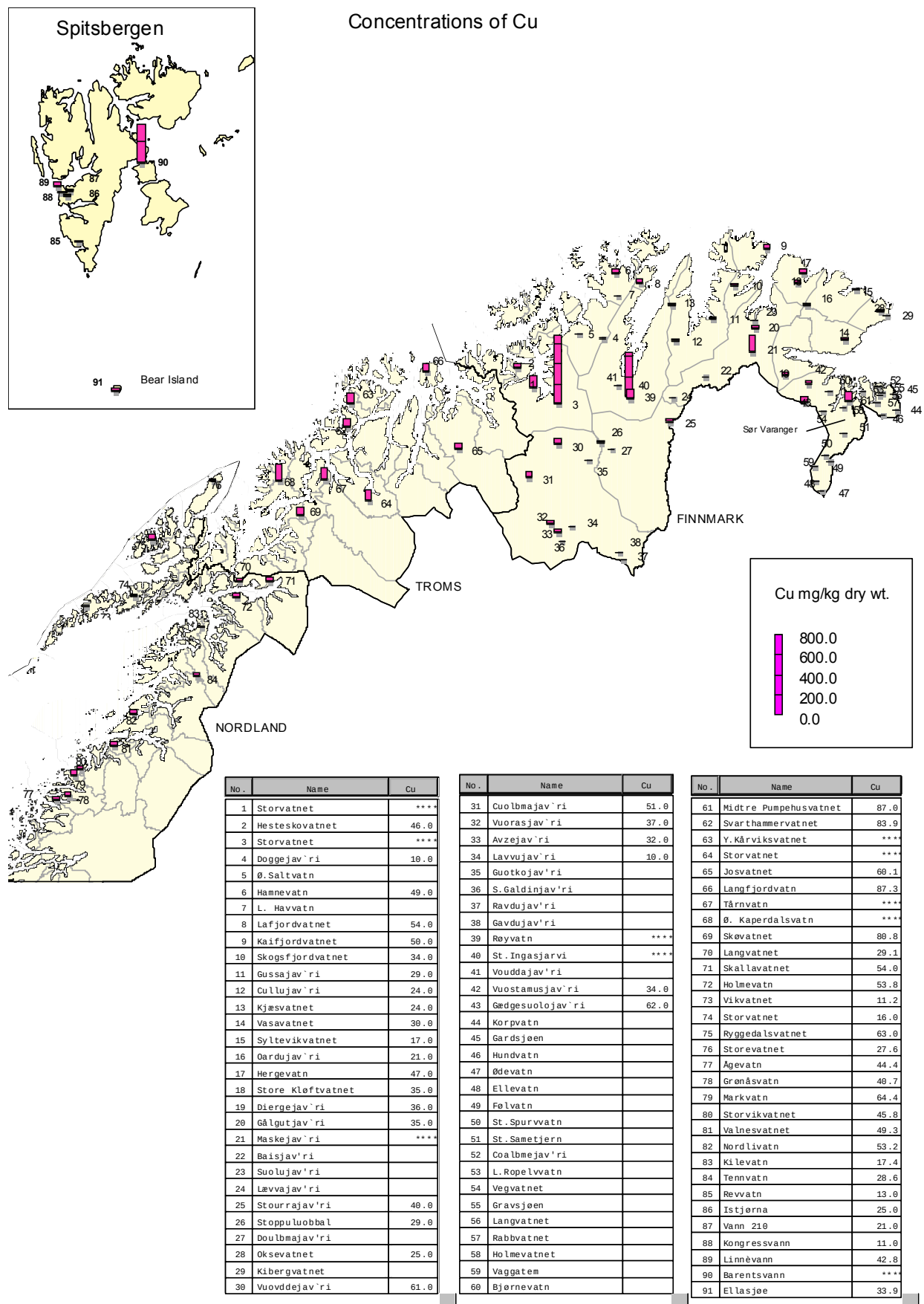


Figure 12. Concentrations of copper ($\mu\text{g/g}$ dry sediment) in surface sediments from lakes on the Northern Norwegian mainland and Norwegian Arctic islands.

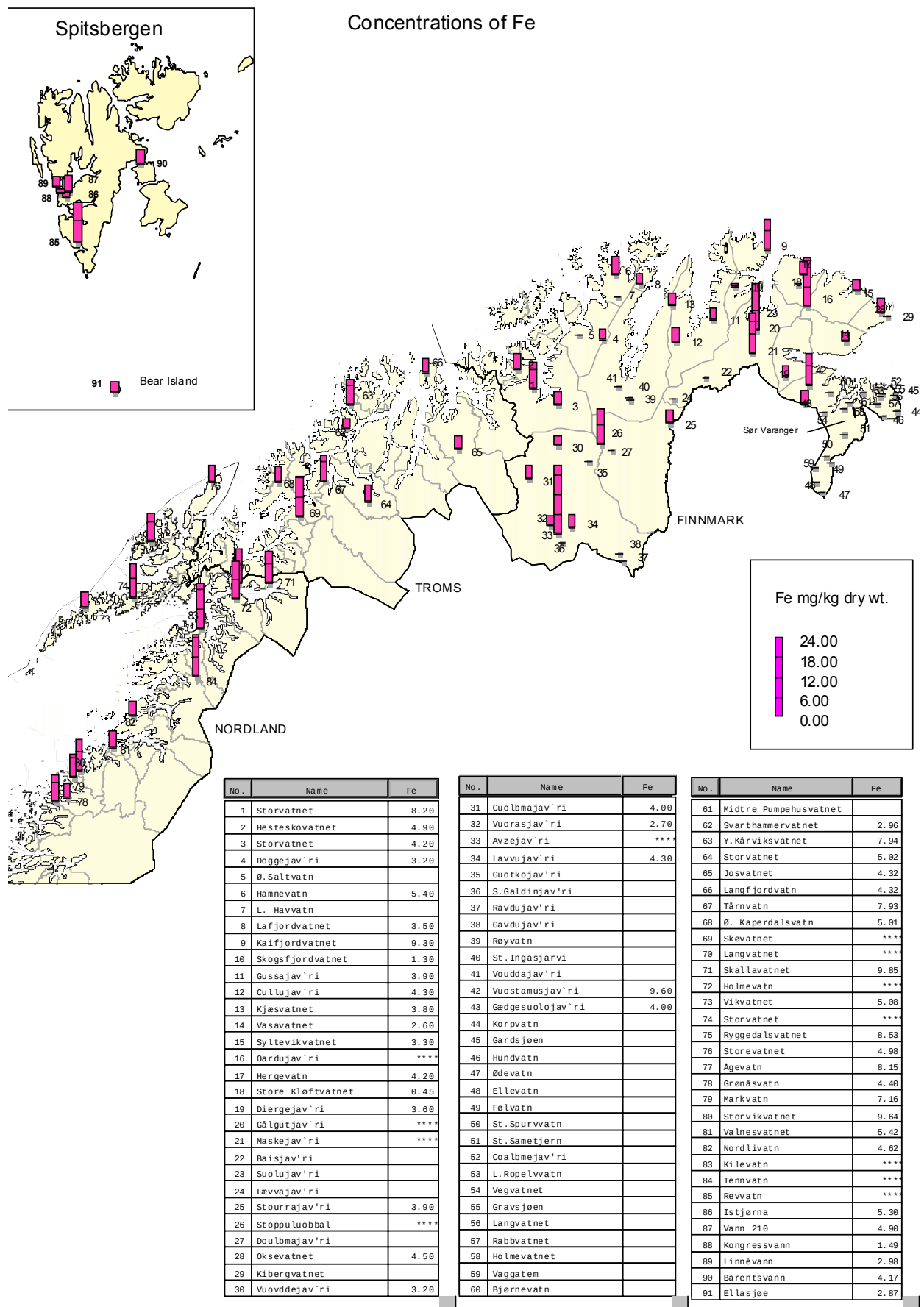


Figure 13. Concentrations of iron ($\mu\text{g/g}$ dry sediment) in surface sediments from lakes on the Northern Norwegian mainland and Norwegian islands.

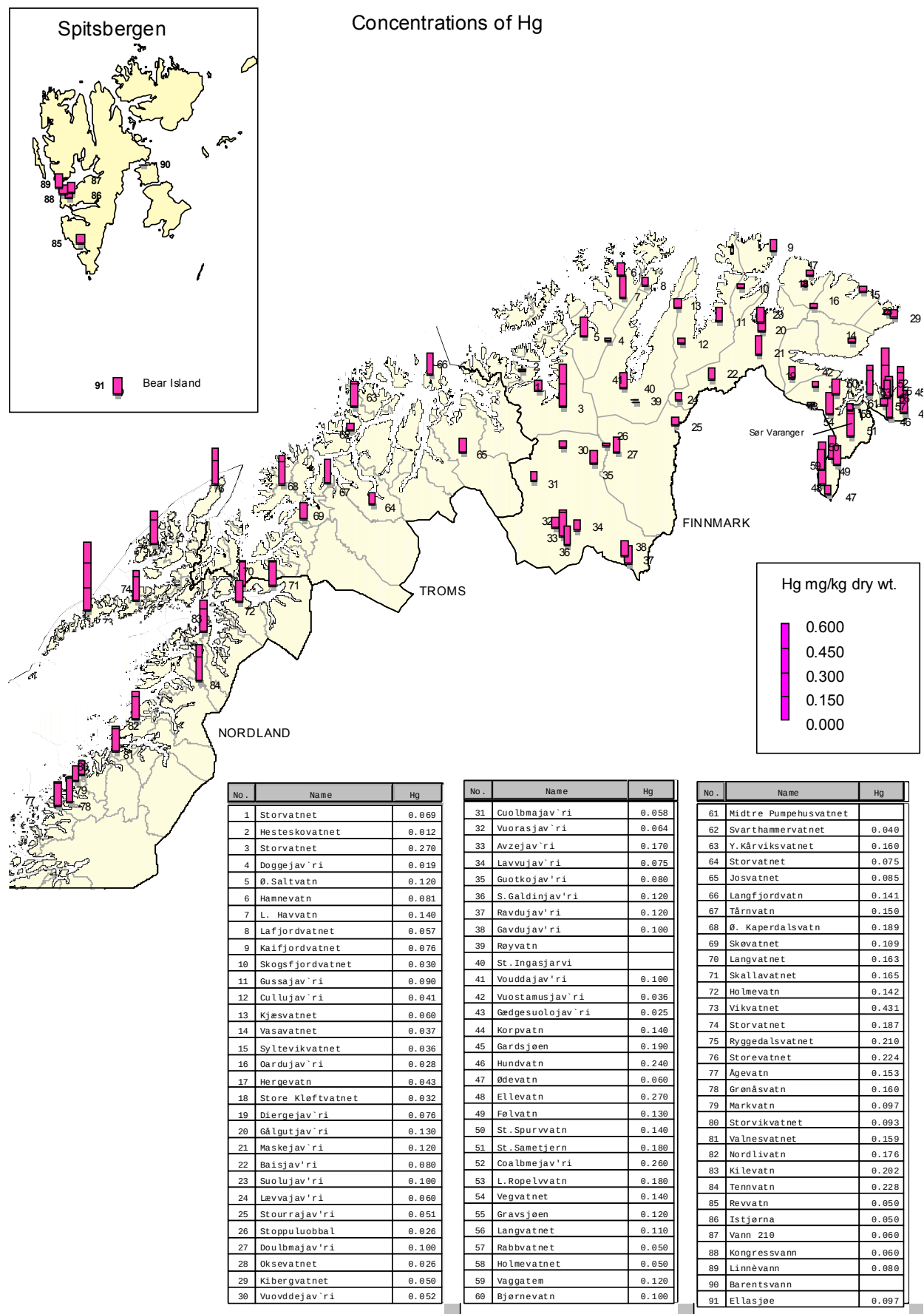


Figure 14. Concentrations of mercury ($\mu\text{g/g}$ dry sediment) in surface sediments from lakes on the Northern Norwegian mainland and Norwegian islands.

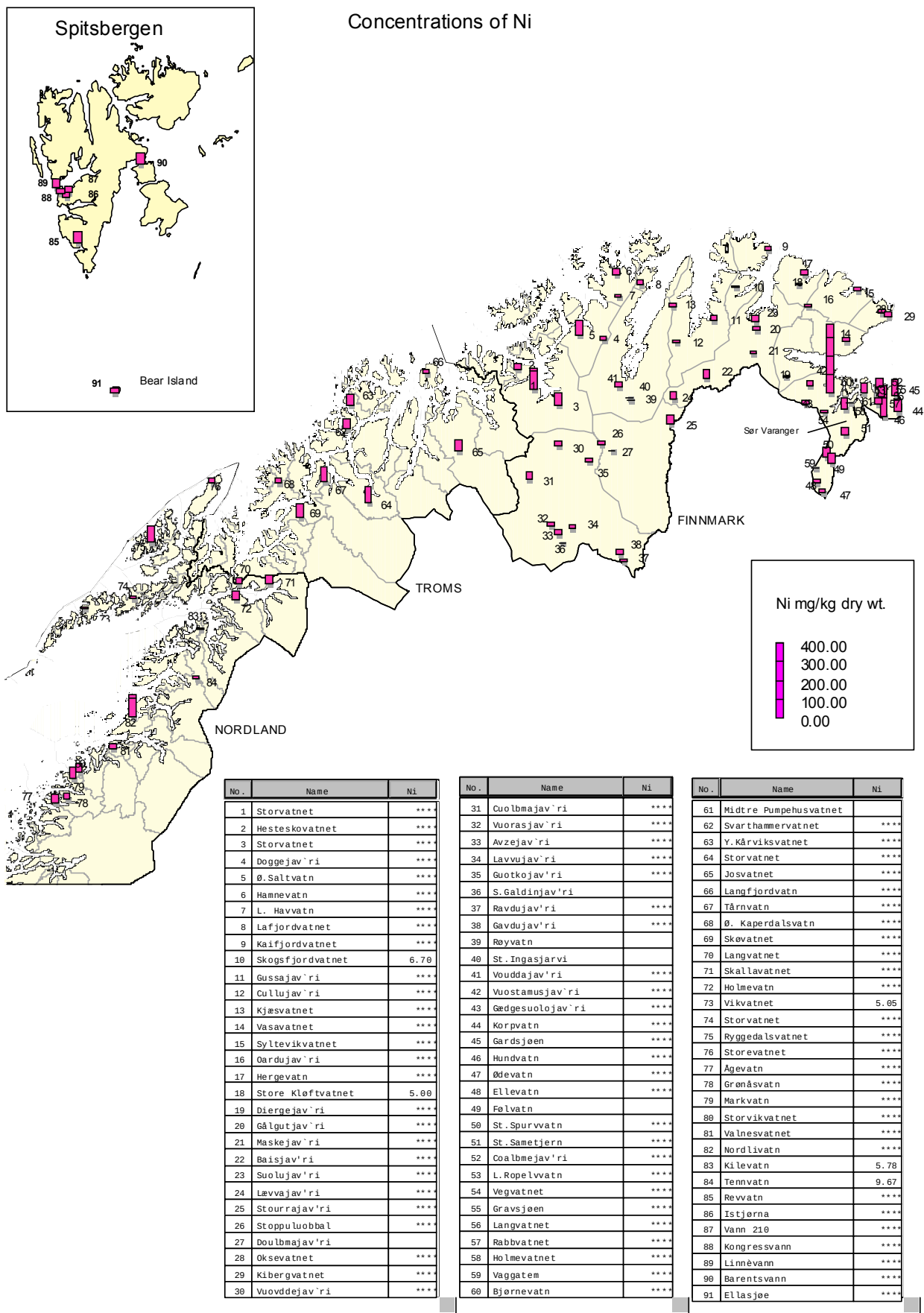


Figure 15. Concentrations of nickel ($\mu\text{g/g}$ dry sediment) in surface sediments from lakes on the Northern Norwegian mainland and Norwegian Arctic islands.

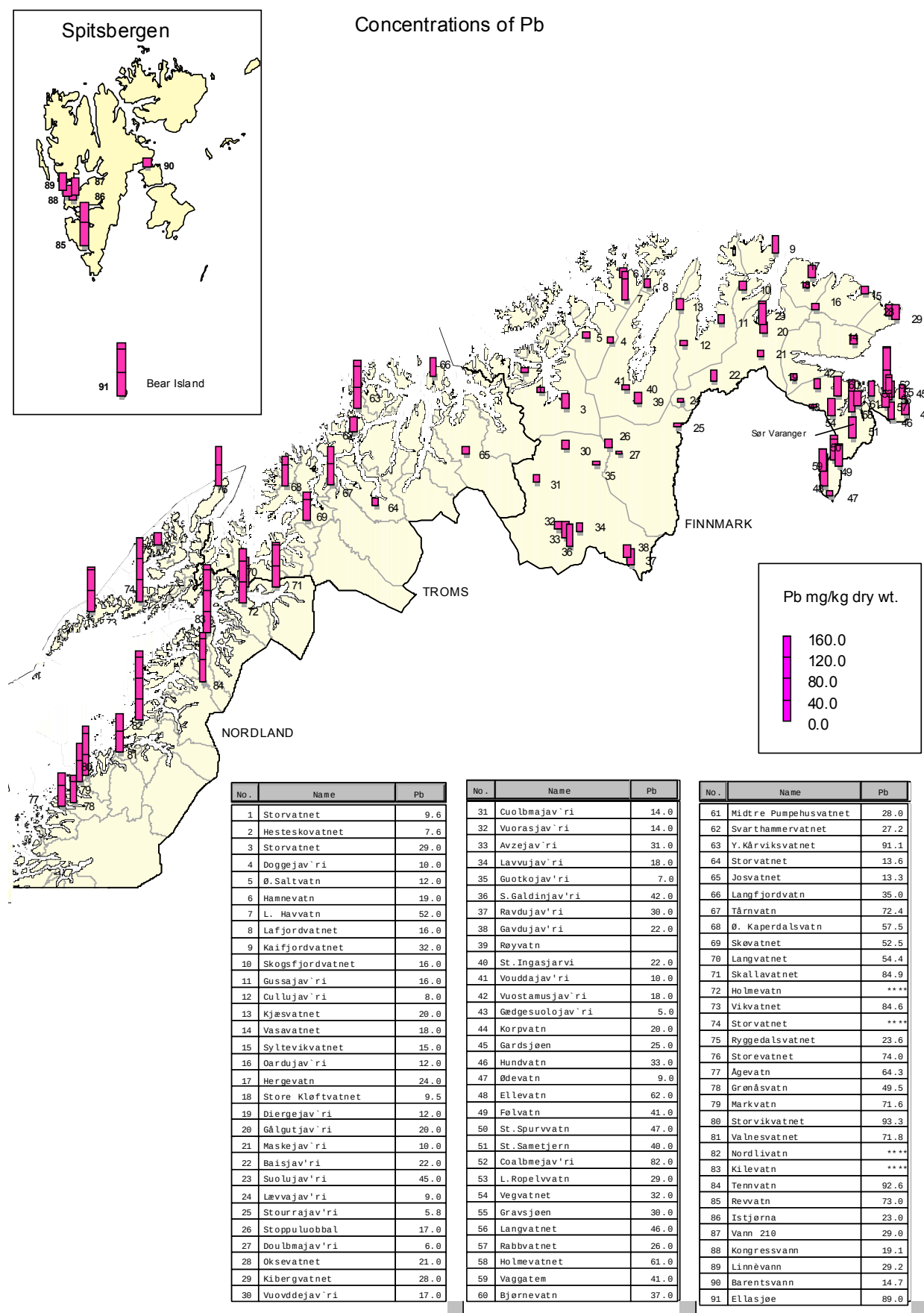


Figure 16. Concentrations of lead ($\mu\text{g/g}$ dry sediment) in surface sediments from lakes on the Northern Norwegian mainland and Norwegian Arctic islands.

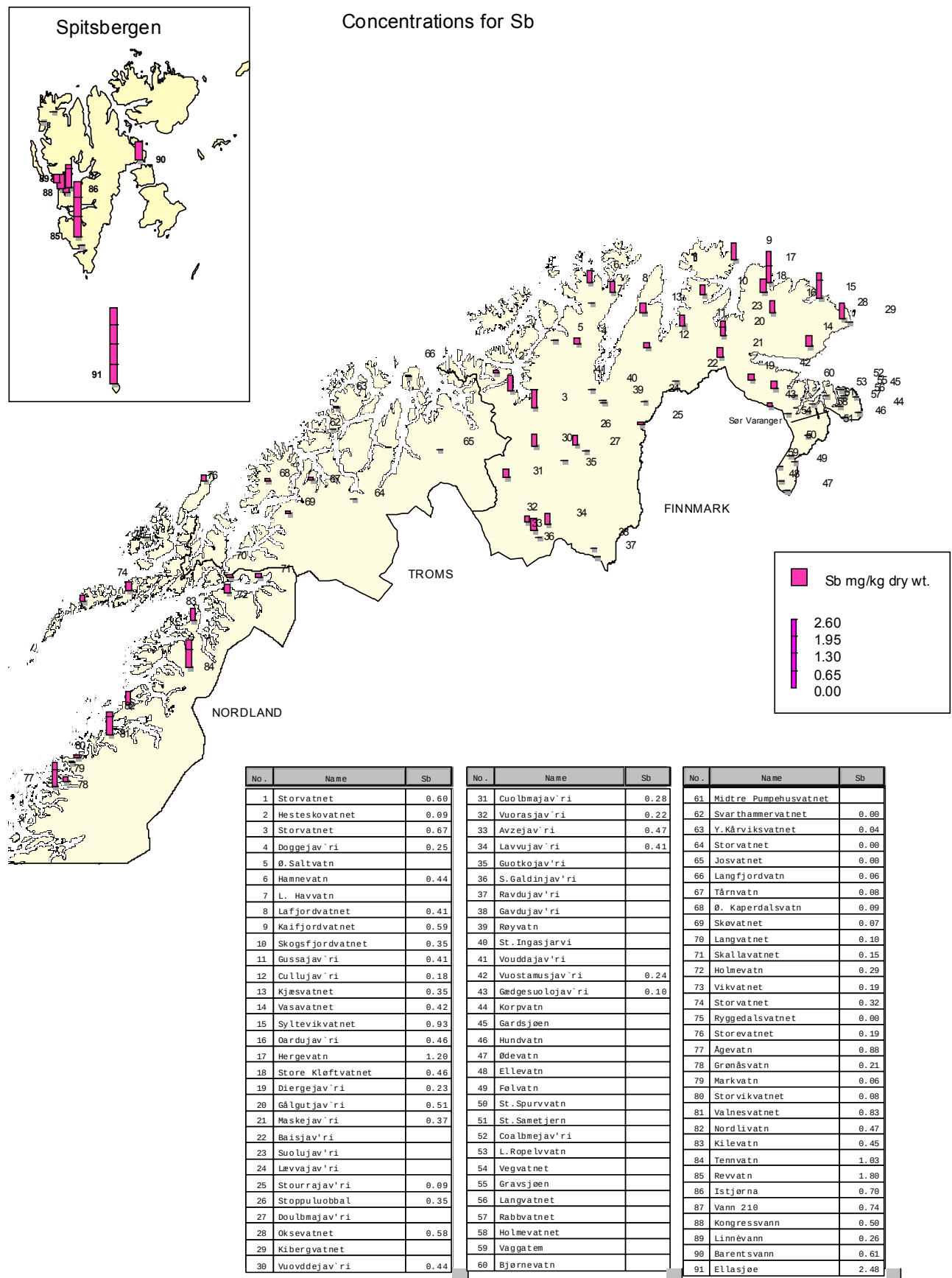


Figure 17. Concentrations of antimony ($\mu\text{g/g}$ dry sediment) in surface sediments from lakes on the Northern Norwegian mainland and Norwegian Arctic islands.

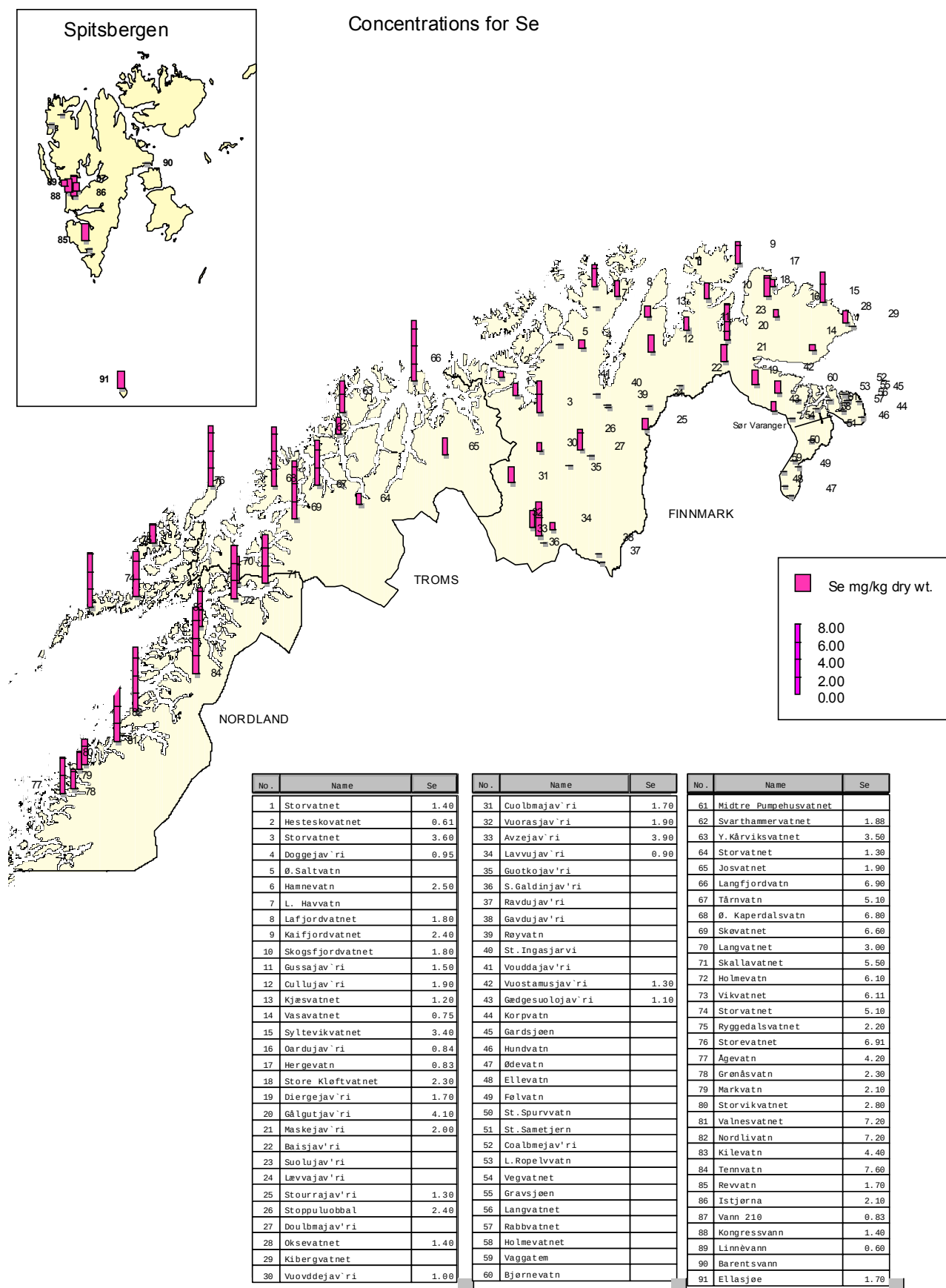


Figure 18. Concentrations of selenium ($\mu\text{g/g}$ dry sediment) in surface sediments from lakes on the Northern Norwegian mainland and Norwegian Arctic islands.

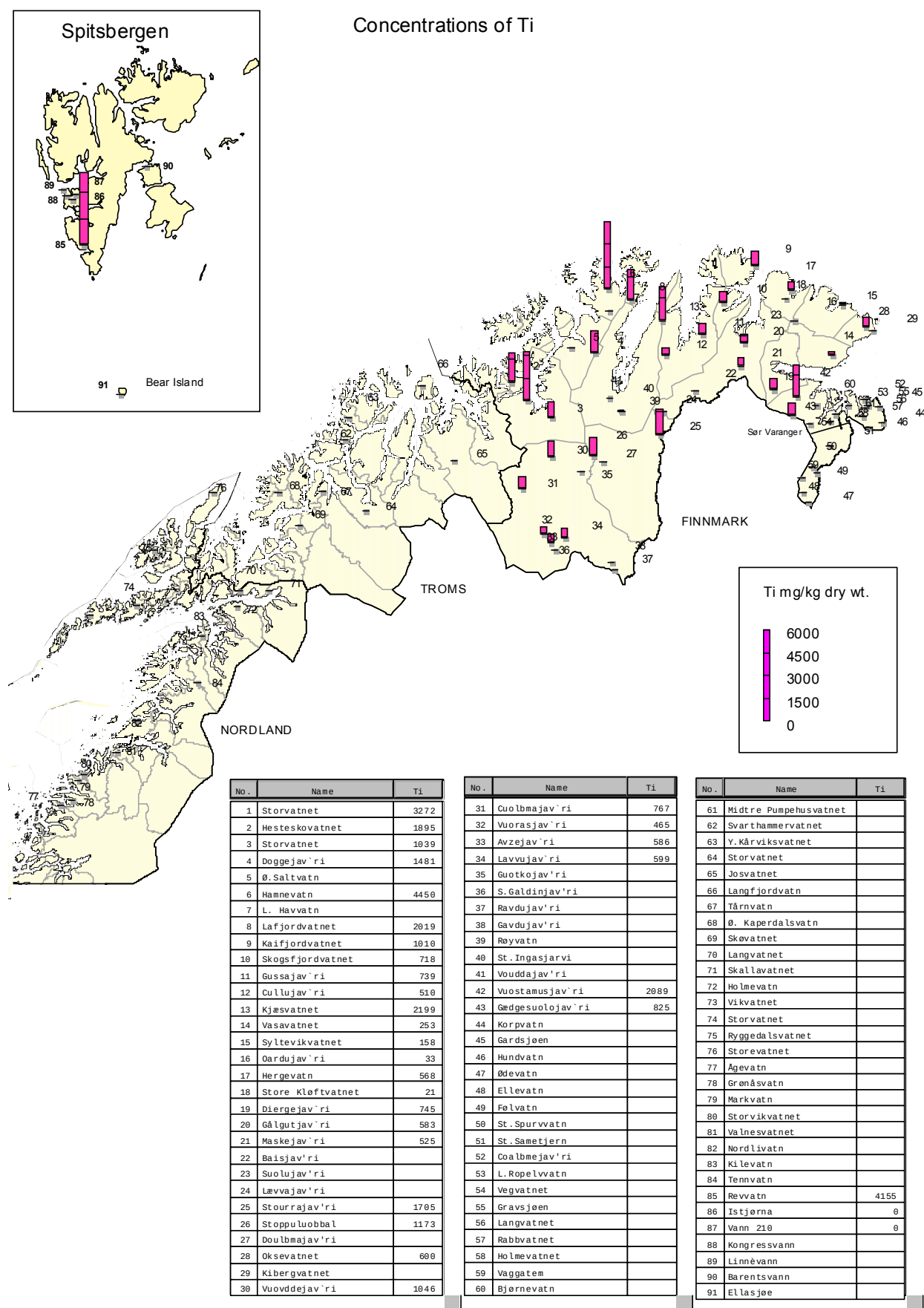


Figure 19. Concentrations of titane ($\mu\text{g/g}$ dry sediment) in surface sediments from lakes on the Northern Norwegian mainland and Norwegian Arctic islands.

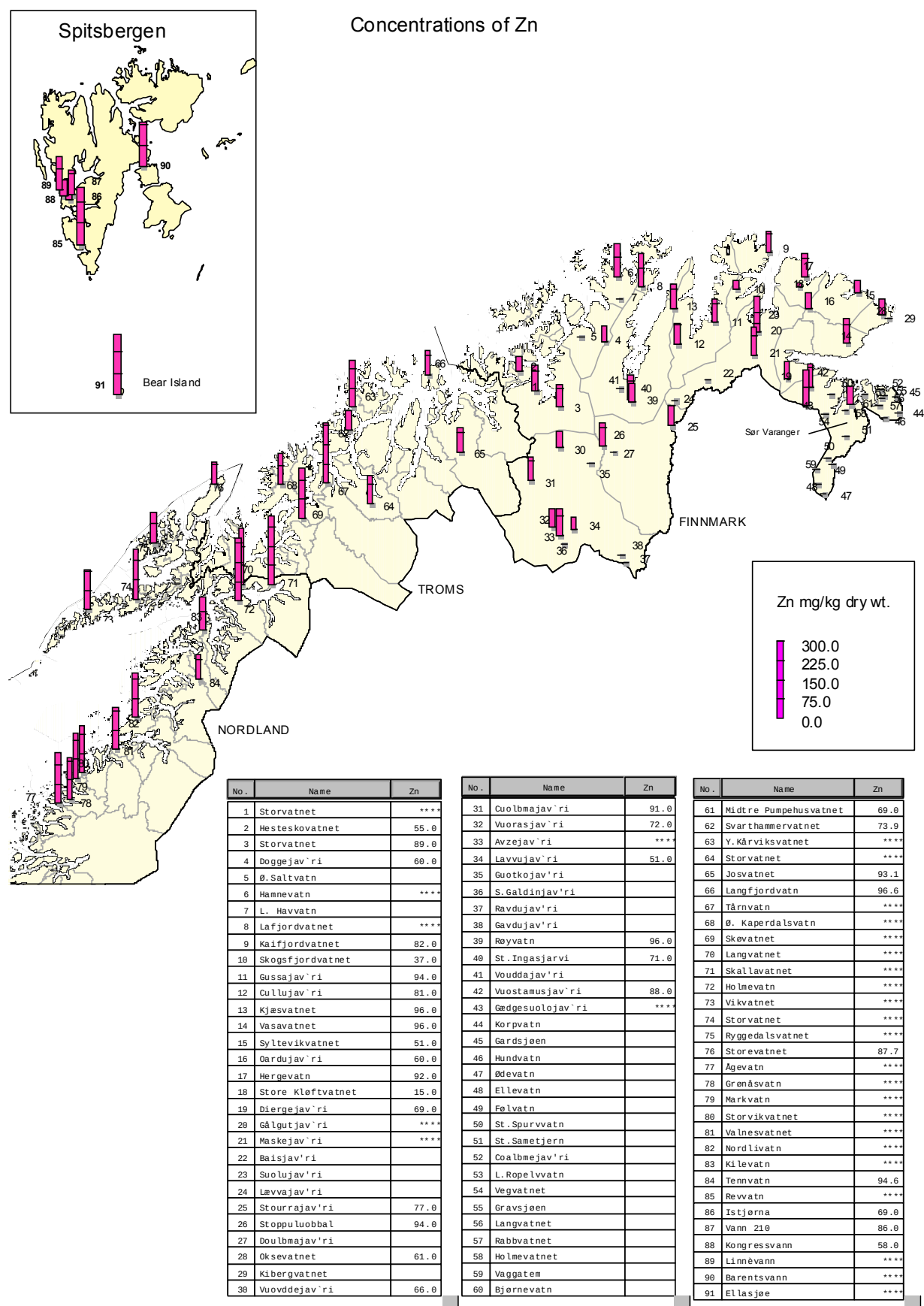


Figure 20. Concentrations of zinc ($\mu\text{g/g}$ dry sediment) in surface sediments from lakes on the Northern Norwegian mainland and Norwegian Arctic islands.

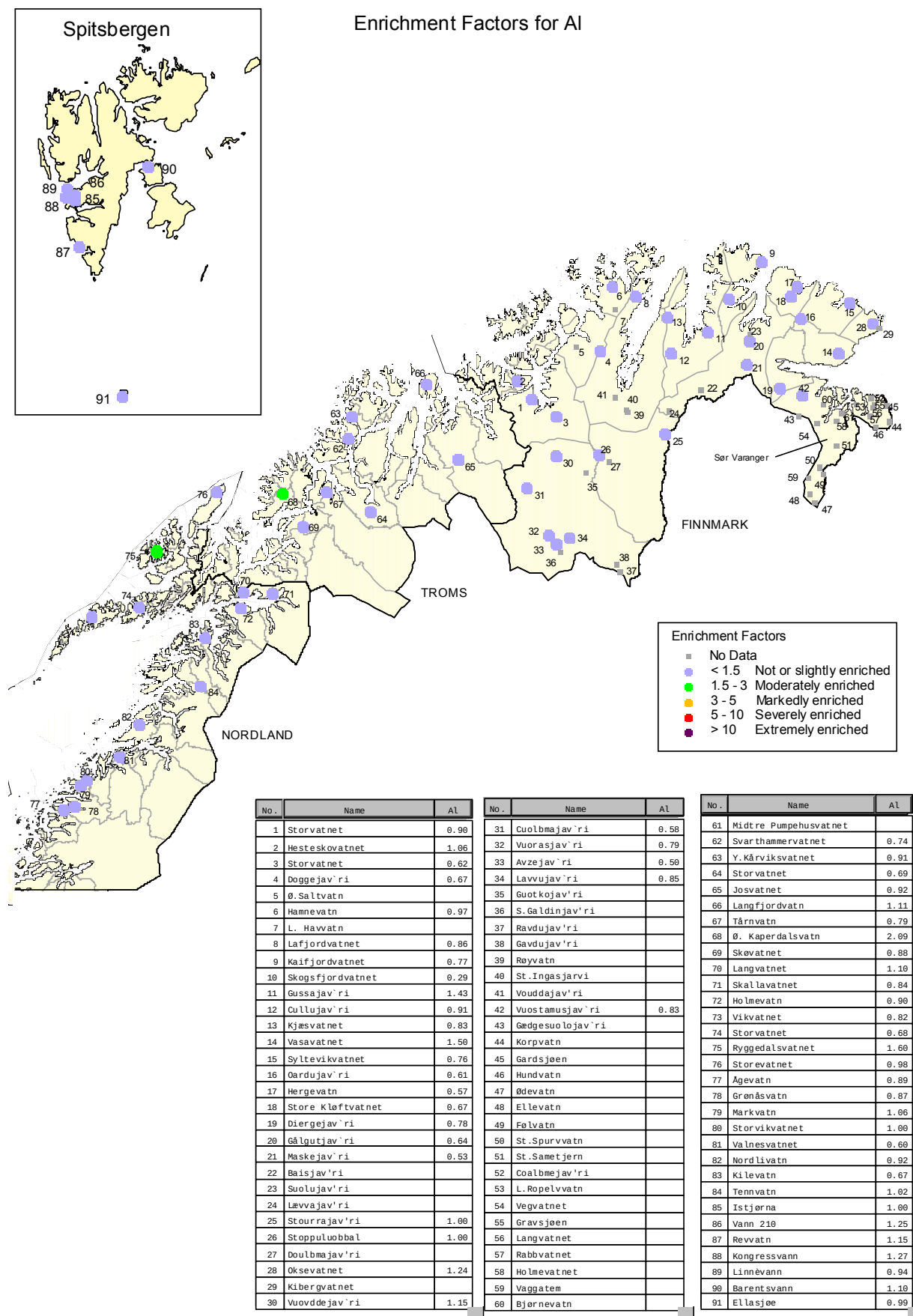


Figure 21. Enrichment factors for aluminium in surface sediments from lakes on the Northern Norwegian mainland and Norwegian Arctic islands.

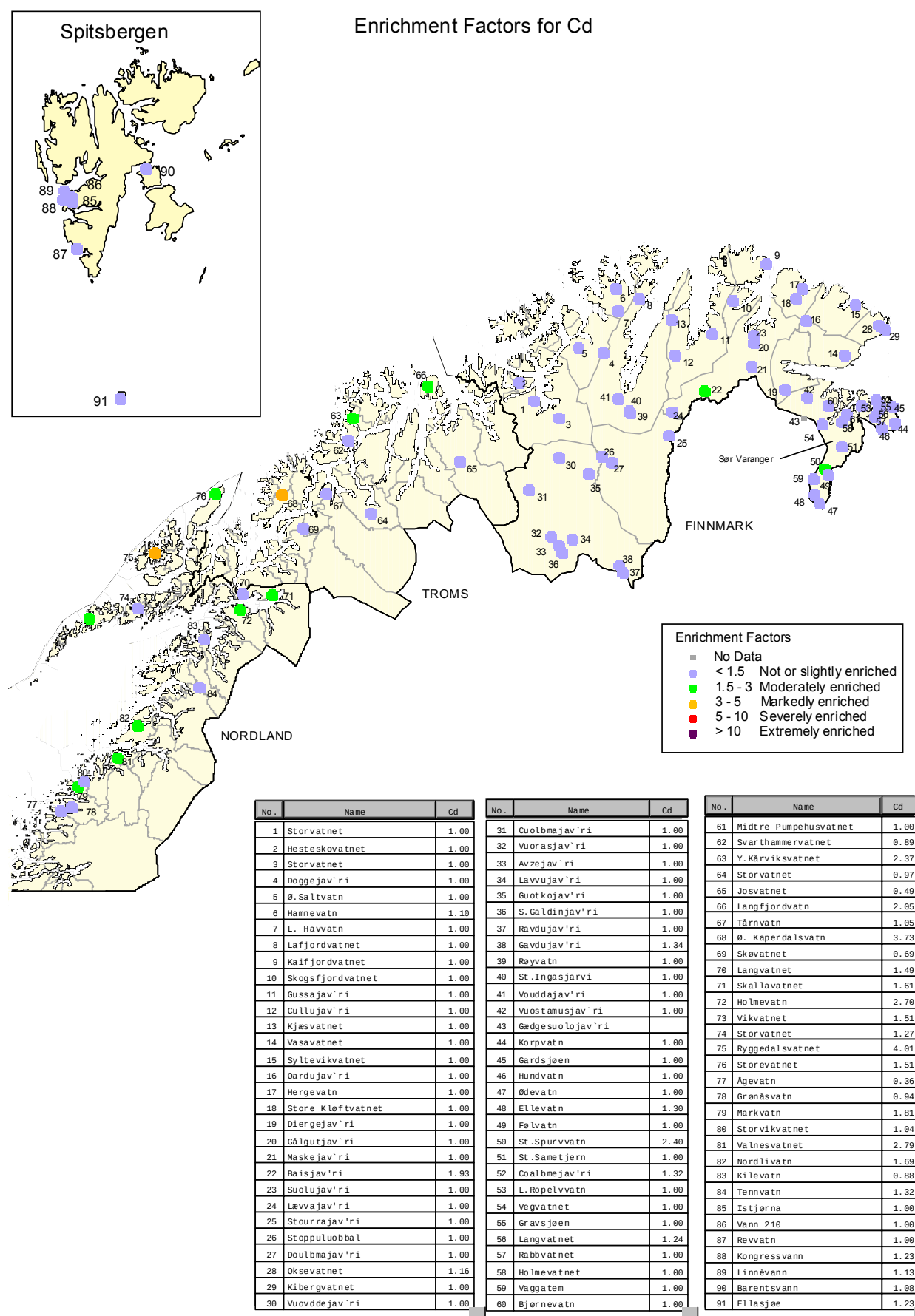


Figure 22. Enrichment factors for cadmium in surface sediments from lakes on the Northern Norwegian mainland and Norwegian Arctic islands.

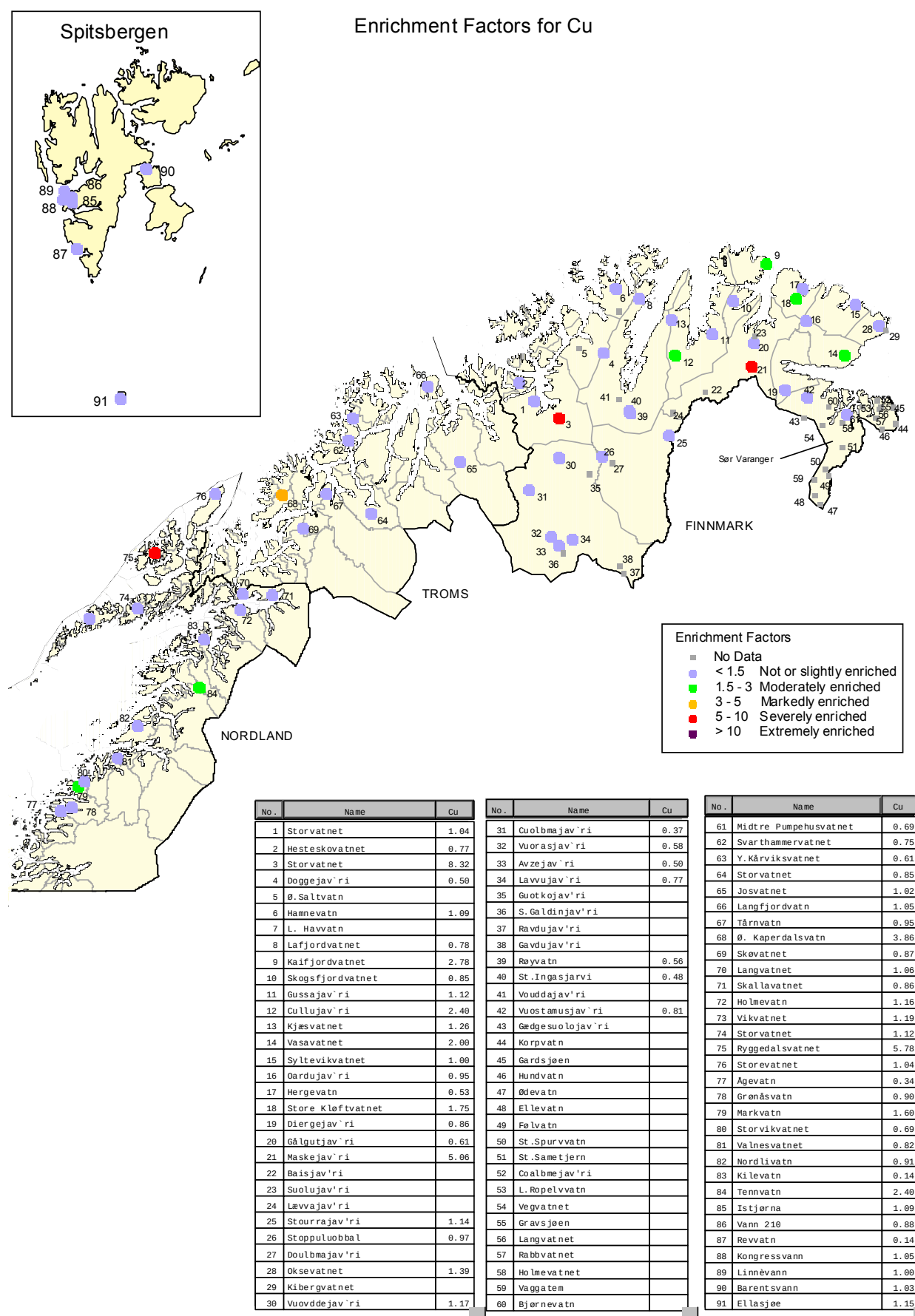


Figure 23. Enrichment factors for copper in surface sediments from lakes on the Northern Norwegian mainland and Norwegian Arctic islands.

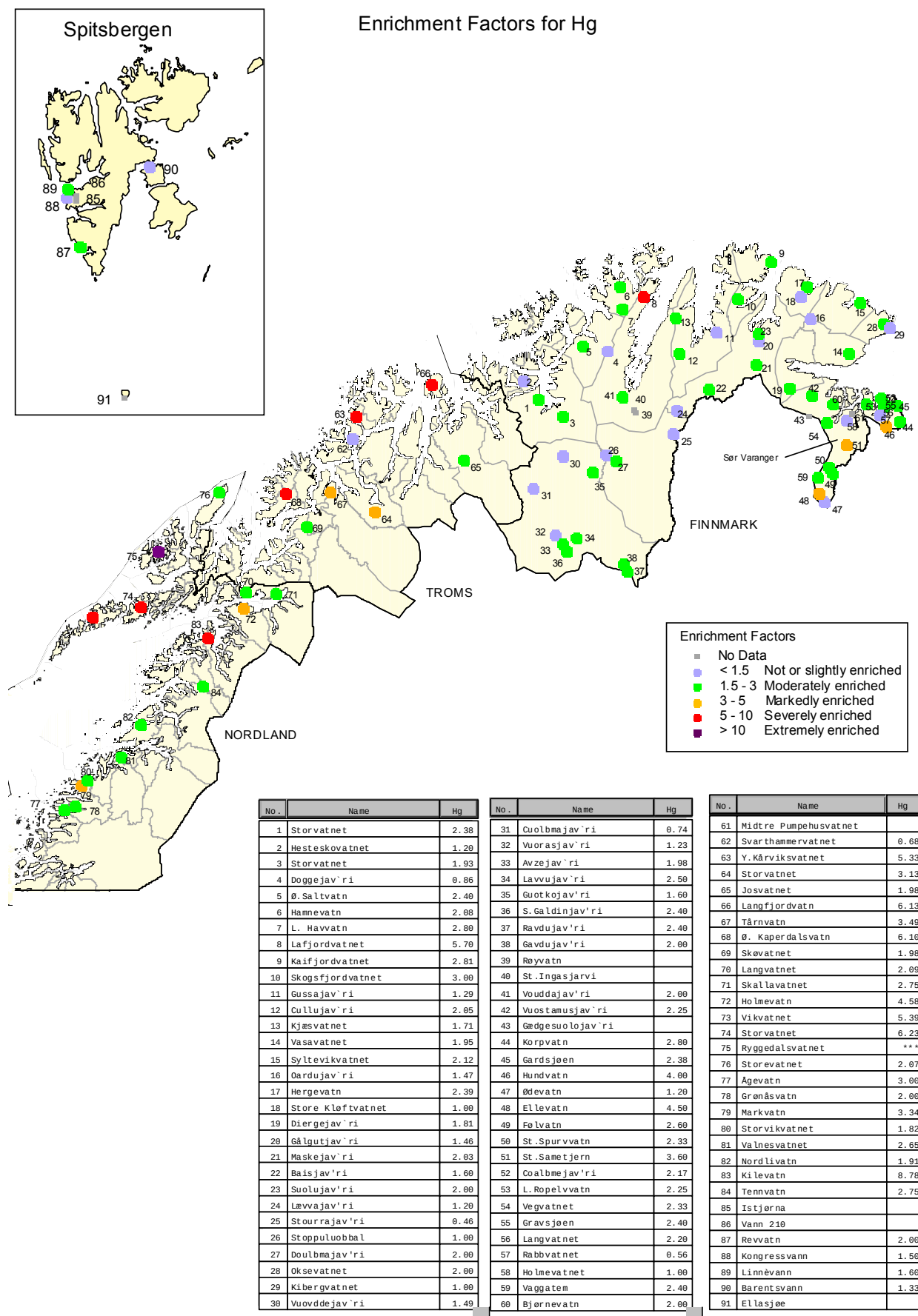


Figure 24. Enrichment factors for mercury in surface sediments from lakes on the Northern Norwegian mainland and Norwegian Arctic islands.

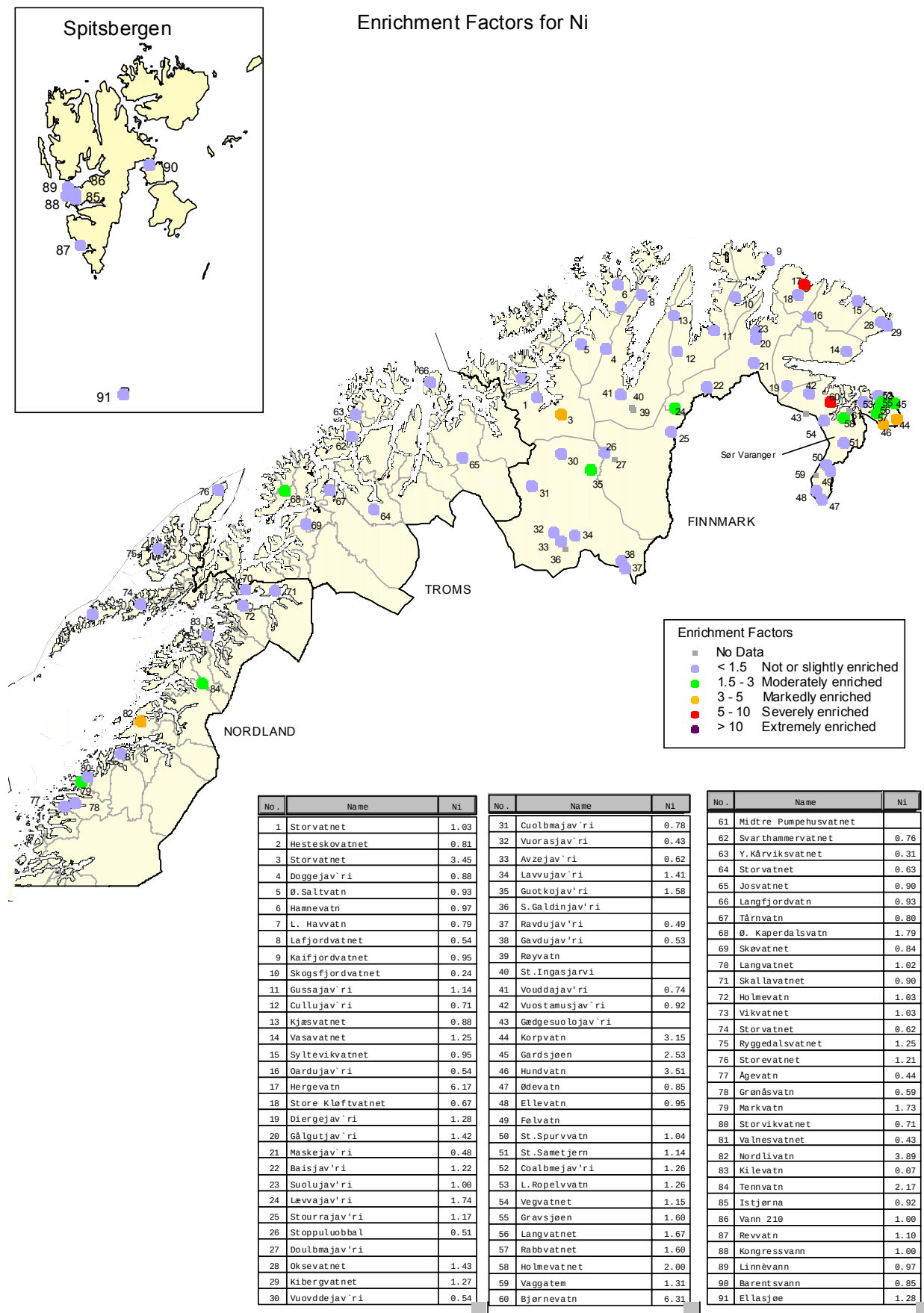


Figure 25. Enrichment factors for nickel in surface sediments from lakes on the Northern Norwegian mainland and Norwegian Arctic islands.

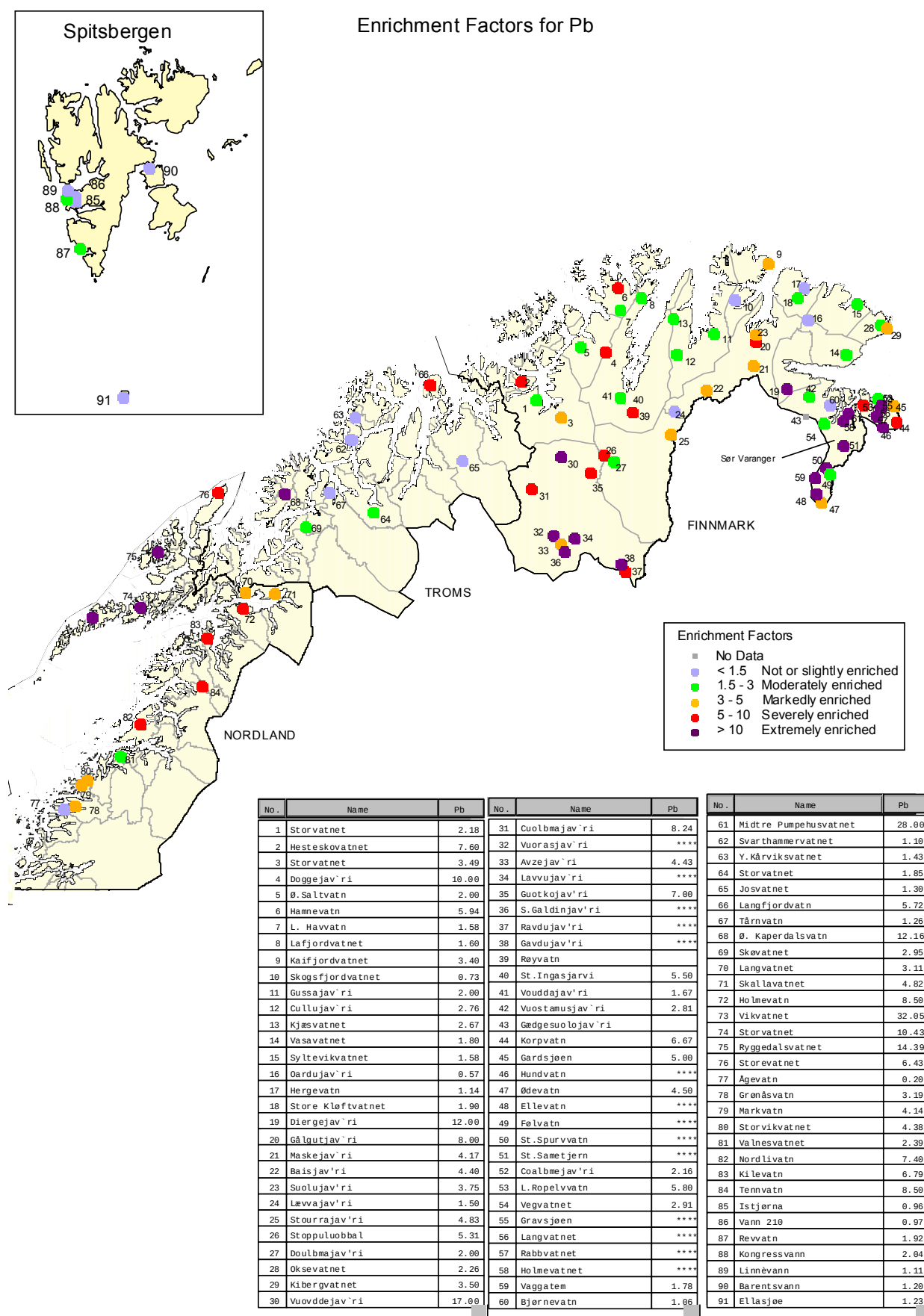


Figure 26. Enrichment factors for lead in surface sediments from lakes on the Northern Norwegian mainland and Norwegian Arctic islands.

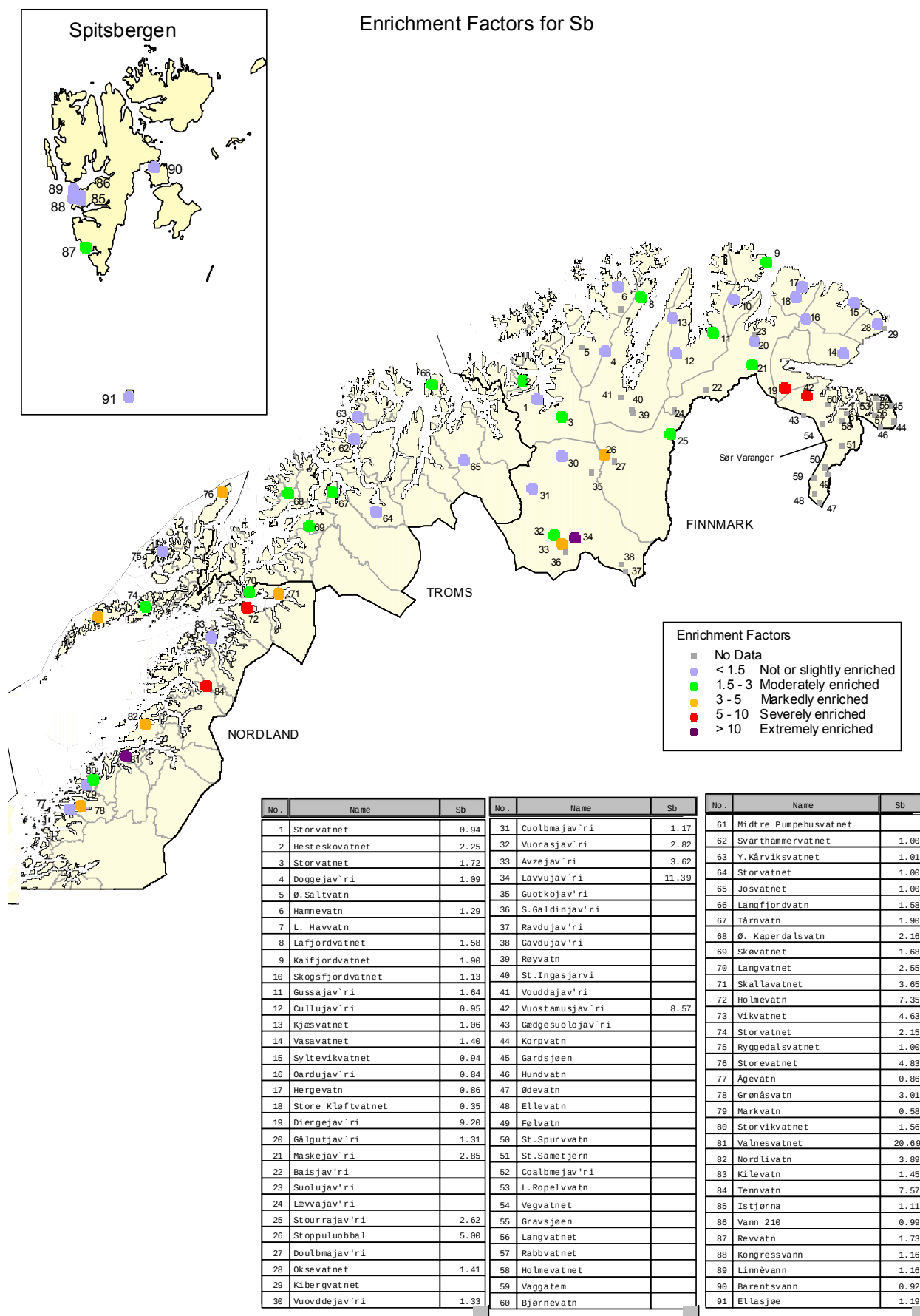


Figure 27. Enrichment factors for antimony in surface sediments from lakes on the Northern Norwegian mainland and Norwegian Arctic islands.

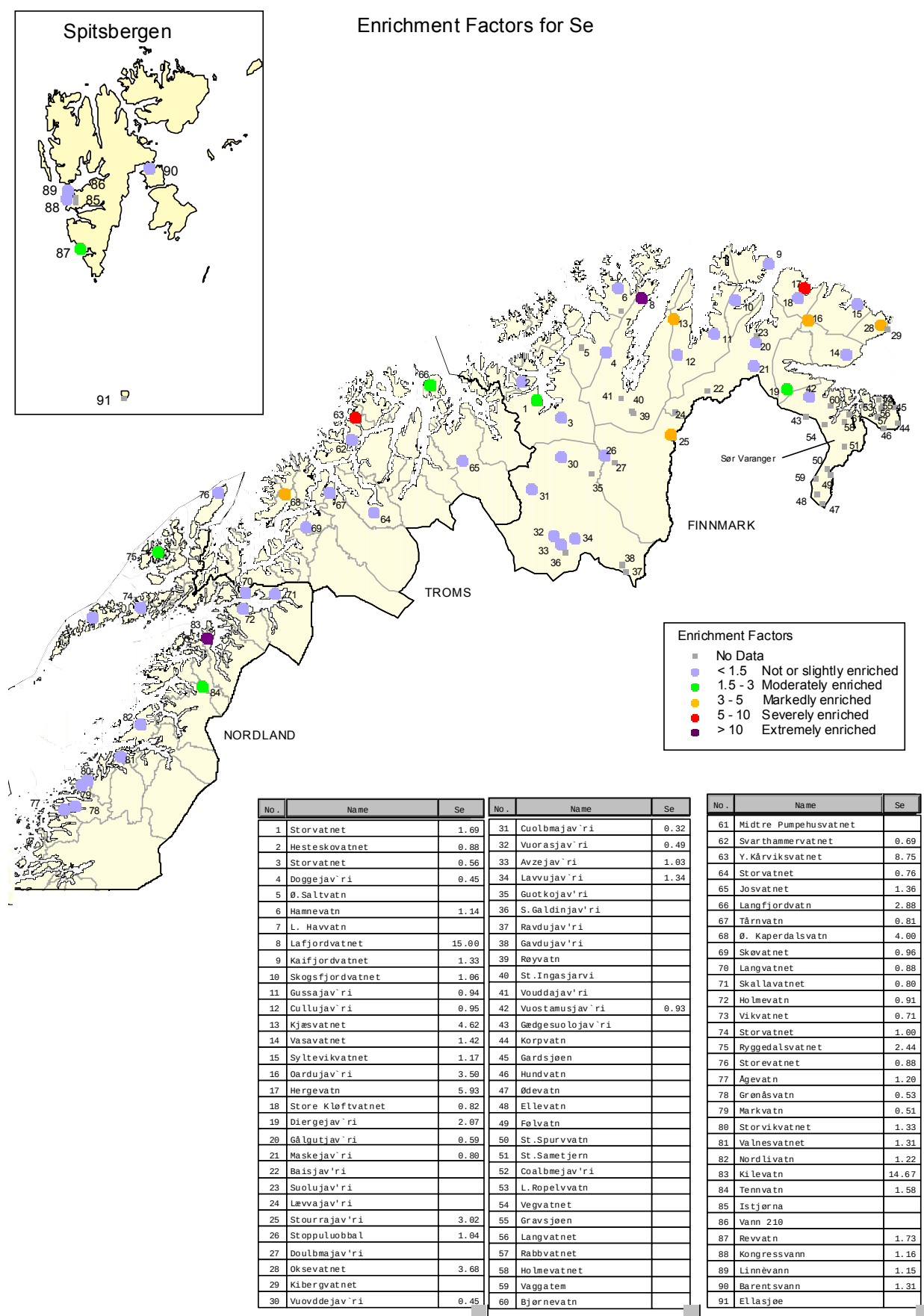


Figure 28. Enrichment factors for selenium in surface sediments from lakes on the Northern Norwegian mainland and Norwegian Arctic islands.

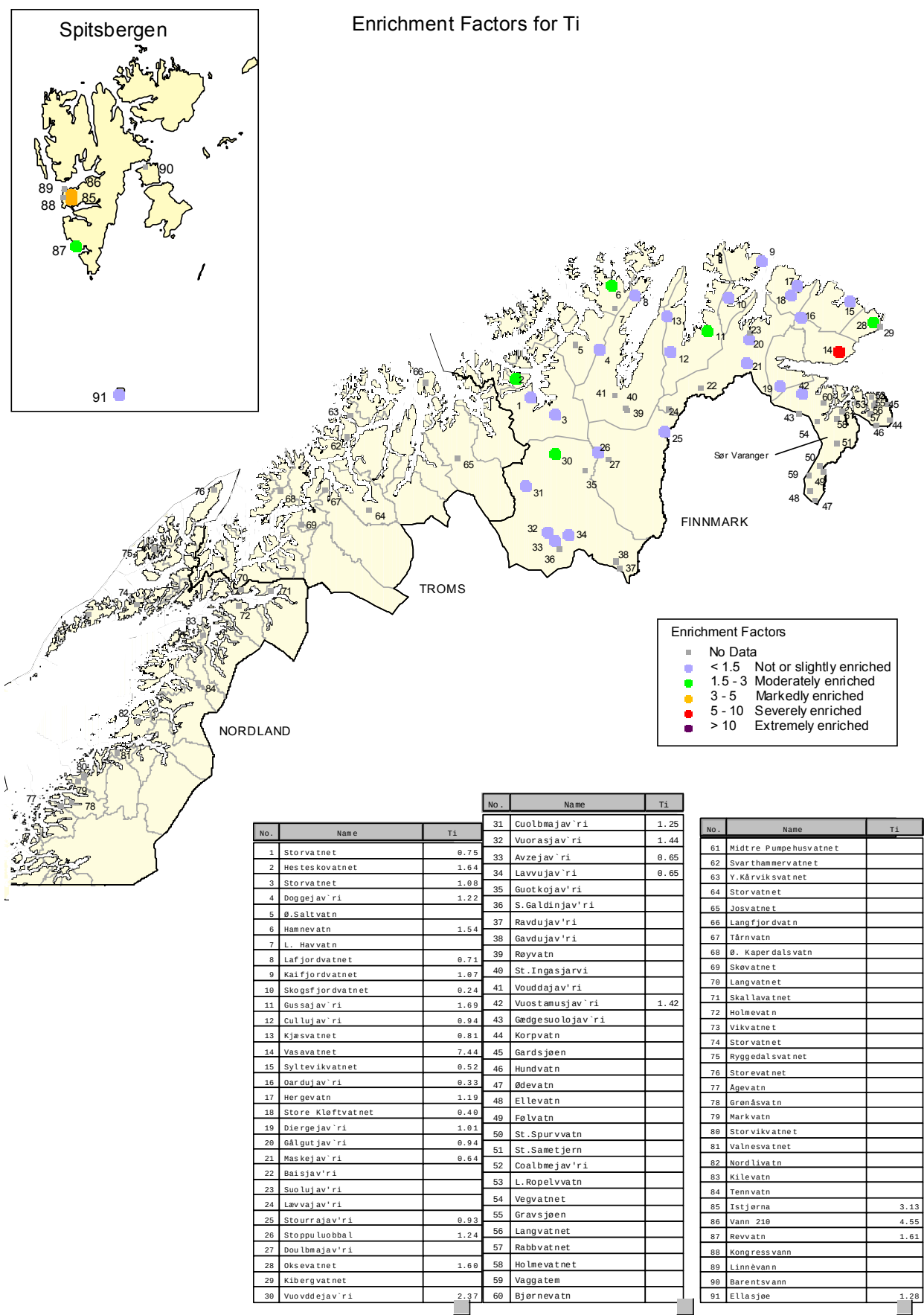


Figure 29. Enrichment factors for titane in surface sediments from lakes on the Northern Norwegian mainland and Norwegian Arctic islands.

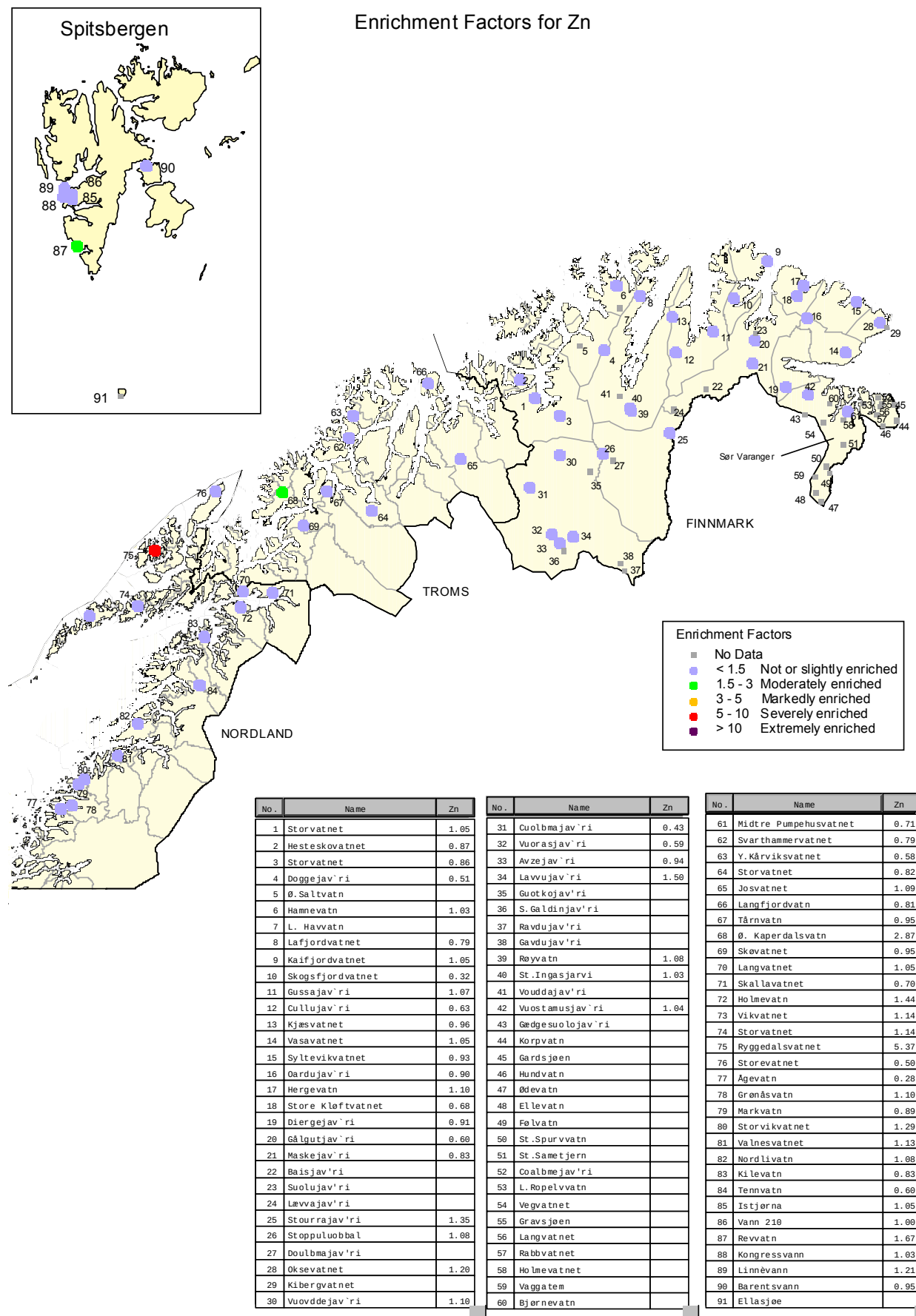


Figure 30. Enrichment factors for zinc in surface sediments from lakes on the Northern Norwegian mainland and Norwegian Arctic islands.

5.4 Persistent organic pollutants and PAHs in sediment

5.4.1 PCBs and HCBz

PCB concentrations in surface sediment from Northern Norway are low. Figure 32 shows that the concentrations of the "Seven Dutch" PCB congeners (Σ_7 PCB) in sediments of investigated lakes on the Norwegian mainland were all below 15 ng/g dry wt. The highest Σ_7 PCB levels on the Northern Norwegian mainland were found in some lakes in the Nordland and Troms Counties, as well as in Sør-Varanger. Surface sediments in lakes in the rest of Finnmark had Σ_7 PCB concentrations below 3 ng/g dry wt.

The highest concentration of Σ_7 PCB in surface sediment was recorded in Lake Ellasjøen on Bear Island, 32.7 ng/g dry wt. Compounds PCB-138, PCB-153 and PCB-180 made up the largest part of this sum value (respective concentrations 9.4, 12.9, and 5.0 ng/g dry wt.).

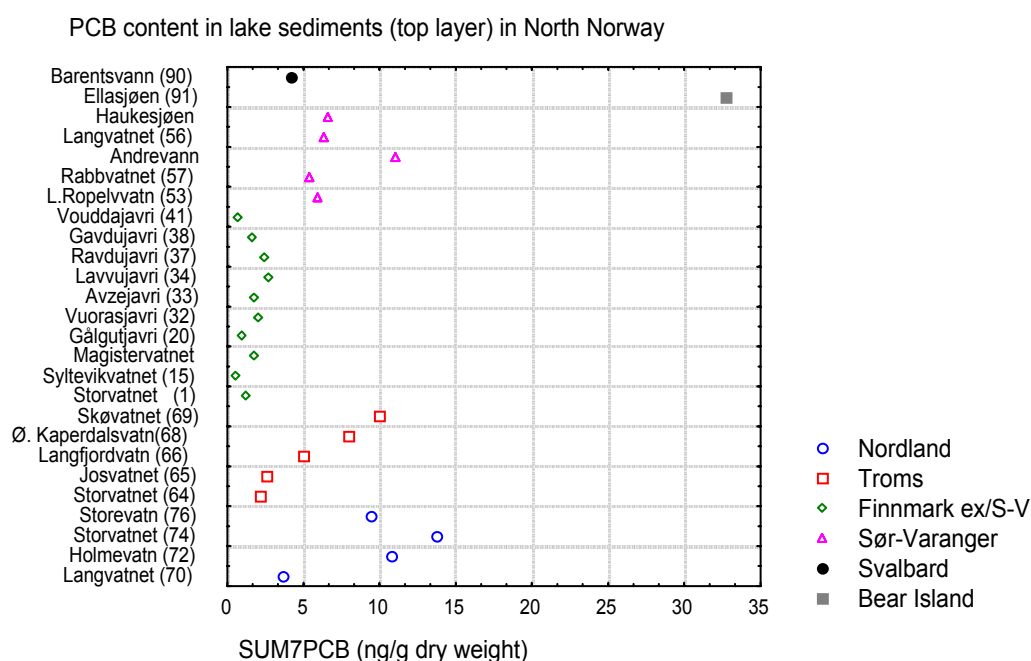


Figure 31. Maximum concentration of Σ_7 PCB in surface sediment (0-1 or 0-2 cm) from lakes on the Norwegian Arctic islands and the Northern Norwegian mainland.

The concentrations of HCBz in surface sediments are presented in Figure 32. HCBz concentrations in sediments from Nordland, Troms, Spitsbergen and Bear Island were below 1.5 ng/g dry wt. The levels in Finnmark were significantly elevated (between 1.5 and 5.5 ng/g dry wt.) compared to the Arctic islands and the other Northern counties. In Arctic lakes in North America, levels of HCBz in the surface sediment ranged from 0.1 to 1.8 ng/g dry wt. (Muir *et al.* 1995). It has to be remarked that the samples from Finnmark were analysed by the NILU laboratory, while the other samples were analysed at NIVA. The recovery percentage of the Finnmark samples was quite low, between 37 and 52% and some samples were below the required 5 times the blind value, due to small amounts of sample.

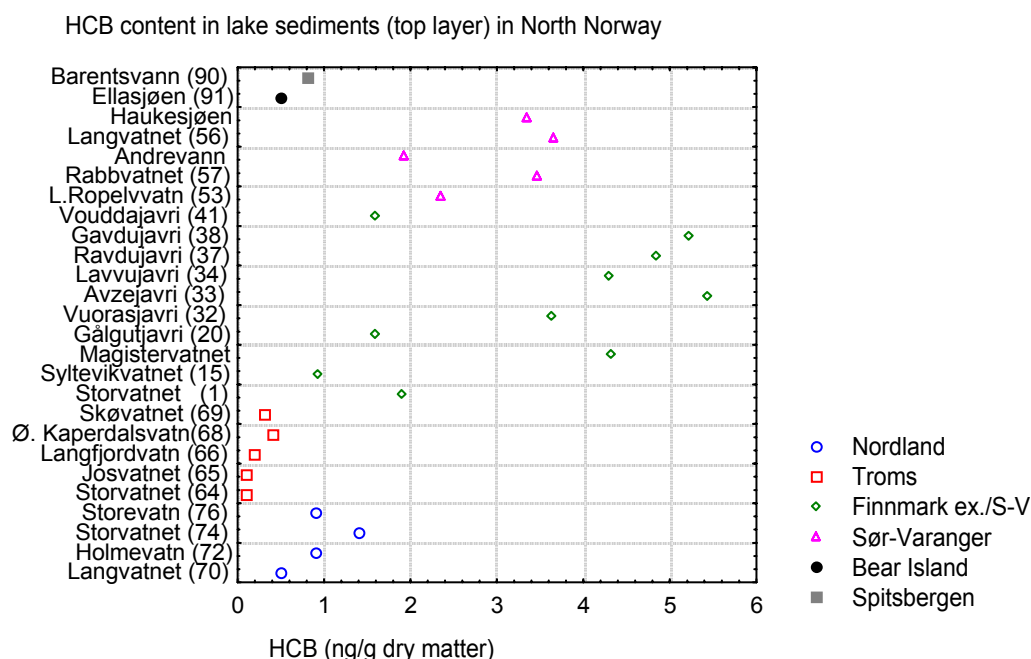


Figure 32. Maximum concentration of HCBz in surface sediment (0-1 or 0-2 cm) from lakes on the Norwegian Arctic islands and the Northern Norwegian mainland.

In addition to sum concentrations, also levels of individual congeners are of interest. Sum concentrations of PCBs and HCBz indicate pollution levels as well as their regional distribution. Of the PCBs, a total of 209 congeners are known, which have different structures, fate and behaviour. It is therefore important to know the levels of individual congeners in the investigated sediments.

Table 5 presents the minimum, maximum and average concentrations of the individual PCB congeners, as well as HCBz in all the surface sediments sampled.

Most of the 26 PCB congeners analysed in the samples from Finnmark were detected in the majority of the samples. The only congener which was not detectable in any of the samples was PCB-123, while PCB-114 and PCB-157 were detectable in only a few samples. Only in a few samples could concentrations of individual congeners be quantified to a satisfactory degree of accuracy, as most of these were present in very low concentrations. The reported concentrations did not exceed the quality standard of five times the blind value, due to low background concentrations and limited amounts of sample. The averages were calculated using all reported (maximum present) concentrations.

From the lakes in Finnmark, most congeners could be quantified in the lakes with the highest PCB concentrations, Andrevann and Røpevatn in Sør-Varanger. Also in Haukesjøen and in Langvatn some (mainly lower chlorinated) PCBs could be quantified.

Table 5. Minimum and maximum concentration, as well as averages of all sediment samples, for HCBz and the individual PCB congeners analysed in this investigation.

< = below detection limit or not quantifiable.

Compound	PCB (IUPAC-no.)	Min. (pg/g)	Max. (pg/g)	Average (pg/g)
HCBz		100	5427	1989.6
PCBs:				
TriCB	18	< 59.2	1634	572.1
	28	< 155	4361	1475.3
	31	< 103	3087	1062.5
TetCB	47	< 20.8	< 383	143.2
	52	< 49.5	3400	683.0
	66	< 0.06	1985	702.2
	74	< 0.06	1070	385.4
PenCB	99	< 29.9	655	185.4
	101	< 68.1	2100	512.0
	105	< 1.05	1200	162.1
	114	< 1.31	< 13.6	6.8
	118	< 3.25	3700	405.1
	123	< 1.4	80.3	10.7
HexCB	128	< 0.18	535	58.2
	138	< 0.16	9400	826.9
	149	< 67.4	1863	388.8
	153	< 83.1	12900	1012.5
	156	< 1.5	1100	156.6
	157	< 1.02	55.5	6.9
	167	< 1.49	227	22.4
HeptCB	170	< 16.4	1111	113.4
	180	< 33.9	5000	522.3
	187	< 30.7	1038	179.1
	189	< 0.36	75	10.8
	209	100	500	336.4
Σsigma(10) congeners		721.5	35000	6453.0
Σ7PCB		598.7	32700	5437.1

Σsigma(10)congeners = Iupac No. 28, 31, 52, 101, 105, 118, 138, 153, 156, 180

Σ7PCB= Iupac No. 28, 52, 101, 118, 138, 153, 180.

5.4.2 Pesticides

Detectable concentrations of only γ -HCH (lindane), trans- and cis-nonachlor, p,p'-DDE and Σ DDT were quantified in almost all lakes in Finnmark. Levels of pesticides in the surface sediments were generally very low (Table 6). Concentrations of γ -HCH, Σ DDT and p,p'-DDE in surface sediment from lakes in Northern Norway are presented in Figure 34, Figure 35 and Figure 33 respectively.

In addition to the above, the compounds which were detectable, but in only a few lakes, were: α -HCH, trans- and cis-chlordane, o,p'-DDE, o,p'-DDT and p,p'-DDT.

The following compounds were not detected at all in the top sediments: β -HCH (<8.88 to <0.18 ng/g), chlordane (<0.08 ng/g), heptachlor (<0.57 ng/g), dieldrin (<0.9 to <0.03 ng/g), aldrin (<0.78 ng/g) and endrin (<0.19 ng/g). Only in Langvatn was dieldrin detectable, at 1.1 ng/g. Since so many compounds were not quantifiable in different samples, these were not presented in diagrams.

Levels of pesticides in surface slices of sediment from Arctic lakes in Canada (Muir *et al.* 1995) are also presented in the following table.

Table 6. Minimum, maximum, average and median concentrations of all pesticides analysed in sediment samples from Northern Norway, under this investigation. For comparison are also the range of concentrations found in Arctic lakes in Canada (Muir *et al.* 1995) presented.

< = below detection limit or not quantifiable. Averages were calculated using those concentrations, which can be considered as the maximum present concentrations.

Pesticides	Min. (pg/g)	Max. (pg/g)	Average (pg/g)	Median (pg/g)	Range Canadian (pg/g)
α -HCH	<24.7	826	239.2	169.6	20-2070
β -HCH	<177	<8878	2082.6	<1302	<10-1700
γ -HCH	66.1	1241	334.4	359	10-620
Chlordane	<2	<86.1	19.3	<11	
Heptachlor	<29.2	<565	215.4	<171	<10-520
trans-Chlordane	<2.2	83.6	21.8	<16	<10-460
cis-Chlordane	<11.9	177	65.5	57.7	<10-1200
trans-Nonachlor	19.1	955	166.3	69.9	40-670
cis-Nonachlor	22	351	97.8	69.5	<10-660
Sum Chlordanes	<58.7	<1330.8	351.4		
Dieldrin	<31.1	1119	241.7	110	50-3170
Aldrin	<61.2	<784	277.0	<251	
Endrin	<14.1	<191	59.9	<51.9	
o,p'-DDE	<3.86	<65.1	21.7	14.9	
p,p'-DDE	64.2	4300	673.7	400	40-5070
o,p'-DDD	6.79	194	82.6	69.6	
p,p'-DDD	26.7	944	283.0	200	<10-4790
o,p'-DDT	<1.61	358	64.6	42	<10-900
p,p'-DDT	<2.41	579	169.1	100	<10-460
Sum DDT	<119	<3846	1135.6		100-10000

γ -HCH concentrations in the sediment ranged from 0.07 to 1.2 ng/g dry weight (Figure 33). The average concentration in the samples from Northern Norway was 0.3 ng/g dry weight.

In surface sediments from 8 Arctic lakes in Canada, the concentrations of γ -HCH were in the same range, between 0.01 and 0.62 ng/g dry weight (Muir *et al.* 1995). Three of those lakes had concentrations over 0.20 ng/g dry weight.

The concentrations of α -HCH in our lakes varied from 0.02 to 0.83, while those in the Canadian lakes were between 0.02 and 2.07 ng/g dry weight (Muir *et al.* 1995). Five of these lakes had concentrations below 0.50 ng/g dry weight.

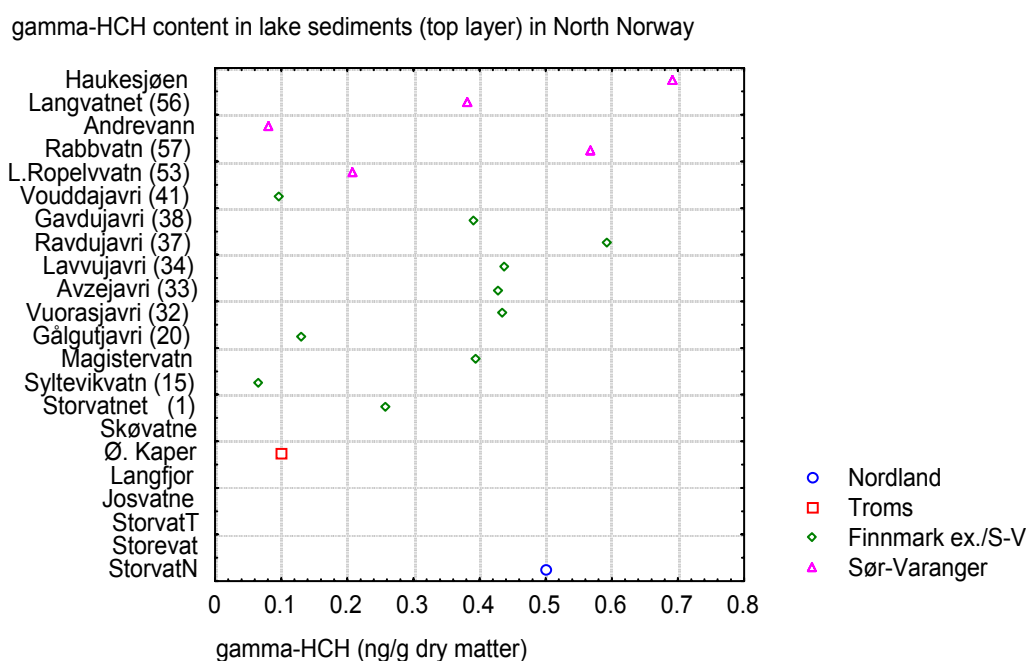


Figure 33. γ -HCH in surface sediment from lakes on the Northern Norwegian mainland.

Concentrations of Σ DDT in the top sediment varied between 0.1 and 3.8 ng/g dry weight (Figure 34). The average concentration over all the investigated lakes in Finnmark was 1.1 ng/g dry weight. The most abundant compound was p,p'-DDE (Figure 35), while o,p'-DDE had the lowest concentrations. o,p'-DDE, o,p'-DDT and p,p'-DDT were only detectable in some of the samples.

Similar levels were found in Arctic lake sediments in Canada. In 8 lakes, concentrations of Σ DDT ranged from 0.1 to 10 ng/g dry weight, with an average of 3.2 ng/g dry weight (Muir *et al.* 1995).

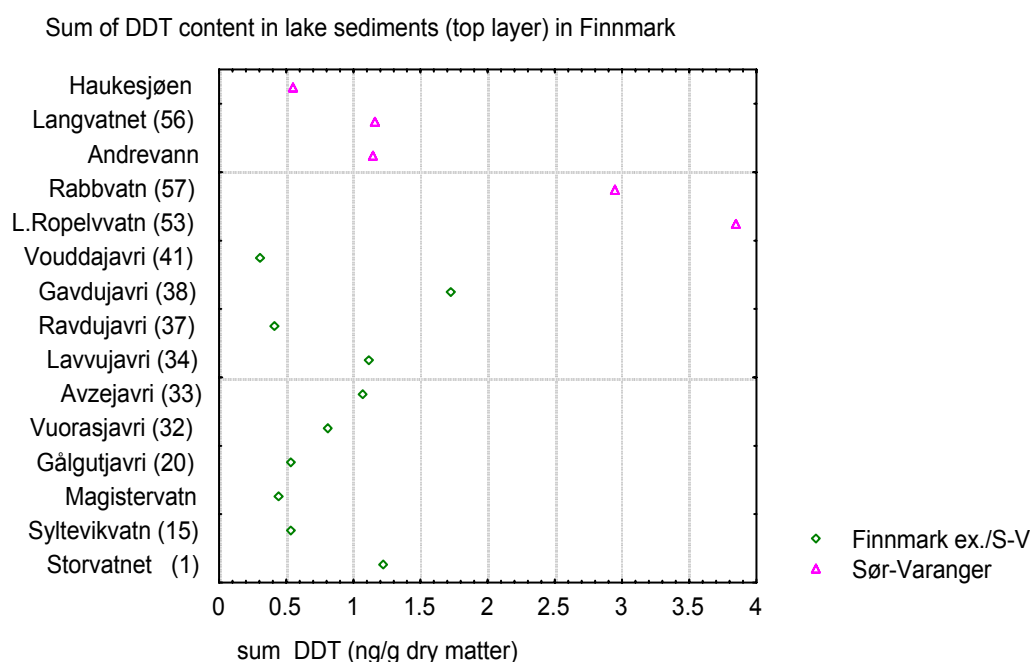


Figure 34. Concentration of Σ DDT in surface sediment from lakes on the Northern Norwegian mainland.

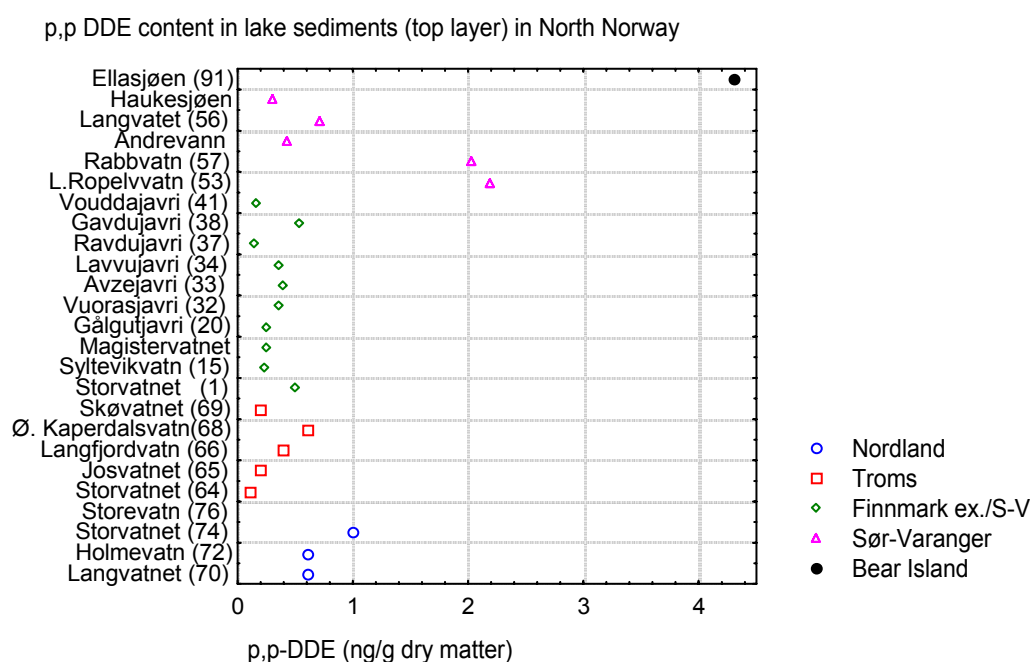


Figure 35. Concentration of p,p'-DDE in surface sediment from lakes on the Northern Norwegian mainland and a Norwegian Arctic island.

5.4.3 PAHs

The PAH concentrations were generally low. Figure 36 shows the sum concentrations of the analysed PAH compounds, except perylene, in the surface sediment layers of the investigated lakes. The highest Σ PAH levels in surface sediment were found in Nordland (4 to 7 $\mu\text{g/g}$ dry wt.). In Troms, Finnmark and Sør-Varanger, the Σ PAH concentrations were low, between 0.1-1.0 $\mu\text{g/g}$ dry wt. Concentrations of Σ PAH in the lakes on Spitsbergen and Bear Island were somewhat higher (0.9 resp. 1.2 $\mu\text{g/g}$ dry wt.) than the low levels found in Northern Norway.

Concentrations of all individual compounds were highest in sediments from Nordland. The concentrations of naphthalenes were lower in samples from Finnmark and Sør-Varanger than in Troms, Nordland, Bear Island and Spitsbergen.

All analysed compounds were quantified in sediment samples from Nordland, Bear Island and Spitsbergen, but some of the (heavier) compounds could not be detected in samples from Troms. In the sediment samples from Finnmark, the heavier PAH compounds (benzofluoranthenes, benzopyrenes and heavier ones) were quantified in all surface sediments. The most volatile and semi-volatile PAHs, on the other hand, were present in all samples from Finnmark, but it was not possible to quantify the concentrations in the samples available. These concentrations did not fulfil the laboratory requirement of being a factor of 10 above the blind-sample value. The actual concentrations are most likely to be below the reported concentrations (A. Mikalsen, NILU, pers. comm.). Thus, the PAH concentrations presented here represent maximum levels of PAHs present in the surface sediment.

PAH content in lake sediments (top layer: 0-2 cm) in North Norway

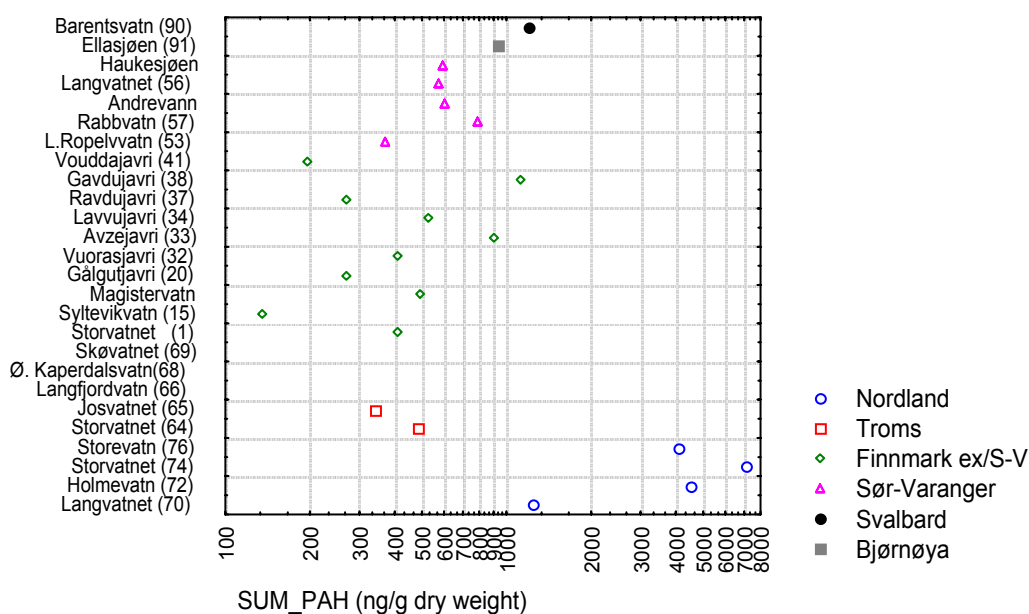


Figure 36. Concentration of Σ PAH (minus perylene) in surface sediment (0-1 or 0-2 cm) from lakes on the Norwegian Arctic islands and the Northern Norwegian mainland.

6. Contaminants in fish

6.1 Mercury (Hg) in fish

Samples for analyses of Hg in fish muscle were collected from 8 lakes in Finnmark County (no local contamination sources), Kautokeino River (some local contamination), Lake Ellasjøen on Bear Island (no local contamination), and Lake Finnsnesvatn (situated in urban surroundings).

After a presentation of Hg concentrations in different fish species (0), the concentrations will be discussed further and correlated to different parameters, such as fish length (0) and Hg levels in the lake sediments (0). In remote areas Hg concentrations can vary greatly. Geographical characteristics of the lake can also have a strong influence on the Hg concentrations (Dellinger *et al.* 1995). Other chemical parameters influencing the accumulation of Hg in fish are, for example, the pH and organic matter content in the water and Se levels in the sediment (Rognerud & Fjeld 1990). In paragraph 0. the consumption of fish with these concentrations is discussed.

6.1.1 Concentration levels

The recorded Hg concentrations in fish from the Northern Norwegian mainland and from Ellasjøen on Bear Island, are shown in Figure 37. The highest concentrations of Hg occurred among the predatory fish species, pike and perch. The concentrations of Hg in whitefish were lower and were similar in all the lakes sampled.

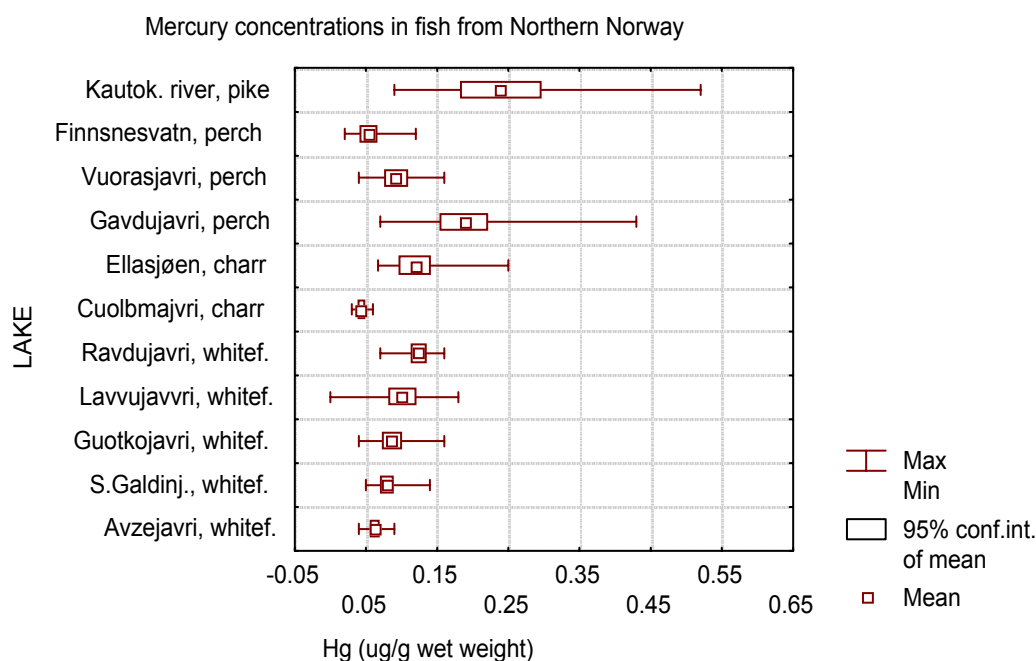


Figure 37. Hg concentrations ($\mu\text{g/g}$ wet weight) in muscle tissue of fish from Northern Norway.

6.1.1.1 Pike

The concentration of Hg in pike from the Kautokeino River varied between 0.09 and 0.52 µg/g wet weight, according to individual size (Figure 37, Table 7). The average concentration in the sampled pike was 0.24 µg/g wet weight. There was no difference in level of Hg contamination between pike caught upstream or downstream of the small village of Masi.

In Lake Iso Valkjärvi (Finland) the concentrations of Hg in pike ranged from 0.2 to 1.4 µg/g wet weight (Rask & Verta 1995). It was suggested that the concentrations depended on the size of individuals, but no data on this were presented. Since this makes correction for the fish lengths impossible, is it difficult to compare these levels directly with those in our study.

In remote lakes in Canada, Hg concentrations in pike were on average higher than those found in this study. In the case of four lakes within the Johnny Hoe River system, the average values were as follows: 0.74, 0.45, 0.35 and 0.48 µg/g wet weight (Stephens 1995). The pike in the Canadian study were of larger average sizes (68, 77, 69 and 51 cm respectively), which contributes to higher average Hg concentrations. Adjusted for body length, pike from two of the Canadian lakes (Keller Lake and Lac Taché) contained the same concentrations as those in our study, while pike from two other lakes (Lac Ste. Thérèse and Tseepantee Lake) had slightly higher concentrations.

6.1.1.2 Perch

Of the three lakes where Hg was analysed in perch, the highest concentrations in muscle tissue were seen in fish from Lake Gavdujavri (Figure 37, Table 7). The average concentrations in perch were 0.05 µg/g wet weight in Finnsnesvatn, 0.09 µg/g in Lake Vourasjavri, and 0.19 µg/g in Gavdujavri. The average lengths of the analysed perch increased in the same order. The concentration in perch from Gavdujavri ranged from 0.07 to 0.43 µg/g. Concentrations in perch from the Lake Iso Valkjärvi in Finland varied from 0.1 to 0.7 µg/g wet weight (Rask & Verta 1995).

6.1.1.3 Whitefish

The average concentrations in whitefish muscle tissue were below 0.13 µg/g wet weight for all lakes (Figure 37, Table 7). The highest analysed concentration in whitefish was 0.16 µg/g (in Lavvujavri). All sampled whitefish in this investigation had concentrations of Hg which were far below the levels of restricted consumption (further discussed in section 0).

Concentrations total Hg in whitefish in Lake Iso Valkjärvi in Finland were between 0.06 and 0.14 µg/g (wet weight) (Rask & Verta 1995). Concentrations in whitefish from Canada were for four lakes: 0.27, 0.06, 0.07, and 0.10 µg/g wet weight. The lengths of these fish ranged from 30 to 55 cm (Stephens 1995).

6.1.1.4 Arctic char

The average concentrations in muscle tissue from Arctic char were 0.04 and 0.12 µg/g wet weight for Cuolbmajavri in Finnmark and Lake Ellasjøen on Bear Island respectively (Table 7). The average lengths were 26 and 28 cm.

The concentrations of Hg in char from Cuolbmajavri and Lake Ellasjøen were lower than those from Hazen Lake on N. Ellesmere Island, Canada. There the average concentration in

muscle tissue from 45 fish was 0.18 ± 0.09 µg/g (Muir & Lockhart 1993a). In Amituk Lake, N.W.T. Canada, 28 cm long fish had a Hg-concentration about 0.5 µg/g wet weight (Lockhart 1995).

Table 7. Data on fish sampled for this investigation and the Hg concentrations in muscle tissue (average, standard deviation and confidence interval).

Averages are calculated from 20 fish collected from each lake.

Lake	Fish species	Length (mm)	Weight (g)	Age (years)	Hg (µg/g)			
		average	average	average	average	st. dev	95% conf. interval	
Avzejavri	whitefish	202	128	6.3	0.063	0.016	0.056	0.069
S.Galdinjavri	whitefish	222	146	8.0	0.079	0.021	0.070	0.089
Guotkojavri	whitefish	250	201	7.3	0.086	0.029	0.073	0.099
Lavvujavri	whitefish	219	161	7.8	0.102	0.041	0.083	0.120
Ravdujavri	whitefish	349	458	7.9	0.123	0.023	0.113	0.134
Cuolbmajavri	Arctic char	261	197	7.7	0.043	0.009	0.039	0.047
Ellasjøen	Arctic char	277		9.4	0.119	0.050	0.097	0.141
Gavdujavri	perch	220	189	10.4	0.189	0.076	0.156	0.222
Vuorasjavri	perch	207	130	7.8	0.092	0.037	0.076	0.108
Finnsnesvatn	perch	199	100	5.4	0.053	0.028	0.041	0.065
Kautokeino River	pike	597	1877	8.8	0.239	0.130	0.182	0.296

6.1.2 Correlation between Hg concentration and fish length

Methylmercury is not easily dissolved in water, but it tends to bind to proteins. Fish do not have an effective mechanism to excrete methylmercury and, like POPs, it is known to bioaccumulate in freshwater biota. Concentrations in fish often depend on the size of the fish (Knutzen & Green 1995), best described by their lengths (Rognerud & Fjeld 1990).

6.1.2.1 Pike and Perch

From Table 8, it is evident that in the case of pike and perch, there is a significant correlation ($p < 0.05$) between fish length and the concentration of Hg in muscle tissue. For Arctic char and most whitefish, the Hg concentration was not significantly correlated with the length of the fish. The correlation between Hg levels and perch length was stronger for the lake with the largest fish (Gavdujavri) than for Lake Finnsnesvatn, which had a stunted population of small perch. Those small fish did not feed on other fish and were on a lower trophic level.

One explanation for the difference between fish species could be that perch and pike generally have different feeding habits from most Arctic char and whitefish. Pike and perch, especially the larger individuals, are predatory fish, feeding on prey fish and invertebrates. Most Arctic char and whitefish feed on a lower trophic level, with zooplankton or bottom-dwelling animals being the main food sources.

Table 8. Spearman rank order correlation between fish length and mercury concentration (Hg) in muscle tissue for each lake.

Lake	Fish species	N	R	p-level
Avzejavri	whitefish	20	0.22	0.34
S.Galdinjavri	whitefish	20	-0.13	0.58
Guotkajavri	whitefish	20	0.48	0.03
Lavvujavri	whitefish	19	0.11	0.65
Ravdujavri	whitefish	20	0.08	0.73
Coulbmajavri	Arctic char	19	0.21	0.39
Ellasjøen	Arctic char	20	-0.12	0.62
Gavdujavri	perch	20	0.74	0.00
Vuorasjavri	perch	20	0.65	0.00
Finnsnesvatn	perch	20	0.46	0.04
Kautokeinoelva	pike	20	0.86	0.00

The concentration of Hg in pike appeared to increase with body size, for fish larger than about 50 cm (Figure 38). The same applies to perch above about 20-25 cm. Hg concentrations over 0.3 µg/g were found in pike with a length of over 65 cm and a body weight of over 2.0 kg.

In the case of perch, the highest concentrations were recorded in the largest individuals (Figure 39). Hg concentrations in the largest perch in Gavdujavri (31 and 32 cm long and a weight of 0.5 kg) were 0.30 and 0.43 µg/g wet weight respectively.

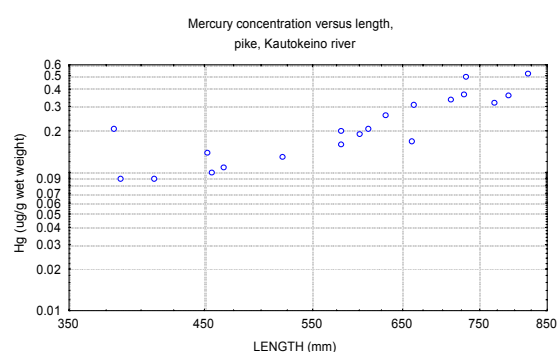


Figure 38. Hg concentrations in muscle tissue (wet weight) versus length of pike from the Kautokeino River.

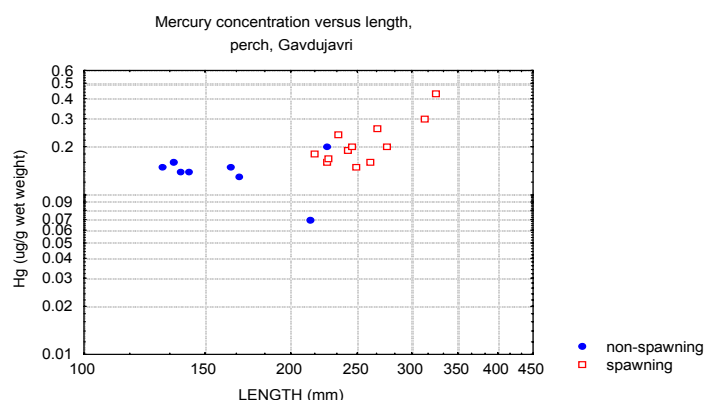


Figure 39. Hg concentrations in muscle tissue versus length of perch from Lake Gavdujavri.

6.1.2.2 Whitefish

For whitefish, there was no significant correlation between body length and the concentration of Hg in the fish muscle. Thus, larger fish do not necessarily contain higher levels of Hg than smaller fish. However, in Avzejavri there was a slight trend for fish which would spawn the coming autumn to have higher Hg concentrations than non-spawning fish of the same length (Figure 40). This was not the case for whitefish from the other lakes, e.g. Lake S. Galdinjavri in Figure 41.

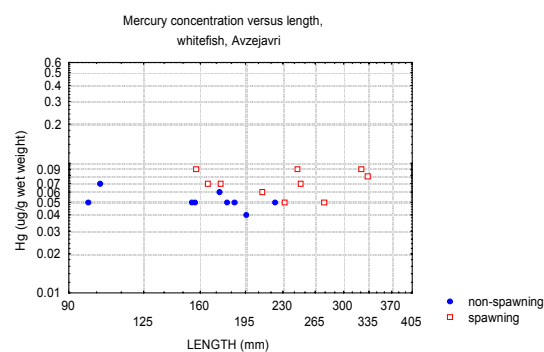


Figure 40. Hg concentrations in muscle tissue versus length of whitefish from Lake Avzejavri. Red symbols indicate those of the fish caught in august, which will spawn during that following autumn.

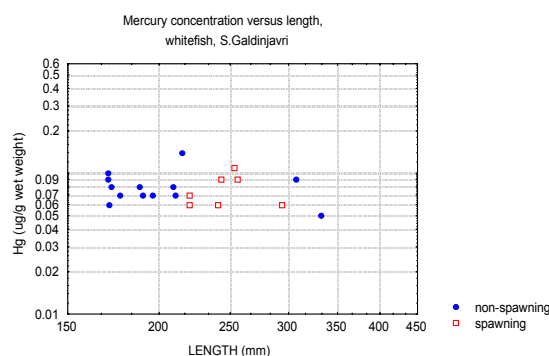


Figure 41. Hg concentrations in whitefish muscle tissue from Lake S. Galdinjavri.

The Hg concentration in spawning whitefish was found to be significantly different from that in non-spawning individuals ($p = 0.0001$) (Table 9). There was also a significant length difference between spawning/non-spawning fish. When the effect of body length on Hg concentration was removed, by using length as a co-variable in ANOVA, the difference in Hg concentration between spawning and non-spawning fish was no longer significant at a 5% level ($p = 0.074$). This indicates that the Hg concentration in the muscle tissue was primarily related to the length, rather than the state of maturity of the individuals.

Comparing whitefish from all the lakes investigated, statistical analysis showed that there were no significant differences in Hg concentrations between male/female whitefish. Neither was there a significant length difference between male/female individuals (Table 9).

Table 9. Results from statistical correlation analyses of Hg concentrations in different groups from the whole material of whitefish.

	Mean values Hg (µg/g)	Mean values Length (mm)
non-spawning	0.075	198
spawning	0.100	280
p-level	0.0001	$2.2 \cdot 10^{-9}$
female	0.091	257
male	0.090	244
p-level	0.96	0.33

6.1.3 Comparison of Hg concentrations in sediment and in fish

6.1.3.1 Perch

There appeared to be a positive relationship between Hg concentration in the perch and that in the sediment. Although the perch were collected from two lakes unaffected by local contamination, Lake Gavdujavri contained the highest Hg concentration in the sediment ($0.100 \mu\text{g/g}$), compared with $0.064 \mu\text{g/g}$ in Vourasjavri (Table 10). The concentration of Hg in perch was also higher in Lake Gavdujavri (average: $0.19 \mu\text{g/g}$) than in Lake Vourasjavri ($0.09 \mu\text{g/g}$). However, this observation should be treated with some caution, due to the scarcity of data available.

6.1.3.2 Whitefish

Whitefish from four of the lakes were of similar sizes (10-35 cm), but fish from Ravdujavri were significantly larger (32-39 cm), although the average age was similar. In general, two types of whitefish are found in Finnmark, one utilising mainly benthic organisms as prey while the other mainly feeds on plankton in the pelagic zone. The bottom feeding whitefish tend to grow to a larger body size than the pelagic-feeding type. The whitefish from Ravdujavri are likely to belong to the bottom-feeding type, being significantly larger at a similar age.

Figure 42 shows that there is no positive correlation between the concentration of Hg in sediment and the average Hg concentration in whitefish. In the previous section, it was shown that there was no correlation between the length of the whitefish and the Hg concentration in the muscle tissue. The higher concentration in Ravdujavri could be explained by different feeding habits between the whitefish populations. It is known from literature that the contamination in fish increases with the trophic level of their prey (Kidd *et al.* 1995).

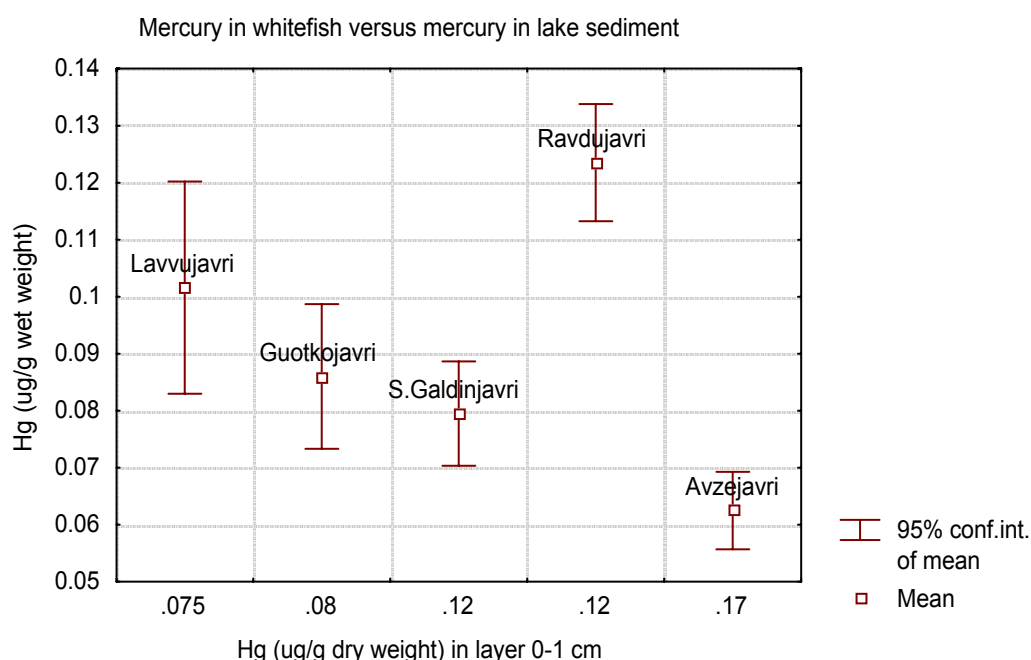


Figure 42. Hg concentrations in whitefish versus Hg concentration in the top layer (0-1 cm) sediment.

Selenium

The lowest Hg concentration in fish sampled from the lake with the highest levels of Hg in sediment can partly be explained by the Se concentrations in the sediment. The concentration of Se appears to reduce the uptake of mercury by biota (Schindler *et al.* 1995; Rognerud & Fjeld 1990). It may function as an antagonist, occupying the same binding sites as mercury. Rognerud & Fjeld (1990) also found a negative correlation between the concentration of Se in the sediment and the concentration in fish from the same lake. In this investigation, we know the Se concentration in two of the lakes with whitefish (Table 10). Although based on scarce data, a similar trend is indicated. The lake with the fish with the highest concentrations of Hg has the lowest concentrations of Se in the sediment, while the sediment in the lake with the lowest Hg concentration in the whitefish has the highest levels of Se.

Another factor influencing the mercury concentrations in fish is the content of organic matter in the water (Rognerud & Fjeld 1990). However, organic matter in the water was not analysed in this study.

Table 10. Some characteristics of the sediment (top layer 0-1 cm) and the catchment areas of the lakes from which fish were analysed for Hg contamination.

Se: Selenium, sed.: sediment, LOI: Loss of ignition (%), OC: Organic carbon content (%).

Navn	Hg sed. µg/g dry wt.	Se sed. µg/g dry wt.	LOI % sed.	OC % sed.	Lake depth m	L.A.S m	Lake area km ²	Catchment area km ²	Swamp area (%)	Wood area (%)
Avzejav'ri	0.170	3.90	33.0	17.4	23	361	1.63	166.4	30	30
S.Galdinjav'ri	0.120		37.0	19.5		396	0.52	2.29	19.1	70
Guotkojav'ri	0.080		22.2	11.7		425	0.81	3.19	7.8	27.5
Lavvujav'ri	0.075	0.90	27.0	14.2	6	476	1.47	29.1	15.5	0
Ravdujav'ri	0.120		15.7	8.3		432	0.78	8.4	4	14
Cuolbmajav'ri	0.058	1.70	15.9	8.4	8	548	0.63	5.1	0	0
Ellasjøen										
Gavdujav'ri	0.100		17.8	9.4		441	2.75	53.4	15	0
Vuorasjav'ri	0.064	1.90	25.8	13.6	8	345	2.55	41.9	9.3	50
Finnsnesvatn	0.13		94.7	48.4	4	5	0.014	0.34		

6.1.4 Limits for consumption

In accordance with the latest EU guidelines, Norway has established new guidelines for mercury levels in fish products (Social and Health Department 1995). Thus the average concentration of Hg in fish products used for consumption should not exceed 0.5 mg/kg wet weight. For pike and lipid-rich species, a higher maximum limit of 1.0 mg/kg wet weight is defined.

The lowest recommended concentration of Hg in fish muscle is the Danish limit of 0.3 mg/kg fresh muscle, which was proposed as the EU-directive (Knutzen & Skei 1990).

The Norwegian Health Directory has previously issued recommendations for the frequency of consumption of fish with different levels of contamination. These recommendations were based on advice from the "Joint Expert Committee on Food Additives" from the World Health Organisation (WHO) and FNs food organisation (FAO). Fish with a Hg concentration over 1 mg/kg should only occasionally be consumed, while fish with concentrations between 0.3 and 0.5 mg/kg should not be consumed more than 3-4 times per week (Rognerud & Fjeld 1990).

All char and whitefish, as well as the small to medium size perch and pike in this investigation, contained Hg concentrations below 0.3 mg/kg wet weight. Only the largest pike (in the Kautokeino River) and the largest perch (in Gavdujavri) had concentrations above 0.3 mg/kg (0).

6.2 Organic pollutants in fish

Organochlorine pollutants (PCBs and persistent pesticides) and PAHs were analysed in fish from Spitsbergen, Bear Island and Finnmark. Concentrations of these components as well as the percentage extractable lipid (fat) were determined in muscle tissue from both individual fish and pooled samples (between 2-6 fish). The pooled samples consisted of fish of similar sex and maturity (spawning/non-spawning). A number of pooled samples were analysed from most lakes in Finnmark and on Spitsbergen (Table 3).

Other studies did not detect a significant correlation between organochlorine concentration and fish length or weight for whole data sets (e.g. Wilson *et al.* 1995). Neither did Wilson find a clear distinction between mean organochlorine concentration for male and female fish. Therefore, this report presents levels of contaminants in fish expressed as averages for each lake. Lake averages were calculated from the pooled samples by weighting the concentration in the pooled sample with two.

6.2.1 Lipid content

Levels of organochlorines in muscle tissue are presented both on a wet weight and lipid basis (per gram extractable lipid). The most hydrophobic compounds are accumulated mainly in association with the lipid fraction, while the more polar organochlorines tend to accumulate in other tissues (Schmitt *et al.* 1990). Although concentrations of organochlorines are often presented on a lipid basis, the use of lipid normalisation is not unambiguous. Lipid normalisation does not generally reduce the standard deviation in the material (Muir & Lockhart 1993b; Hammar *et al.* 1993). Lipid normalisation can lead to erroneous interpretation of the results, if the contaminant concentration does not vary in direct proportion to the lipid content (Herbert & Keensleyside 1995). Care should be taken when comparing results from different methods of lipid or fat analysis, which may extract different fractions of the lipids in the sample.

The lipid content in muscle tissue depends on various size and condition traits (Hammar *et al.* 1993). An overview of the fish samples analysed, as well as the lipid contents in the samples is presented in Table 3.

Lipid contents in muscle tissue from whitefish ranged from 0.19 to 2.61% of the wet weight, with most samples containing less than 1%. In perch, the lipid contents ranged from 0.44 to 1.56% and in Arctic char from Spitsbergen and Bear Island from 0.37 to 6.2%.

The percentage of extracted lipid in whitefish from Northern Sweden was similar to the levels in our study, 0.66% (Jansson *et al.* 1993). The lipid content in whitefish from Canada was higher, and varied between 1 and 3.5% (Evans 1994; Kidd *et al.* 1994). Lipid concentrations in Arctic char from Northern Sweden ranged from 0.9 to 1.9% (Andersson *et al.* 1988).

6.2.2 PCBs and HCBz

6.2.2.1 PCB levels

PCB concentrations were the highest in muscle tissue from Arctic char from Bear Island. The total concentration of the seven Dutch PCBs (Σ_7 PCB) was 1292 ng/g on a wet weight basis and on a lipid basis 50 000 ng/g extractable lipid (Figure 43 and Figure 44). This sample was reanalysed several times by the NILU laboratory for quality control, resulting in the same ex-

tremely high level. This level is several times higher than the worst case reported from the American Arctic. Even if this concentration is based on a limited material (one fish), and needs to be verified, this level is so high that it must be classified as alarming. In order to find a possible explanation for those values a new investigation on Bear Island was conducted in 1996. Samples were collected from a large range of fish sizes, as well as from their prey items.

In Arctic char from Spitsbergen, the $\Sigma_7\text{PCB}$ ranged from 2.4 to 34.4 ng/g w.w. and from 64 to 5182 ng/g extractable lipid. Concentrations of over 1500 ng/g lipid were recorded in small sized Arctic char from Lakes Linnèvann and Kongressvatn on Spitsbergen.

Low concentrations were measured in whitefish; all below 1 ng/g w.w. and between 45 and 200 ng/g lipid. In perch, concentrations of $\Sigma_7\text{PCB}$ were higher than in whitefish, up to 6 ng/g, with averages for the two lakes of 2 and 3 ng/g w.w. and 25 to 700 ng/g lipid respectively.

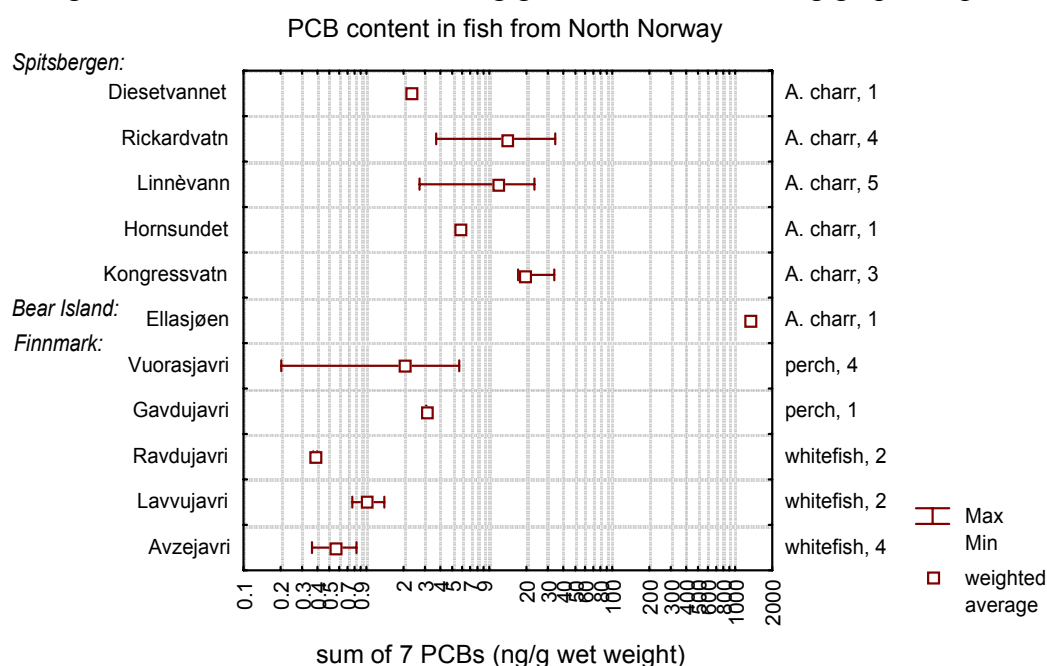


Figure 43. Minimum, maximum and average concentrations of $\Sigma_7\text{PCB}$ in muscle tissue (ng/g w.w.) of fish from lakes on the Northern Norwegian mainland and a Norwegian Arctic island.

The number behind the fish species refers the number of (pooled or individual) samples from that lake, used to calculate weighted average concentrations.

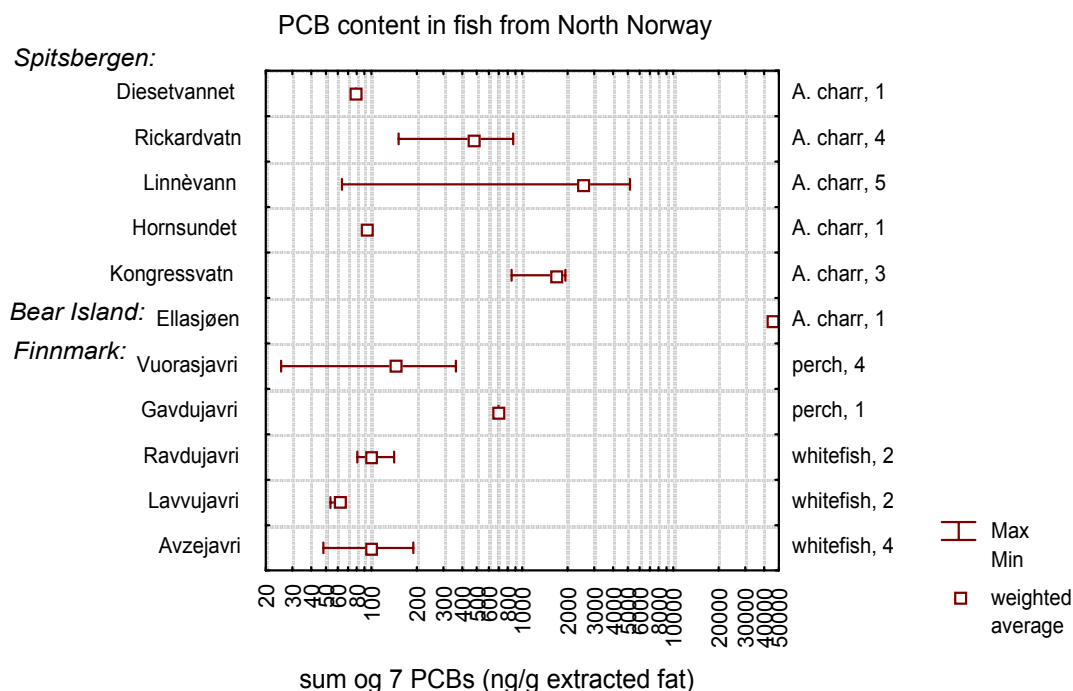


Figure 44. Minimum, maximum and average concentrations of $\Sigma_7\text{PCB}$ in muscle tissue (ng/g lipid) of fish from lakes on the Northern Norwegian mainland and a Norwegian Arctic island.

Explanation of the number behind the fish species in previous figure.

Arctic char from three remote lakes in the Northwest Territories in Canada had average ΣPCB (90 congeners) concentrations of 7.3 ± 1.8 , 71.0 ± 48.6 and 289 ± 118 ng/g w.w., in Buchanan Lake, Amituk Lake and Char Lake respectively (Muir & Lockhart 1993a). These levels are similar and higher than the levels found in fish from the remote lakes on Spitsbergen. The concentration of $\Sigma_7\text{PCB}$ in Arctic char from Bear Island was approximately one order of magnitude higher than those recorded in the Canadian lakes. However, one remote Canadian lake which is known for high concentrations of organochlorines is Lake Laberge. The average concentration ($\pm$ S.D.) of ΣPCB in male lake trout from this lake was 1177 ± 1219 ng/g w.w. and 14300 ± 21760 ng/g lipid (Muir & Lockhart 1993b, a).

The average concentrations of sum PCBs (unspecified congeners) in Arctic char from Northern Sweden ranged from 700 to 3000 ng/g lipid, in fish which had 1.2-1.6% extractable lipid (Andersson *et al.* 1988). In the polluted Lake Vättern in South Sweden, muscle samples from Arctic char with a mean weight of 741 g (560-1007 g) were pooled. The recorded concentration of $\Sigma_7\text{PCB}$ was 3 612 ng/g lipid in Arctic char with 5.3% extractable lipid (Jansson *et al.* 1993).

The concentrations of ΣPCB in the muscle tissue of whitefish from Finnmark were lower than in whitefish from Canada. In whitefish from the Mackenzie River (Ft. Good Hope) the ΣPCB ranged between 2.3 and 11.2 ng/g w.w. (Lockhart *et al.* 1992). Average concentrations of ΣPCB in whitefish from three Yukon lakes were 280, 6.4 and 21 ng/g w.w. (Kidd *et al.* 1994). The ΣPCB (11 of the heavier congeners) in whitefish from the Great Slave Lake was 5.5 ng/g w.w. (Evans 1994). Concentrations of PCB in whitefish from Finnmark (< 200 ng/g lipid) were also lower than in whitefish from Northern Sweden (580 ng/g lipid). Examples of PCBs concentrations on lipid basis in whitefish from the Northwest Territories in Canada were 38 ± 6 ng/g lipid in Lake Laberge and 672 ± 122 ng/g lipid in Fisherman Lake (Muir & Lockhart 1993a).

6.2.2.2 Arctic char from Spitsbergen

Arctic char on Spitsbergen show some typical growth patterns. Land-locked populations in particular have a slow growth rate and a high longevity. One of the effects of growth of the fish on contamination levels is growth dilution (Hammar *et al.* 1993). Large, slow growing fish generally accumulate higher concentrations of organochlorines than smaller, faster-growing fish do (Schindler *et al.* 1995). The oldest fish in our material was 22 years old at the time of capture, and contained very high Σ_7 PCB concentrations of 34 ng/g w.w. and 4% lipid.

In the Lakes Linnèvann and Kongressvannet, very small fish were collected, which were 9 years old and weighed only 15 g. The pooled samples of these land-locked 'dwarf forms' contained very little lipids (0.45 and 0.98% respectively), compared to larger specimens from the same lakes (between 3.5 and 4.5% extractable lipids). The concentrations of Σ_7 PCBs in these dwarf fish were around 20 ng/g w.w. and from around 2000 to over 5000 ng/g lipid.

Spearman rank order correlations showed that there was a significant negative correlation between the PCB concentrations in Arctic char from Spitsbergen and their lipid content, weight, and growth ($p < 0.0001$; $R = \text{circa } -0.8$; here double weighting was used for pooled samples).

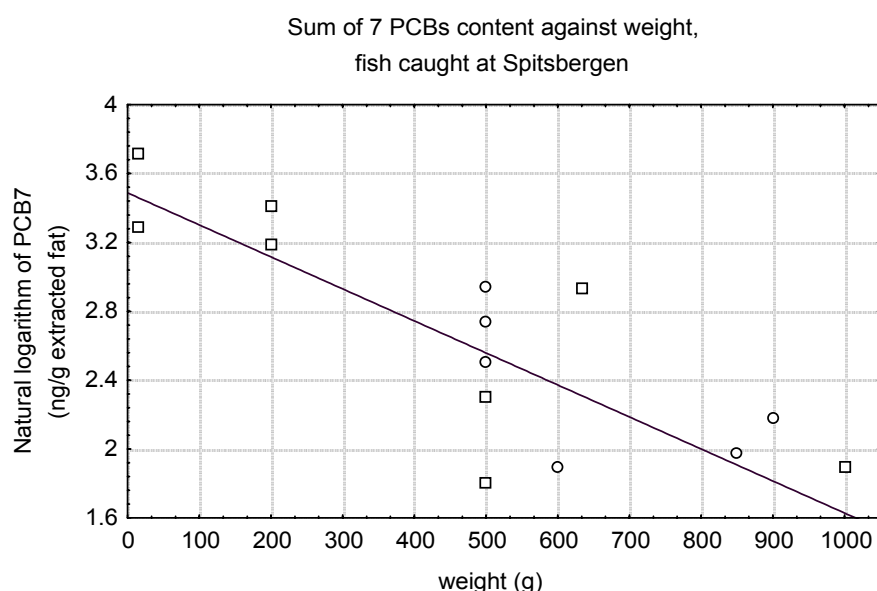


Figure 45. Concentration of Σ_7 PCB (ng/g lipid) in Arctic char from Spitsbergen versus the weight of the fish. Squares: stationary char, circles: anadromous char.

6.2.2.3 HCBz levels

The HCBz concentration in Arctic char from Bear Island was 1.3 ng/g w.w. or 45 ng/g extractable lipid. In Arctic char from Spitsbergen, the HCBz concentrations ranged from 1.1 to 6.8 ng/g w.w. and from 26 to 322 ng/g extractable lipid (Figure 46 and Figure 47). For these latter samples, the laboratory registered that the recovery was less than the required 40%. The highest recorded concentration, 6.8 ng/g w.w., was in a fish (900 g, 2.4% lipid) from Rickardvatnet. Other fish from this lake had HCBz concentrations of 1.2-1.4 ng/g w.w.

In whitefish, the minimum concentration was 0.04 ng/g w.w. and the maximum was 0.28 ng/g w.w. In perch, the concentrations ranged from 0.07 to 0.14 ng/g w.w.

Σ CBz (sum of tetra- and pentachlorobenzene and HCBz) concentrations in Arctic char from Northern Canada were between 3.1 and 5.5 ng/g w.w., in fish with 4.1 to 4.6% lipid (Muir & Lockhart 1993a). Arctic char from Finnish Lapland, with 0.4 to 2.9% lipid, had Σ CBz concentrations between 1 and 2.5 ng/g w.w. (Korhonen *et al. in prep.*). In Arctic char from Northern Sweden, the lipid based concentration, 18 ng/g lipid, was lower than the levels we recorded on the Arctic islands (M. Olsson, unpublished manuscript). In lake whitefish from Northern Sweden, the HCBz concentration, 0.09 ng/g w.w., was at a similar level as in Northern Norway (Jansson *et al.* 1993).

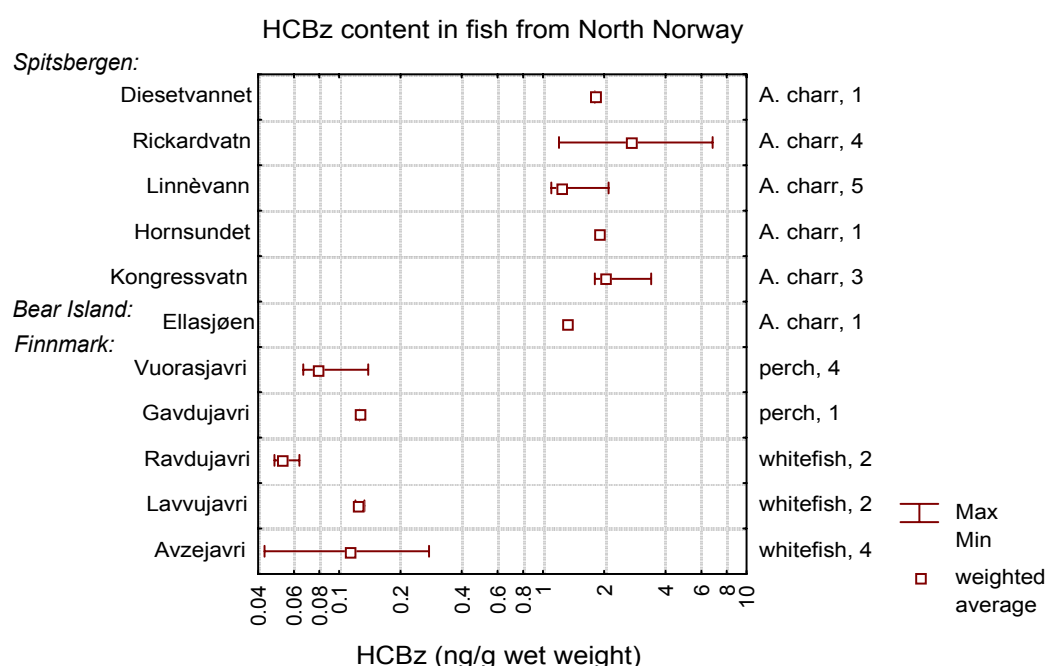


Figure 46. Minimum, maximum and average concentrations of HCBz in muscle tissue (ng/g w.w.) of fish from lakes on the Northern Norwegian mainland and a Norwegian Arctic island.

The number behind the fish species refers the number of (pooled or individual) samples from that lake, used to calculate weighted average concentrations.

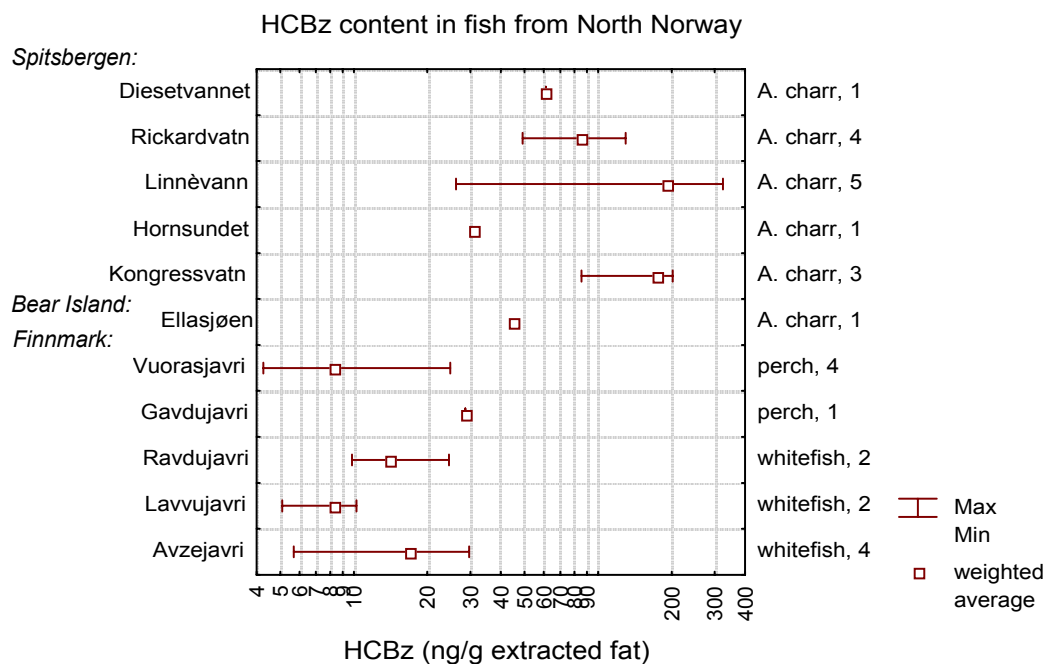


Figure 47. Minimum, maximum and average concentrations of HCBz in muscle tissue (ng/g lipid) of fish from lakes on the Northern Norwegian mainland and a Norwegian Arctic island.

Explanation of the number behind the fish species in previous figure.

6.2.3 Organochlorine pesticides

6.2.3.1 HCHs

The concentration of γ -HCH in fish from Finnmark was evenly distributed, with both perch and whitefish containing γ -HCH concentrations of around 0.05 ng/g w.w. (Figure 48). The concentration of γ -HCH in Arctic char from Bear Island was higher, at 0.18 ng/g w.w.

When γ -HCH concentrations were calculated on an extractable lipid basis, the concentrations were in the same range for all samples. Average concentrations were between 4 and 15 ng/g extracted lipid. The maximum concentration was 22 ng/g lipid. The concentration in Arctic char was 6.3 ng/g extracted lipid, slightly lower than most perch and whitefish samples.

Of the other HCH compounds, β -HCH was not detectable in any of the samples (detection limits varied, depending mainly on the amount of material, from <0.02 ng/g to <0.40 ng/g (w.w) for whitefish and perch samples and <1.24 ng/g for char from Bear Island).

The levels of α -HCH were not quantifiable in most samples, as concentrations were below 0.06 ng/g w.w. Concentrations of α -HCH which could be quantified in whitefish and perch samples were less than 0.10 ng/g. In Arctic char from Bear Island, the concentration of α -HCH was 0.40 ng/g w.w.

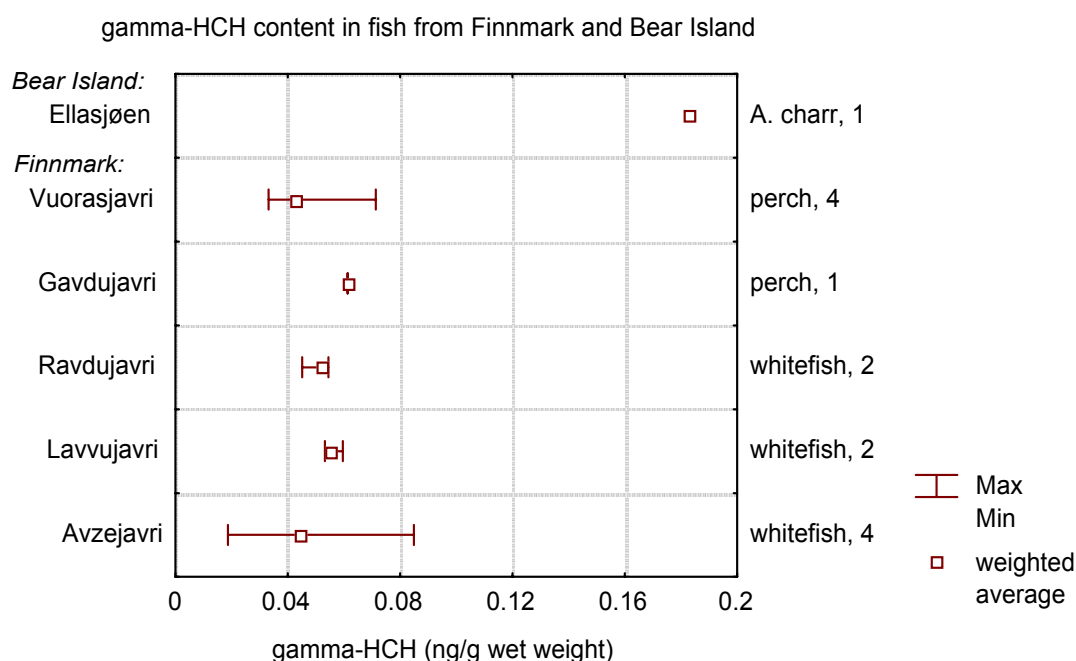


Figure 48. Minimum, maximum and average concentrations of γ -HCH (ng/g w.w.) in muscle tissue of fish from Northern Norway.

The number behind the fish species refers the number of (pooled or individual) samples from that lake, used to calculate weighted average concentrations.

Concentrations in whitefish from Northern America were generally higher than in Finnmark. Concentrations of Σ HCH in whitefish from the Mackenzie River (Ft. Good Hope) were between 0.5 and 2.0 ng/g (Lockhart *et al.* 1992). The mean concentration of Σ HCH in whitefish from the Yukon Lakes Laberge, Fox and Kusawa were 1.7, 0.7 and 0.5 ng/g w.w. respectively (N = 6, 5 and 9) (Kidd *et al.* 1994).

In lake trout from Schradar Lake in Alaska, the concentration of α -HCH + γ -HCH was 1.0 ± 0.6 ng/g w.w. (11 samples ranged from 0.2 to 2.4 ng/g) (Wilson *et al.* 1995).

The average concentration of γ -HCH in whitefish from Northern Sweden was 16 μ g/g extractable lipid (N=15) (Andersson *et al.* 1988).

6.2.3.2 DDTs

All concentrations of Σ DDT in perch and whitefish from Finnmark were below 1 ng/g w.w., with average concentrations between 0.17 and 0.6 ng/g w.w. (Figure 49). Of the DDT components and their metabolites, the largest proportion was made up of p,p'-DDE. The concentration of p,p'-DDE comprised approximately 80% of the Σ DDT concentration in perch and whitefish. The concentration of Σ DDT in Arctic char from Bear Island was two orders of magnitude higher, at 76.4 ng/g w.w. This extremely high concentration was also confirmed by reanalyses. p,p'-DDE made up about 97% of the Σ DDT concentration.

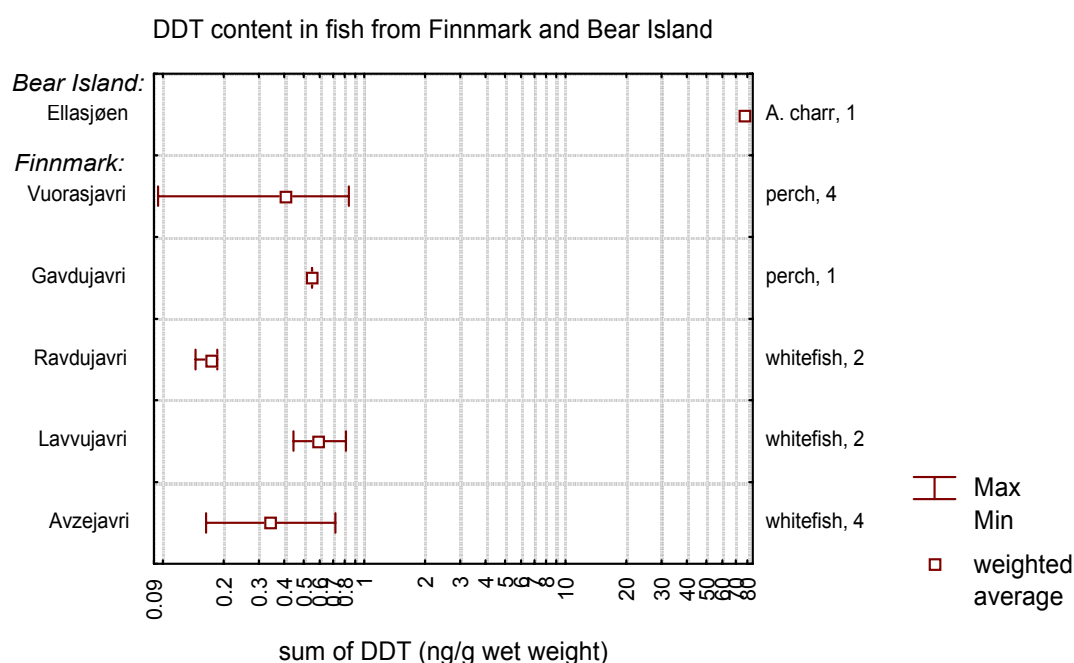


Figure 49. Minimum, maximum and average concentrations of Σ DDT (ng/g w.w.) in muscle tissue of fish from Northern Norway.

The number behind the fish species was explained in Figure 48.

In whitefish, concentrations of Σ DDT on a lipid basis were all below 90 ng/g lipid (Figure 50). The highest concentrations in Finnmark were in perch from Gavdujavri, 125 ng/g lipid. Arctic char from Bear Island had a Σ DDT concentration of 2 634 ng/g lipid.

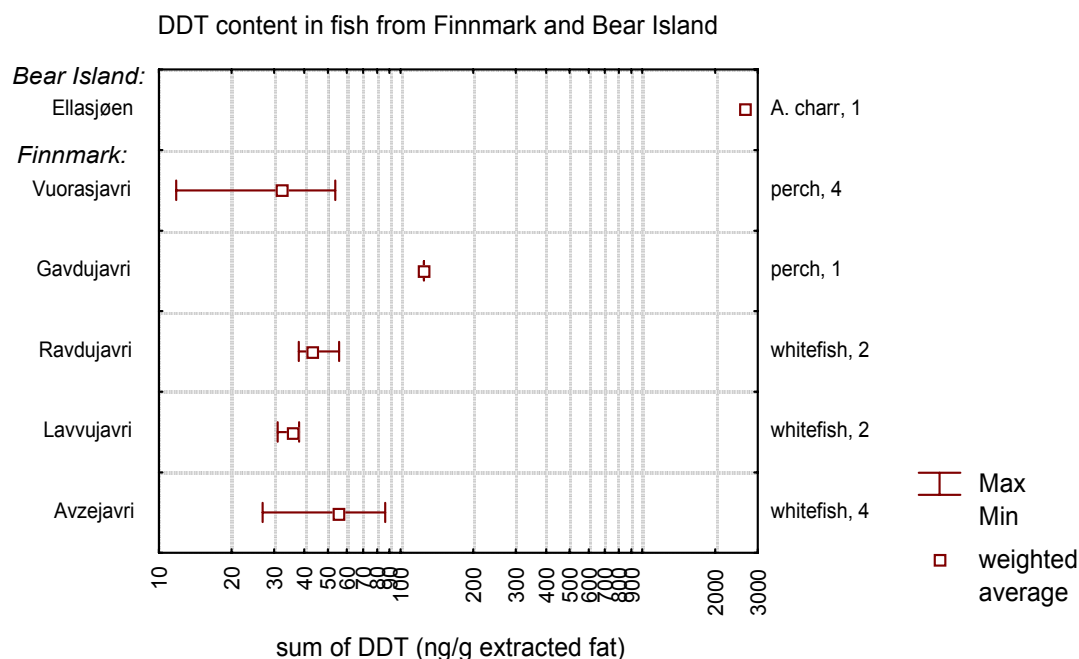


Figure 50. Minimum, maximum and average concentrations of Σ DDT (ng/g extractable lipid) in muscle tissue of fish from Northern Norway.

The concentrations of Σ DDT in whitefish from the Mackenzie River were between 0.2 and 3.6 ng/g w.w., which is higher than those in our samples from Finnmark. In three Yukon lakes, average concentrations of Σ DDT in whitefish were 155, 6.1 and 15 ng/g w.w. (Kidd *et al.* 1994).

In whitefish (N=15) from Northern Sweden, the average concentration of Σ DDT was 450 ng/g lipid (Jansson *et al.* 1993). The concentrations in Arctic char from Northern Sweden were between 200 and 2600 ng/g extractable lipid (Andersson *et al.* 1988).

In Canada, the average concentrations of Σ DDT in whitefish from, for example, Lake Laberge were 13 ± 9 ng/g lipid and in whitefish from Great Slave Lake 44 ± 6 ng/g lipid (Muir & Lockhart 1993a).

Concentrations of Σ DDT in Arctic char from the Lakes Buchanan, Amituk and Char Lake in the North West Territories, Canada, were 3.7 ± 0.9 , 31.2 ± 14.9 and 114 ± 52.9 ng/g w.w. respectively (Muir & Lockhart 1993a). Concentrations of DDE in lake trout from Schrader Lake, Alaska, ranged from 0.8 to 11.4 ng/g w.w. (Wilson *et al.* 1995). On a lipid basis, the concentration of Σ DDT in lake trout from Amituk Lake was 1870 ± 3150 ng/g lipid (Muir & Lockhart 1993a).

6.2.3.3 Chlordanes

The total concentrations of chlordane-related compounds (Σ chlordanes) were below 0.5 ng/g w.w. for all samples from lakes on the mainland (Figure 51). The average concentrations per lake were between 0.035 and 0.2 ng/g w.w. Concentrations in perch and whitefish were in the same range. The concentration of Σ chlordanes in Arctic char from Bear Island was more than an order of magnitude higher, at 4.1 ng/g w.w.

The individual compounds which contributed most to the concentrations of Σ chlordanes were *trans*- and *cis*-nonachlor, followed by oxychlordane and *cis*-chlordane. Chlordane, heptachlor

and heptachlor epoxide were not detectable in the samples. In Arctic char from Bear Island, oxychlordanes and *trans*-nonachlor were the major contributors to Σ chlordanes.

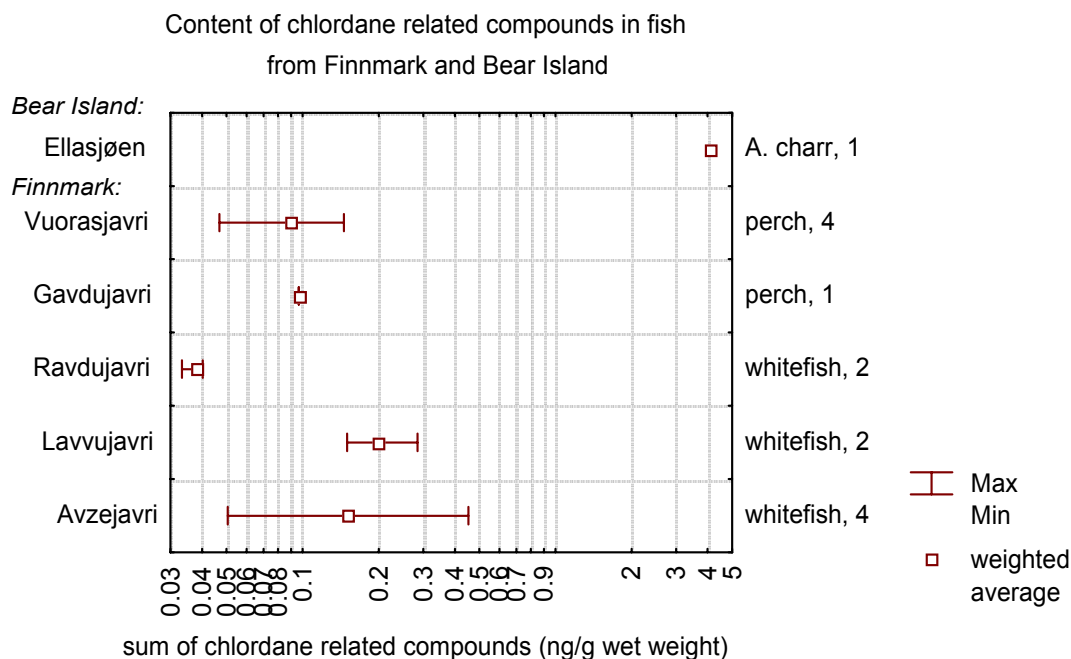


Figure 51. Minimum, maximum and average concentrations of Σ chlordanes (ng/g w.w.) in muscle tissue of fish from Northern Norway

The concentrations of chlordanes related compounds on a lipid basis had the same pattern as the distribution on wet weight base. All measured concentrations in perch and whitefish were below 50 ng/g (lipid) and averages were below 25 ng/g (Figure 52). On a lipid basis, char from Bear Island also contained by far the highest concentration of Σ chlordanes, 142 ng/g.

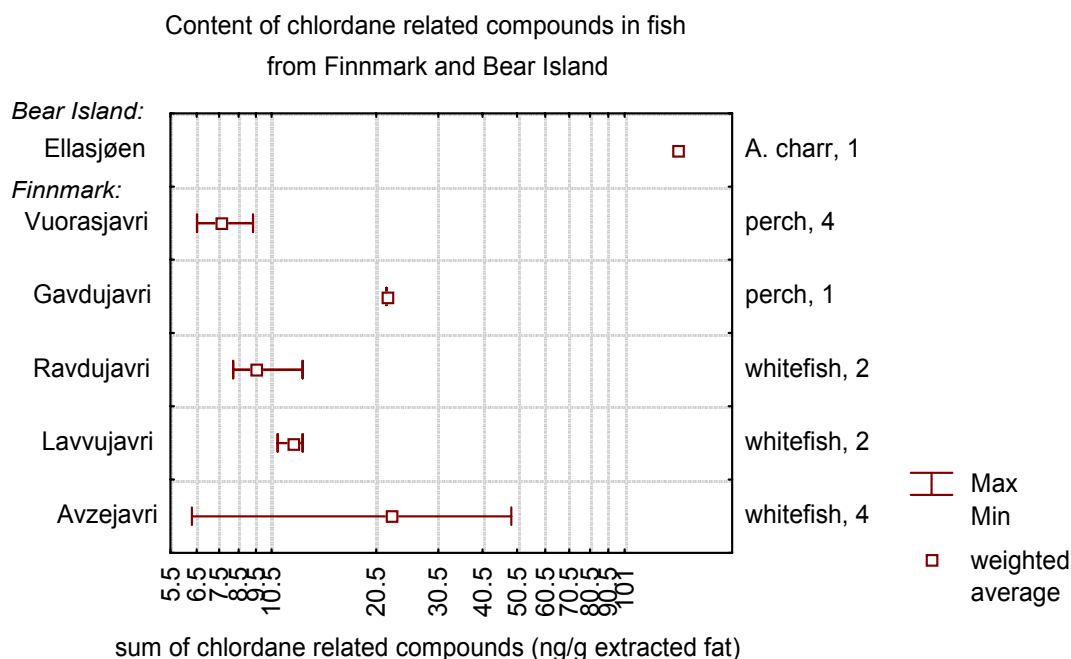


Figure 52. Minimum, maximum and average concentrations of Σ chlordanes (ng/g extracted lipid) in muscle tissue of fish from Northern Norway.

Explanation of the number behind the fish species in Figure 48.

The sum of chlordanes-related compounds in whitefish from the Mackenzie River (Ft. Good Hope) were between 0.2 and 4.8 ng/g w.w. (Lockhart *et al.* 1992). This is partly higher than in Finnmark. The concentration of Σ chlordanes in whitefish from Lakes Laberge and Lutselk'e were $10.9 (\pm 5.7)$ and $4.2 (\pm 2.6)$ ng/g w.w. respectively (Muir & Lockhart 1993b; Kidd *et al.* 1994).

The concentrations of Σ chlordanes in Arctic char from the North West Territories were higher than on Bear Island, 10.3 ± 3.2 w.w. in Buchanan Lake, 43.8 ± 10.3 w.w. in Amituk Lake and 17.8 ± 7.9 w.w. in Char Lake (Muir & Lockhart 1993a).

6.2.3.4 Other pesticides

Dieldrin could not be detected in all whitefish and perch samples, but where dieldrin was detectable, the concentrations in the two species were in the same range, ranging from 0.02 to 0.07 ng/g w.w. However, in most samples, the recorded value deviated more than 20% from the theoretical value, due to interference or instrument noise. The recorded concentration of dieldrin in Arctic char from Bear Island was 0.43 ng/g w.w., but some interference may also have been present.

Concentrations of dieldrin in whitefish from the Mackenzie River were one order of magnitude higher than those found on Bear Island, ranging from 0.1 to 0.6 ng/g w.w. (Lockhart *et al.* 1992). In Lake Laberge, the average concentration in whitefish was $0.3 (\pm 0.2)$ ng/g w.w. (Muir & Lockhart 1993b).

The pesticides aldrin, endrin and trifluralin were not quantifiable in any of the samples analysed. Endosulfan was quantified in all samples, with concentrations ranging from 0.003 to 0.03 ng/g w.w.

6.2.4 Polycyclic aromatic hydrocarbons

In the fish samples, only the most volatile PAHs were detected, from naphthalene to pyrene (Table 4). The sum of the quantified PAHs varied from 4.4 to 13.7 ng/g. Concentrations of individual components in whitefish, perch and Arctic char were in the same range.

The components with the highest detected concentrations were as follows: naphthalene (1.4 to 4.7 ng/g w.w.), 2-methyl naphthalene (0.25 to 1.4 ng/g), 1-methyl naphthalene (0.19 to 1.3 ng/g), and phenanthrene (0.3 to 2.3 ng/g). The heavier PAHs, which include the carcinogenic compounds, could not be detected in the samples available. These PAHs, from benzo(a)-fluorene to coronene, were present in concentrations below 0.1 ng/g w.w.

Due to low concentrations of PAHs, and the restricted quantity of material, the recorded concentrations did not fulfil the quality criteria for these analyses. The measurements were below the required concentration of ten times the concentration in the blind sample. Thus, the uncertainty of the analyses is higher than 15%.

In general, the more volatile PAHs are detected in sediment and in fresh water fish (muscle and liver) (Lockhart *et al.* 1992). Fish metabolise PAHs readily and there may be large variations in concentrations between individual fish (Lockhart *et al.* 1992). A more appropriate measure for PAH contamination of fish may therefore be analyses of metabolites of PAH in the bile.

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Appendix 1. Primary data on the lakes included in this investigation.

COUNTY/ ISLAND		Map No.	Longitude	Latitude	Depth	Elevation	Lake	Catchment	Bog	Forest	Year	Sediment		Fish	
No.	Lake name				m	m.a.s.l.	area km ²	area km ²	area %	area %		H.M.	POP/PAH	Hg	POP/PAH
SPITSBERGEN															
87	Revvatn	B12	15.370	77.200		30	0.81	14.24		0	92	X			
85	Istjørna	C9	15.100	78.027		235	0.32	0.97		0	92	X			
86	Vann 210	C9	15.225	78.003		210	0.04	0.38		0	92	X			
90	Barentsvann	E8	21.180	78.579		100	0.37	1.94		0	94	X	X		
88	Kongressvann	B9	13.965	78.010		95	0.5	4.85		0	94	X			X(PCBs)
89	Linnèvann	B9	13.810	78.050		10	3.54	38.84		0	94	X			X(PCBs)
93	Rickardvannet	A4	12.330	79.412			3.04	13.03		0	92				X(PCBs)
94	Hornsundet	B12 + C13	16.000	77.000		0				0	92				X(PCBs)
95	Diesetvannet	A6	11.319	79.218		15	3.77	35.05		0	92				X(PCBs)
BEAR ISLAND															
91	Ellasjøen	D20	18.920	74.460	43	21	0.69	3.25			94	X	X	X	X
FINNMARK															
1	Storvatnet	1835 II	22.925	70.025	39	30	1.21	138.3	1	15	92/95	X	X		
2	Hestekovatnet	1835 III	22.596	70.178	40	354	0.36	2.5	0	0	92	X			
3	Storvatnet	1934 IV	23.469	69.891	50	94	1	13.9	10	86	92	X			
4	Doggejav`ri	1935 I	24.540	70.375	14	285	2.75	14.7	8	0	92	X			
5	Ø.Saltvatn	1935 IV	23.998	70.422		274	0.75	32.1	0	0	92	X			
6	Hamnevatn	2036 IV	24.931	70.864	22	25	0.63	51	0	0	92	X			
7	L. Havvatn	2036 III	24.935	70.693		324	0.69	8.9	0	0	92	X			
8	Lafjordvatnet	2036 I	25.463	70.772	20	149	0.81	55.7	0	0	92	X			
9	Kaifjordvatnet	2236 I	28.484	70.937	35	31	1.5	85.2	3	0	92	X			
10	Skogsfjordvatnet	2236 II + III	27.631	70.686	26	377	0.67	3.3	0	0	92	X			
11	Gussajav`ri	2235 IV	27.070	70.453	10	56	0.31	6.9	6.7	47	92	X			
12	Cullujav`ri	2135 IV	26.178	70.315	15	132	1.5	93.2	4	50	92	X			
13	Kjæsvatnet	2136 III	26.164	70.591	37	47	4.25	66	4	40	92	X			
14	Vasavatnet	2435 III	29.978	70.169	5	135	0.94	23.4	2.7	0	92	X			
15	Syltevikvatnet	2436 I	30.396	70.537	10	21	0.75	12.5	7.7	0	92/95	X	X		
16	Oardujav`ri	2335 I	29.231	70.473	15	225	1.94	37.7	5.2	0	92	X			
17	Hergevatn	2336 II	29.249	70.714	22	64	0.21	5.37	2.4	0	92	X			
96	Magistervatn	2336 II	29.186	70.529	26						95		X		
18	Store Kløftvatnet	2336 II	29.085	70.651	15	168	0.62	5.6	2.5	0	92	X			
19	Diergejav`ri	2334 IV	28.543	69.964	10	172	1.62	135.6	5.2	60	92	X			
20	Gålgutjav`ri	2235 I	28.002	70.347	12	95	1	7.5	24	60	92	X	X		

COUNTY/ ISLAND		Map No.	Longitude	Latitude	Depth	Eleva- tion	Lake area	Catchment area	Bog area	Forest area	Year	Sediment analysed for		Fish analysed for	
No.	Lake name				m	m.a.s.l.	km ²	km ²	%	%		H.M.	POP/PAH	Hg	POP/PAH
21	Maskejav`ri	2235 II	27.872	70.175	15	61	1.12	152.9	18	80	92	X			
22	Baisjav`ri	2135 II	26.777	70.023		284	0.98	30.3	3	14	92	X			
23	Suolujav`ri	2235 I	28.002	70.409		85	0.94	9.4	4.4	50	92	X			
24	Lævvajav`ri	2134 IV	26.008	69.883		434	0.75	16.3	8.9	0	92	X			
25	Stourrajav`ri	2134 III	25.889	69.703	23	114					92	X			
26	Stoppuluobbal	1934 II	24.373	69.576	15	384					92	X			
27	Doulbmajav`ri	1934 II	24.587	69.525		425	0.63	3.6	12.5	0	92	X			
28	Oksevatnet	2535 IV	30.855	70.347	12	143	2.75	9.0	2	0	92	X			
29	Kibergvatnet	2535 IV	30.989	70.310		42	0.25	3.2	38	0	92	X			
30	Vuovddejav`ri	1934 III	23.435	69.582	18	418	0.49	57.4	6	2.4	92/95	X		X	
31	Cuolbmajav`ri	1833 IV	22.773	69.344	8	548	0.63	5.1	0	0	92/95	X		X	
32	Vuorasjav`ri	1832 I/ 1833 II	23.204	68.984	8	345	2.55	41.9	9.3	50	92/95	X	X	X	X
33	Avzejav`ri	1832 IV	23.351	68.912	23	361	1.63	166.4	30	30	92/95	X	X	X	X
34	Lavvujav`ri	1932 IV	23.655	68.951	6	476	1.47	29.1	15.5	0	92/95	X	X	X	X
35	Guotkojav`ri	1933 I	24.081	69.455		425	0.81	3.2	7.8	27.5	92/95	X		X	
36	S.Galdinjav`ri	1932 IV	23.434	68.850		396	0.52	2.3	19.1	70	92/95	X		X	
37	Ravdujav`ri	2032 III	24.693	68.680		432	0.78	8.4	4	14	92/95	X	X	X	X
38	Gavdujav`ri		24.632	68.742		441	2.75	53.4	15	0	92/95	X	X	X	X
39	Røyvatn	2034 IV	25.053	69.919		149	0.04	0.1	0	70	92	X			
40	St.Ingasjarvi	2034 IV	25.065	69.901		133	0.66	7.4	2	70	92	X			
41	Vouddajav`ri	2035 III	24.819	70.012		291	1.2	14	5	3	92/95	X	X		
42	Vuostamusjav`ri	2334 I	29.037	69.881	12	230	1.5	46	3	2	92	X			
43	Gædgesuolojav`ri	2334 I + II	28.889	69.742	10	131	2	5	25	51	92	X			
44	Korpvatn	2534 III	30.862	69.590		197	1	24	16	14	92	X			
45	Gardsjøen	2534 III	30.855	69.714		82	0.6	9	3	17	92	X			
46	Hundvatn	2434 II	30.541	69.566		171	0.5	14.4	16	70	92	X			
47	Ødevatn	2333 II	29.001	69.061		88	2.7	28.9	22	68	92	X			
48	Ellevatn	2333 II	28.908	69.138		120	11	50	10	68	92	X			
49	Følvatn	2333 I	28.928	69.256		177	2.3	11	10	75	92	X			
50	St.Spurrvatn	2333 I	29.186	69.327		164	4.8	21.1	7.7	65	92	X			
51	St.Sametjern	2433 IV/ 2434 III	29.640	69.470		96	7.4	76	15	40	92	X			
52	Coalbmejv`ri	2434 I	30.546	69.787		221	0.16	1.7	0	0	92	X			
53	L.Ropelvvatn	2434 I + II	30.188	69.765		52	1.1	47.4	4	30	92/95	X	X		
54	Vegvatnet	2334 II	29.264	69.660		101	0.3	1.6	0	15	92	X			

Appendix I. Continued

COUNTY/ ISLAND		Map No.	Longitude	Latitude	Depth	Eleva- tion	Lake area	Catchment area	Bog area	Forest area	Year	Sediment analysed for		Fish analysed for	
No.	Lake name				m	m.a.s.l.	km ²	km ²	%	%		H.M.	POP/PAH	Hg	POP/PAH
55	Gravsjøen	2434 II	30.609	69.735		118	1.5	14.5	0.5	10	92	X			
56	Langvatnet	2434 II	30.554	69.721		87	0.8	15.2	0	11.5	92	X	X		
57	Rabbvatnet	2434 II	30.463	69.663		83	0.3	1.6	9	85	92/95	X	X		
97	Haukesjøen	2334 I	29.260	69.917	31	31					92/95		X		
58	Holmevatnet	2434 III	29.716	69.663		241	0.06	0.4	0	0	92	X			
59	Vaggatem	2333 I	29.269	69.275		51					92	X			
60	Bjørnevatn	2434 IV	29.453	69.799		98	2.1	7.1	3.5	50	92	X			
61	Midtre Pumpehusvatn	2434 III	29.856	69.711		88	0.3	3.4	2	0	95	X			
TROMS															
62	Svarthammervatnet	1534 III	18.802	69.723	20	390	0.15	1.4	0	0	93/94	X			
63	Y.Kårviksvatnet	1534 IV	18.873	69.896	26	107	0.3	4.4	31	20	93/94	X			
64	Storvatnet	1533 II	19.325	69.165	60	128	1.6	15.4	0.1	35	93/94	X	X		
65	Josvatnet	1734 III	21.262	69.569	36	72	0.5	26.5	0.1	10	93/94	X	X		
66	Langfjordvatn	1635 II	20.536	70.154	36	65	0.5	1.3	0	0	93/94	X	X		
67	Tårnvatn	1433 I	18.362	69.310	62	108	3.7	27.5	60	10	93/94	X			
68	Ø. Kaperdalsvatn	1333 I	17.386	69.280	33	199	0.5	21.2	0	4	93/94	X	X		
69	Skøvatnet	1433 III	17.888	69.033	110	180	6.1	40.1	7	25	93/94	X	X		
NORDLAND															
70	Langvatnet	1333 I/ 1332 III	16.690	68.501	16	16	0.8	49.6	15	55	93/94	X	X		
71	Skallavatnet	1332 II	17.310	68.513	22	164	0.3	43.1	20	60	93/94	X			
72	Holmevatn	1331 IV	16.658	68.375	13	373	0.15	0.5	9	36	93/94	X	X		
73	Vikvatnet	1031 II	13.579	68.200	32	10	0.5	4.7	0	5	93/94	X			
74	Storvatnet	1131 I	14.544	68.321	25	17	1	3.3	4	42	93/94	X	X		
75	Ryggedalsvatnet	1132 I	14.785	68.759	72	22	0.8	8.4	1	3	93/94	X			
76	Storevatnet (Bleikvt.)	1233 II	15.961	69.250	25	28	1.7	8.3	4	34	93/94	X	X		
77	Ågevatn	1928 III	13.466	66.706	17.5	23	0.2	4.5	2	56	93/94	X			
78	Grønåsvatn	1928 III	13.674	66.750	37.5	83	1	6	2	62	93/94	X			
79	Markvatn	1928 I	13.748	66.911	30	26	2.2	12.2	0	36	93/94	X			
80	Storvikvatnet	1928 I	13.851	66.960	40	183	0.7	2.78	1	33	93/94	X			
81	Valnesvatnet	2029 III	14.450	67.165	92	121	4.6	59.9	8	40	93/94	X			
82	Nordlivatn	2029 I	14.774	67.422	30	55	0.12	13.9	4	30	93/94	X			
83	Kilevatn	1231 II	15.967	68.130	65	94	5.9	19.1	20	60	93/94	X			
84	Tennvatn	2130 I	15.961	67.762	54	339	1.0	4.3	4	71	93/94	X			

Appendix

Comparison of regression coefficients and significance level:

	P		C		B			
	Full datamatrix		unweighted groups		Dobble weighting		N as weight	
	r	p	r	p	r	p	r	p
GROWTH	-0.69	< .001	-0.74	< .001	-0.74	< .001	-0.79	< .001
WEIGHT	-0.68	< .001	-0.68	0.0017	-0.71	< .001	-0.77	< .001
LENGTH	-0.66	< .001	-0.68	0.0018	-0.71	< .001	-0.76	< .001
AGE	0.49	0.007	0.51	0.02	0.53	0.012	0.58	0.0007
CONDITION	-0.39	0.032	-0.31	0.2	-0.43	0.046	-0.45	0.013
FAT%	-0.27	0.143	-0.23	0.36	-0.32	0.15	-0.46	0.01

	D		D		E			
	Full datamatrix		unweighted groups		Dobble weighting		N as weight	
	r	p	r	p	r	p	r	p
GROWTH	-0.70	< .001	-0.74	< .001	-0.75	< .001	-0.79	< .001
WEIGHT	-0.68	< .001	-0.69	0.0015	-0.72	< .001	-0.77	< .001
LENGTH	-0.67	< .001	-0.69	0.0015	-0.71	< .001	-0.77	< .001
AGE	0.49	0.007	0.51	0.031	0.52	0.012	0.58	< .001
CONDITION	-0.40	0.03	-0.32	0.2	-0.35	0.08	-0.45	0.012
FAT%	-0.28	0.13	-0.24	0.34	-0.33	0.14	-0.47	0.009

Changes of different statistics using groups weighted with N instead of the individual values of the full data set.

	Valid N	Mean	Median	Min	Max.	Quartile	Quartile	Range	Variance	STD
LENGTH	0 %	0 %	-1 %	0 %	0 %	1 %	4 %	12 %	-11 %	-6 %
WEIGHT	0 %	0 %	-6 %	0 %	0 %	0 %	11 %	17 %	-11 %	-6 %
GROWTH	0 %	0 %	-7 %	0 %	0 %	0 %	7 %	10 %	-12 %	-6 %
FAT	0 %	0 %	2 %	0 %	-32 %	22 %	4 %	-33 %	-53 %	-32 %
PCB	0 %	0 %	0 %	278 %	0 %	0 %	-16 %	-21 %	-12 %	-6 %
PCB_FAT	0 %	-2 %	13 %	194 %	0 %	-17 %	-22 %	-23 %	-11 %	-6 %

Changes of different statistics using a reduced data set with double weighting of pooled observations compared to using the full data set.

	Valid N	Mean	Median	Min	Max.	Lower Quartile	Upper Quartile	Quartile Range	Variance	STD.
LENGTH	-27 %	-3 %	-4 %	0 %	0 %	1 %	-3 %	-14 %	-1 %	-1 %
WEIGHT	-27 %	-7 %	-13 %	0 %	0 %	-2 %	-11 %	-16 %	4 %	2 %
GROWTH	-27 %	-12 %	-30 %	0 %	0 %	-14 %	2 %	7 %	-11 %	-5 %
FAT	-27 %	-4 %	-2 %	0 %	-32 %	2 %	-3 %	-14 %	-52 %	-31 %
PCB_FAT	-27 %	15 %	15 %	278 %	0 %	29 %	0 %	-8 %	2 %	1 %
PCB	-27 %	16 %	52 %	194 %	0 %	8 %	-8 %	-12 %	5 %	3 %

The original data from the laboratory analyses for contaminants are given in two separate appendix reports.

Appendix report A: "Heavy metals, POPs and PAH in sediment from lakes in Northern and Arctic regions of Norway. Analytical results from laboratory reports."

Appendix report B: "Mercury, POPs and PAH in fish from lakes in Northern and Arctic regions of Norway. Analytical results from laboratory reports."