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STATUS AND POLICY ON CONTAMINATED SITES IN SWEDEN

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History

During the 70-ties and 80-ties the remediation activity in Sweden was occasional and mainly due to private or supervisory initiatives or demands in environmental permits. In the 90-ties guidelines on inventories, investigation, quality control, risk-assessment, project management, remediation technology among others were produced by the EPA. This is the same time as many other European countries started to more actively work with these issues. The small national funds made a preliminary inventory and a few pilot-sites for remediation possible.

In 1999 the new environmental law, the Environmental Code, with the special chapter 10 for old polluted sites entered into force. The same year the Environmental Quality Objectives were launched by the parliament. One of the objectives, A non-toxic environment, stipulates that the environment must be free from man-made or extracted compounds and metals that represent a threat to human health or the biological diversity within one generation. Following this development the EPA got increased funds for remediation of sites where no responsible party can be found. The money could not be used for salaries or administrative costs and an organisation for this task began slowly to take form within the existing structure.

Inventory and priority

The potentially contaminated sites in Sweden are now identified and the first part of the interim target to 2005 was therefore achieved. In the ongoing inventory 11 000 of these sites have been risk-classified according to the MIFO-methodology (NV-report 4918) giving an even better picture of how large the problem with contaminated sites is. Approximately 1500 of the sites belong to the highest risk-class, potentially posing very high risk to human health or the environment. The authorities are, as far as it's possible, targeting their efforts to these sites to fulfil the specified national objective to solve the main environmental problem with contaminated sites at the latest 2050.

Remediation with state funding

With governmental funding 31 sites have been remediated since 1999, 28 is now under remediation and 43 have remediation plans. Some hundred sites that the counties prioritise are in some state of investigation. These are primarily sites where no responsible party has been found and they are generally quite large and complicated. A few sites costs over 10 ME, while the mean today is 4 ME. Since the market is still stabilising, the ambition is high and we focus on sites with very high risk the prediction is that this mean will go down in time.

The government organisation

Knowledge, organisation and market have expanded during the last years largely because of the environmental objective from 1999 and the government grant of now 50 ME/year spent on orphan sites. The orphan sites are singled out, investigated and then remediated by the regional and local authorities that can apply for financial support from the EPA. Four persons at the EPA and about 100 persons at the regional level now have their main tasks within the field. An ordinance for the state grants and extended guidelines from the EPA supports this system, as well as the co-operation with other agencies like SGU (Geological Survey of Sweden), SGI (Swedish Geotechnical Institute) and ITM (Department of Applied Environmental Science).

A growing sector

The activity is also vivid outside the stately funded projects. Other government organisations are working with the sites where they have a responsibility for remediation, Spimfab has remediated hundreds old petrol stations, exploitation in the major cities is a great contributor to the total amount of remediated sites, SGU have an obligation to investigate sites polluted by a government organisation that doesn't exist today, programs and courses are arranged all over the country and Renare mark, The clean soil network, gathers hundreds of members to their annual meetings.

The legislation

The 10. chapter in the Environmental Code is applicable to land, water areas, buildings and structures that are so polluted that they may cause a damage or detriment to human health or the environment. Also the general rules of consideration in the 2. chapter, the chapter on environmentally hazardous activities and health protection and the 26. chapter on supervision are frequently used when handling contaminated sites. A very simplified summary of the 10. chapter would say that it identifies the polluter or polluters liable for remediation if the activity was on-going after 30. June 1969. When the extent of the liability is assessed several factors should be taken into account, like the time since the contamination occurred, if the responsibilities were accomplished at that time and if several operators contributed to the contamination. If the polluter can't be held liable, demands can in some cases be directed to the land-owner These cases may be if the property was acquired after 1999, if there is a leaking storage on the property or if the value of the property rises as a result of the remediation. The clean-up insurance in the 33. chapter should finance remediation on sites where the polluter has no financial means. The insurance does however not meet the expectations.

Supervision of contaminated sites

The remediation work is still in an expanding phase and measures to render more effectiveness are taken concurrently. One identified bottleneck is the still largely untested 10 chapter of the Environmental Code. About one third of the sites that might pose a very high risk to human health or the environment are assumed to have a liable party or parties and one third have a partially liable party. Since the government funds can't finance these sites other measures have to be taken reach the target. Since 2005 the government grant can also be used at the county administrations for supervisory efforts to ensure that remedial work is carried out at contaminated sites for which responsible

operators can be identified and state grants cannot be made. This will probably, in time, generate court decisions clarifying the extent of liability for remediation according to the law and speed up the work.

Ongoing work at the EPA

The knowledge program Hållbar sanering, Sustainable Remediation, finance over 50 projects collecting and preparing information from within and outside the country. Some may be the foundation for further work on new or existing guidelines. The first reports can already be found on the internet. The EPA just took another step towards updating the guidelines on risk-assessment and are now also aiming to better connect the assessment to the so called risk-valuation, the remediation goals and targets. If the timetable holds the material will be remitted externally in the later half of next year. At about 60-70 % of the prioritised sites the supervising authority is the local environmental board. More efforts are now canalised into knowledge transfer and networks to help the municipalities primarily in the work of supervision and special planning.

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STATUS OF AND POLICY ON CONTAMINATED SITES IN FINLAND

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1. Contaminated sites in Finland

There are about 20 000 sites in the Finnish soil inventory. Of these, around 16 500 are potentially contaminated – based on knowledge of earlier or ongoing activities on these sites. The rest are either known to be contaminated or have already been cleaned up. About half of the sites in the inventory are located less than 100 metres from residential, groundwater or surface water areas, or less than 200 metres from a source of water supply.

The inventory was conducted for the first time in the early 1990's and it was updated during 1998 and 1999. A national database system called MATTI on the quality state of soil has now been created. This will help improve the handling of contaminated sites and dissemination of information between, for instance, landowners and local and regional authorities. MATTI will be taken into full-scale operation next year after the checking of information is completed.

Contamination of soils is, in most cases, a result of careless use of oil and chemicals or of mere ignorance. In Finland, typical activities or establishments that have caused soil contamination have been the distribution and storage of fuels, sawmills, impregnation plants, various industries, depots and garages, greenhouses and shooting ranges. Moreover, soil material of mixed origin and even waste material have been used for the filling of large areas, mainly along shores.

The most common harmful substances causing soil contamination are heavy metals (e.g. lead and arsenic), oil products, polyaromatic hydrocarbons, polychlorinated biphenyls, chlorophenols, and pesticides.

2. Legislation

In the following there is a short, but not comprehensive, description of the Finnish legislation concerning contaminated soil and groundwater, with some linkages to existing and upcoming EU legislation. Overall, the effects of the future Soil Framework Directive (SFD) are still in the air. More will be known when the Commission finally gives the proposal on the Thematic Strategy on Soil Protection, which, according to the latest information, should be this autumn.

2.1 Environmental Protection Act (EPA)

Finland has no special legislation on soil protection or contaminated sites. These matters fall under the Environmental Protection Act (86/2000), which came into force in 2000. Before that, problems related to soil contamination were addressed under the waste and waste management legislation, but these were quite inadequate in their scope concerning soil contamination.

The central sections in the EPA are:

- Soil pollution prohibition (§ 7)
- Groundwater pollution prohibition (§ 8)
- Duty to treat soil and groundwater (§ 75)
- Duty to notify (§ 76)
- Duty to investigate (§ 77)
- Restoration of soil (§ 78)
- Ordering restoration (§ 79)
- Reporting duty concerning a polluted area (§ 104)

According to the EPA it is forbidden to pollute soil or groundwater. Under the law, any party whose activities have caused pollution of soil or groundwater is required to restore the soil or groundwater to a condition that will cause no harm to health and the environment or represent a hazard to the environment.

Depending on when the contamination occurred, the liability to remediate falls under different regulations. The EPA is applied only when soil contamination has taken place after 1 January 1994. According to this act, either the polluter or the holder of the property is responsible for remediation. If the property holder of the polluted area or real estate is not able to treat the polluted soil, the local authority (municipality) has to establish the need for soil treatment and carry out the work itself. Responsibility for old damages is insufficiently regulated in the old laws and the decisions have to be partly based on case law. Waste disposal sites that were closed down before 1 January 1994 or contamination that took place before that date are regulated by the old Waste Management Act (WMA). According to this act, the party with primary responsibility for cleaning up the site is the polluter and, secondarily, the owner or holder of the property. In cases where the polluter or owner has neglected his or her obligations, municipalities have sometimes been in charge of the remediation, together with the state.

Anyone who sees that contamination has occurred, for example, during a construction project, must report it to the local environmental or health authority or the responsible regional environment centre, and, if it poses an immediate emergency, also to the fire and rescue authority. The regional environment centre can order the party responsible for clean-up to investigate and clean up the site.

The environmental authority must approve treatment or removal of contaminated soil or treatment of groundwater. This approval is given in a notification or environmental permit decision. Notification is used instead of a permit in more than 90% of cases. Targets of the remediation are sustained in the decision. The competent permit and supervisory authority is the regional environment centre and in the city of Helsinki the local environmental authority.

If someone is selling or renting a polluted site, he must tell the new owner or tenant what kind of activities have been carried out on the site and if any pollution might have occurred. If this obligation is neglected, the buyer has the right to demand that the agreement be cancelled, that the price be lowered or that the seller covers the damages.

2.2 Contaminated soil and groundwater

2.2.1 Soil

There is no exact definition of contaminated soil. According to the soil pollution prohibition in the EPA, the contamination of soil is related to the effects, and not to the concentration of the harmful substances. This seems to be in line with preliminary proposals for the future SFD.

Soil pollution prohibition (EPA § 7): Waste or other substances shall not be left or discharged on the ground or in the soil so as to result in such deterioration of soil quality as may endanger or harm health or the environment, substantially impair the amenity of the site or cause comparable violation of the public or private good.

Old generic guidelines (the so-called SAMASE guidelines and limit values) have been commonly used as planning and decision-making tools. Risk-based land management has been possible, but it has not been used widely.

The upcoming and long-awaited Government Decree on the Assessment of Soil Contamination and Remediation Needs will emphasize risk assessment. The assessment of soil contamination and the remediation need is based on the effects of harmful substances on health and the environment. Some level of case-specified assessment is always required. Soil quality guidelines (upper and lower guideline values) for about 50 substances or substance groups will be included to help in the assessment procedure. In all likelihood, an intervention value will also be included. The decree is supposed to be accepted by the Government this year, followed later by instructions.

2.2.2 Groundwater

Finland also has no exact definition of contaminated groundwater. The groundwater pollution prohibition of the EPA is a bit complex in its interpretations – it is, at the same time, both absolute and relative.

Groundwater pollution prohibition (EPA § 8)

A substance shall not be deposited in or energy conducted to a place or handled in a way that:

- 1) groundwater may become hazardous to health or its quality otherwise materially deteriorate in areas important to water supply or otherwise suitable for such use;
- 2) groundwater on the property of another may become hazardous to health or otherwise unsuitable for usage; or
- 3) the said action may otherwise violate the public or private good by affecting the quality of groundwater.

The action referred to above in paragraph 1 is also deemed to include action separately prescribed upon by decree and the discharge into groundwater of substances hazardous to the environment and health as prohibited by decree. A decree may only pertain to action referred to in the relevant directive of the European Community.

At the moment, groundwater contamination is mainly assessed by using drinking water criteria, either national criteria (Ministry of Social Affairs and Health, 461/2000) or sometimes WHO criteria. The national criteria are based on the principals of prevention and therefore the values are more suitable for use than the mainly higher WHO criteria when assessing contamination. The groundwater classification has an important role in risk assessment. The remediation targets are usually stricter in the areas important for water supply or suitable for such use than in other areas.

The Water Framework Directive (WFD) and the upcoming Groundwater Daughter Directive (GWDD) set three main objectives for groundwater: to achieve a good status of groundwater, to prevent or limit the inputs of pollutants and to reverse significant and sustained upward trends in the concentration of pollutants. Quality standards will be set only for nitrates and pesticides in the GWDD and they will be based on existing EU directives. The threshold values will be left for member states to define, either nationally or even for groundwater body level. A harmonized methodology for defining the threshold values is being developed in a research project (BRIDGE). The Finnish national threshold values for groundwater are being prepared by the Finnish Environment institute and they will be presented in a Government decree probably at the end of 2008 (depending on the timetable of the GWDD). These threshold values will be important in assessing the need for remediation of contaminated sites and, in addition, they will be used (75% of the parametric values) to indicate the starting point for implementing measures to reverse the significant and sustained upward trends.

2.3 Contaminated soil as waste

Excavated contaminated soil is waste and an environmental permit is required for its recovery and disposal (EPA § 28). Large-scale treatment facilities require environmental impact assessment according to the Environmental impact assessment Law (468/1994). Assessment process is based on EIA-directive and it has been very useful in practise. Waste management activities are regulated under the waste legislation.

According to the ECJ Judgement C-1/03, unexcavated contaminated soil is also waste. The proposed exclusion of unexcavated contaminated soil from the scope of the Waste Framework Directive is welcomed, but the wording of this exclusion has to be thought out carefully.

Excavated contaminated soil is classified either as ordinary or hazardous waste depending on the toxic qualities and the concentrations of substances in the soil.

The Council decision (2003/33/EC) establishing criteria and procedures for the acceptance of waste at landfills has been implemented just recently (1.9.2006) in Finland. This regulation is very important for the management of contaminated soils, because disposal at landfills is very common. One of the challenges is how to meet the organic carbon criteria, as some soils have a rather high content of organic matter. A typical example is contaminated topsoil from shotgun ranges.

The recently amended (No. 1195/2006) EC regulation (No. 850/2004) on persistent organic pollutants set EU limits for 14 POPs in waste (50 mg/kg, except for dioxins and furans, which are 15 µg/kg). Any waste containing organic pollutants above these concentrations must be "destroyed or irreversibly transformed" using physico-chemical treatment or incineration. This will probably have some influence also on the management of certain contaminated soils, like hot spots from old wood-impregnation plants. There is a separate element of the 2004 POPs regulation that will allow derogations from this "destroy" obligation.

2.4 Environmental damages

The Act on Compensation for Environmental Damage (737/1994) entered into force in 1995. Compensation is paid for a loss defined as an environmental damage. Damage can be caused by activities carried out in a certain area and can result from pollution of water, air or soil.

Primarily, the business that has caused the environmental damage has the liability to compensate for the damage. For this purpose the business can take environmental damage insurance. Voluntary environmental damage insurance has not been very popular because of the high costs. Obligatory environmental damage insurance ensures that compensation can be paid to those who have suffered from the damage, if the compensation cannot be recovered from the polluter. The Environmental Damage Insurance Act (81/1998) came into force in 1999. This act guarantees full compensation for environmental damage in cases where those liable for compensation are insolvent, or the liable party cannot be identified. Thus, the act creates a complementary compensation scheme for environmental damage occurring in Finland. However, the act is not retroactive. It does not cover compensation for oil spills either, because there is a specific Oil Pollution Compensation Fund from which compensation for oil spills is paid. The scheme is financed by special insurance which is compulsory for the companies whose activities cause risk to the environment. All parties holding an environmental permit are obliged to take out such insurance. Additionally, businesses that hold a permit to handle or store hazardous chemicals must have this insurance.

The Directive on Environmental Liability (2004/35/CE) is still undergoing implementation. It seems that it will result in only some minor changes in the Environmental Protection Act in the sections concerning polluted soil and groundwater treatment, as the existing legislation goes further than the Directive.

2.5 Soil contamination and land use

Soil contamination should be taken into account when planning land use and granting building permits. According to the Land Use and Building Act (132/1999), planners must be aware of any contamination wherever land use changes are planned. The health and environmental damages caused by polluted soil may hinder the granting of building permits. Local authorities are in charge of granting building permits.

3. Strategic planning

Finland has no national strategy or plan for contaminated sites, but they are included in some other environmental plans like the National Waste Plan. This plan is now undergoing revision and there has been much discussion about targets and instruments. One problem with strategic plans is always how to turn good principles into real practice. The question of whether economic drivers like taxation should be used has been raised. Legislation on the taxation of wastes disposed of at land-fill sites was set out in the 1996 Waste Tax Act. At the moment, contaminated soil is not subject to tax.

The future proposal for the SFD will probably require every member state to prepare a national remediation plan. In Finland, at this time, there is a national remediation plan which is restricted only to sites that are planned for remediation using state budget funding.

4. Remediation

The cleaning up of contaminated sites started at the beginning of the 1990s. Since then about 3000 sites have been remediated (excluding small clean-ups after oil and chemical accidents). Nowadays, work on the remediation of around 300–400 contaminated sites is started every year. Last year the number was somewhat under 300. This was probably due to the expectations and uncertainties related to the upcoming Government Decree on the Assessment of Soil Contamination and Remediation Needs.

Many activities are known to cause soil contamination. In urban areas, the need for a site investigation usually arises because of a change in land use or because of corporate or real estate transactions. Remediation is also needed when operations are closed down, harmful effects of past activities are revealed and accidents occur. One important reason for remediation of sites is to ensure groundwater quality protection in groundwater areas.

The average amount of excavated contaminated soil is 500 000 tonnes per year, varying from year to year. In recent years Finland has had some huge remediation operations. The most dramatic is the remediation of the Myllypuro landfill area in Helsinki. In the 1970s housing was built above an abandoned landfill in the Myllypuro area. During the remediation work, nine apartment buildings have been torn down. When the work is completed, the site will be turned into a park.

The most common remediation method for contaminated sites is mass exchange: the contaminated soil is replaced by clean soil. Polluted soil is predominantly treated *off site*, but *on site* and *in situ* remediation are also used. The most commonly used *in situ* technologies have been soil vapour extraction and pump and treat. Additionally, reactive wall barriers, MNA (monitored natural attenuation), phytoremediation and some other techniques have been used, but not yet on a very large scale.

A wide range of methods exist for treating contaminated soils in Finland – like bioremediation by composting, soil washing, thermal desorption, solidification or stabilization, and isolation or containment. Nowadays, there are many treatment facilities that can take on different types of contaminated soil. During the last few years, landfill disposal has been widely used. Slightly contaminated soil is usually deposited in landfills, where it is used, among other things, for the building of roads and handling areas, the coating of wastes in daily practices, or in the closing phase. Today, also more heavily contaminated soils have been disposed of in landfills – as such (after sieving) or as, for instance, stabilized material. Special landfills have been built just for contaminated soil (monofills). The utilization of contaminated soil outside waste handling areas is not common.

5. Costs

The annual costs for remediation of contaminated soils in Finland will amount to 50–70 million euros, at least during the following two decades. For the period 2005–2025, it is estimated that the total cost of remediation of contaminated soils in Finland will be 1.2 billion euros. The annual cost per citizen amounts to some 10 euros, which is close to the European average. About one-third of the remediation work is funded by public money (municipalities, the state).

6. Funding

The party causing the contamination is economically responsible for cleaning up the site, if he is still operating at the site or if he is still in existence. If the polluter cannot be identified or if he no longer exists, then the owner of the property has the responsibility to clean up the site. This can be a heavy burden.

Finland has a waste management system under which the state can take action concerning remediation, if the property owner is not able to pay and if the costs are high compared to the costs of waste management in a municipality. Under the waste management system the state can, in cooperation with municipalities or property owners, participate in the remediation work or finance, on average, 45% of the remedial actions taken. So far the system has been applied to about 300 con-

taminated sites. However, the system suffers from both a shortage of appropriate funds, whether from local authorities or the government, and it is very strict in terms of application. In recent years this state budget money has been only about 3 million euros per year.

The Oil Pollution Compensation Fund OPCF (approx. 2 million euros per year) can be used for cleaning up abandoned sites which have been contaminated by oil. The SOILI programme (since 1996), which is based on an agreement between petroleum industry businesses and public bodies, includes responsibility for the remediation of polluted decommissioned service station sites. The application period for the public part funded by the OPCF ended in 2005. To date, remedial action has been taken at 350 sites and applications for 1100 sites have been submitted to the SOILI programme.

7. Contaminated sediments and dredged material

There is no systematic inventory of sites with (possible) contaminated sediments. Some attention was paid to these during soil inventories. Finland also has no national guidelines or limit values for assessing sediment contamination. Some remediation work on contaminated sediments has been carried out in connection with contaminated land management (like cleaning up contaminated sawmills and landfills). In the future there might be a need to remediate some areas with high concentrations of organotin compounds.

The most contaminated sediments in Finland can be found in the Kymijoki river, which is highly polluted by dioxins and mercury. Kymijoki has been studied widely. The decision on how to proceed with remediation will be based on risk analysis. However, it is already clear that the possible remediation work will focus only on hot spot areas.

The issue of contaminated sediments is today more commonly raised in Finland during dredging. The disposal of contaminated dredged material has become an important environmental challenge in recent years. A permit is needed for major dredging activities and the dumping of dredged material in Finland. The Ministry of the Environment has given general instructions (Environmental Guide 117, 2004) for dredging and depositing dredged material. These instructions take into account both the recommendations of HELCOM and OSPAR and they present indicative quality criteria or action levels (level 1 and 2) for trace metals, arsenic and organic contaminants in dredged material to be used in assessing the possibility of depositing dredged material into the Baltic Sea. When the concentration is under level 1, the dredged material is clean, when it is above level 2, it is contaminated and disposal at sea is not an option unless it is the environmentally best solution. When the concentration is between level 1 and 2, the material is possibly contaminated, and this requires a case-by-case assessment. Some examples of the challenges are the vast occurrence of organotin compounds and the lack of suitable treatment and disposal facilities. Undoubtedly, much remains to be done in this field.

Contaminated soil in Iceland *status, impact and future policy*



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Introduction

Background levels of hazardous pollutants are rising alarmingly worldwide. Accumulation of several persistent pollutants are also entering the news with regular intervals, with DDT, PCBs and dioxins being best known as they can enter the food chain wherever, whenever, threatening wildlife and mankind.

In recent years, examples have made clear that contaminated sites can be found in Iceland too. Apart from some smaller cases, typically point-source oil contamination on fuel filling stations, 7 'bigger' contaminated sites have been identified so far: the Hringrás-area in Reykjavik (scrap-metal recycling), Heiðarfjall at Langanes and Stafnes near Keflavik (former Nato-supervised areas), Leirdalur east of Reykjavik (shooting range), Gufunes in Reykjavik (former landfill), the Nickel area in Keflavik (fuel storage tanks) and recently a shipyard in Reykjavik. The Hringrás case has made clear that a revision of the legal framework is desirable to determine liability for soil contamination and prevent years of ongoing legal fights, taking substantial amounts of time and resources. As there is no landfill for hazardous waste operational in Iceland today, the above mentioned cases have also made clear that a such a facility is an essential link in the chain of policy-making and (cost)effective actions needed in handling soil contamination cases.

An inventory study was carried out in Iceland in 2005 to identify, categorize and assess possibly contaminated sites, deepening our general understanding of today's situation as well as the increasing load of pollutants on Icelandic habitats and ecosystems.

Current situation

The study indicated that quite many sites in Iceland may be polluted with a wide range of pollutants. This was, in fact, not at all expected, though some sites had already been identified through the years. The table herebelow shows the number of potentially contaminated sites in Iceland, their typical contaminants and the estimated percentage of the total of (possibly) contaminated soil within the country.

Type of activity	Number of sites	Possible contaminants	Estimate of total potentially surface contaminated(%)
Fuel filling stations	> 200	Fuel, oil, PAH, BTEX	15
Fuel storage tanks > cap. 5.000 tons/year	Ca. 40	Fuel, oil, PAH, BTEX	10
Private oil tanks	?	Fuel, oil, PAH, BTEX	?
Scrap-metal recycling	3	Fuel, oil, PCB, heavy metals, PAH	1
Shipyards, harbour sediments	50	Oil, PCB, TBT, PAH, heavy metals	6
(Former) military sites (NATO)	5	Heavy metals, PCB	2
(Former) landfills for municipal waste	60+	All possible contaminants	40
Quarrying	> 2000	Oil, heavy metals	10
Shooting ranges	5	Pb, As	2
Others (electric transformers, smaller companies etc.)	?	All possible contaminants	10
Metal production industry	3	Acid, metals, oil	4
Dry cleaning	?	AOX	?

Legal framework in Iceland

In 2002 the Icelandic EPA finalized a draft-regulation on soil contamination in Iceland. The draft-regulation is based on a systematical approach, describing the responsibility and interference of different parties involved on different authorial levels, risk-assessment methodologies and suggested limit-values for remediation in relation to (future) land-use. In the action chain of events, a distinguishing is made between hazards by acute soil pollution (accidents) and long-term pollution.

Some general preliminary conclusions can be made after this preliminary work:

- The subject of liability is not properly regulated, as the current legal framework is not sufficiently supporting the 'polluter pays principle', i.e. if the polluter is known. Therefore a revision of the legal framework is needed to settle liability issues and to make it possible to distinguish between "old" and "new" cases, relevant for liability determination and costs
- There is a need for determination, classification and registration of both (possibly) polluted and sensitive areas
- Zero-options (do nothing) could be *feasible* in some cases and even *desirable* in others in the scope of minimisation of environmental impacts
- In what extent is the landowner and/or polluter allowed to take part in remediation activities?
- A working group on determining limit-values is needed to reach consensus on a national level
- How to address prevention? (Insurance, funds, registration on a regional and national level, clean-soil certification etc.)

Furthermore, it has to be assessed how the current legal framework compares to other legislation, e.g. waterframe-directive that has to be implemented in icelandic legislation (ground- and surface water), allowed daily intake og different substances, desired background levels etc. Furthermore, it is clear that the ongoing work on the thematic strategy on soil protection has its impact on the development of policies in Iceland, though the results and consequences of this work are far from clear today.

Soil removal and treatment ex-situ

In those cases that decisions are made to remediate a site by removing the contamination with the soil and treat it elsewhere, the soil becomes **waste** in the scope of the landfill directive and subsequently has to be treated as such (Law 55/2003 on treatment of waste, implementing the Landfill Directive 1999/31/EC and Council Decision 2003/33/EC). In case of landfilling, a leaching test of the soil has to be performed to determine the waste's characteristics and on which type of landfill the waste is allowed to be disposed of. Landfilling of waste is subject to a permit of the EPA Iceland. In case of re-use or recovery of the waste a permit of the regional authority is required. However, ongoing discussions on when waste ceases to be waste have not been settled in the scope of the current revision of the Waste Framework Directive.

Some experience has been gained in Iceland with the treatment of relative small amounts of oil-contaminated soil, eg. by aeration and mixing with horse-manure. The mix is used as a thin top-soil layer on existing landfills for general waste. Regular tests of leachate from these landfills have shown no significant increase in typical oil-products so far, indicating that the contaminants are fixed and/or have degraded in time. In most other cases contaminated sites have been treated in-situ or a zero-option (do nothing) was applied.

Aspects to address in future work and policy-making

The following issues need to be addressed on a national level for making consistent policies regarding soil contamination:

1. To further develop a methodology for the classification of sites (e.g. clean, suspected, contamination confirmed, very high risk, high risk, moderate risk, low risk, etc.) and prioritise remediation actions needed, pin down contaminants, their concentrations and model their dispersion
2. To determine legal boundaries (e.g. limit values, desired background levels) for remediation, accreditation of samplers and researchers, safe land-use related to contamination level, and site-definition (e.g. geological, geographical and hydrological characteristics), identification of desired background levels and allowed environmental load on a regional and national basis
3. To revise the current legal framework, aiming to clarify liability, funding and to determine a pathway for those cases where the polluter is not known or liable
4. A more extended observation is needed to assess the extent of contaminated sites in Iceland and their environmental impacts, specific behaviour and characteristics, given the natural circumstances, e.g. climatological, hydrological, geological and soil parameters of the range of different soil types that can be found in Iceland. Assessments have to be made on the environmental impact of contamination of different sorts, both site-specific, regionally and on a national level and proposals on prioritisation of remediation actions is needed
5. It is important to work towards a general understanding and standardization of testing methods as well as a consistent interpretation of test-results through time
6. It is important to make an inventory on acceptable back-ground levels for different pollutants, to set limit-values, eg. in relation to soil-type, hydrology and (future) land-use. With determining limit-values natural background levels, environmental, scientific, financial and practical aspects have to be considered
7. The relation between relevant legislation on a national and European level has to be further investigated, also taking the ongoing Thematic Strategy on Soil Protection into account

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Status and policy on contaminated sites in Norway

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Abstract

Most contaminated land sites originate from earlier industrial and mining activities or from landfills containing hazardous waste. SFT has identified more than 3000 sites from which leachate or gas release have or may have adverse impact on the surrounding environment. By the end of December 2005, a 3 years project dealing with soil remediation was accomplished for 87 of the 93 most seriously contaminated sites in Norway. Almost 360 known sites are identified as heavily contaminated in Norway, many of which are due to seepage of contamination into ports and fjords along the coast. Work is in progress to remediate the remaining sites. The ongoing work with remediation in marine sites, are mainly based on gained experience from the soil remediation project.

High levels of heavy metals and organic pollutants have been registered in seabed sediments in ports and fjords along the Norwegian coast. This is mainly a result of earlier industrial discharges. Pollution of the seabed limits the scope for using such areas. During 2005, county action plans for contaminated sediments have been drawn up for the most polluted areas. For the remaining coastal and fjord areas, plans shall be made by 2009. Cleanup operations face a number of challenges e.g. gaps in knowledge, technological challenges, high costs and lack of clarity as to financial liability. It has often proved difficult to find suitable areas for disposal of contaminated sediments. So far, disposal sites have been located in deep- or shallow water close to the cleanup area. The «polluter pays» principle is the basis for dealing with cleanup of contaminated sediments and soil.

Introduction

Most contaminated land sites originate from earlier industrial and mining activities or from landfills containing hazardous waste. SFT has identified more than 3000 sites from which leachate or gas release have or may have adverse impact on the surrounding environment. Problems are often related to inadequate waste disposal from industrial operations and other activities. Over the years, large amounts of chemicals hazardous to health as well as the environment have been dumped in landfills, and soils have been contaminated due to leakages and accidents. The main problems are related to old landfills containing hazardous waste and industrial sites contaminated by hazardous chemicals. Contamination has been caused by both public enterprises and private companies. In many cases, contamination was caused by operations or waste disposal methods that are not considered acceptable today.

The state of Norwegian marine areas is generally good. However, in ports and fjords along the Norwegian coast, high levels of pollutants have been registered in seabed sediments. These high levels of heavy metals as well as persistent organic pollutants are mainly a result of earlier industrial discharges. Large quantities of hazardous substances were discharged into Norwegian fjords from e.g. smelting plants, the chemical industry, mining activity and the mechanical engineering branch. These discharges have been reduced in recent years. However, in some areas industry is still a significant source. In addition, seepage from old landfills, contaminated soil and agriculture, sewage, public transport and rivers contribute by transporting pollutants. Pollution of the seabed restricts the scope for using areas for fishing and fish farming activities. In some fjords, the Norwegian Food Safety Authority advises

against consumption of seafood, and has in certain cases introduced restrictions on sale of fish and shellfish. Most warnings are issued because of the presence of PCB or PAHs in fish liver or in mussels. Altogether advice against consumption applies to 31 Norwegian fjords.

Strategy for contaminated soil

By the end of December 2005, a 3 year project dealing with soil remediation was accomplished for 87 of the 93 most seriously contaminated sites. The areas were mainly old disused industry properties. The remediation costs were estimated to approximately 100 million E (Euro). Investigations have been carried out in 508 other sites of which further remediation are essential in approximately 140 sites. Continuous registration and cleanup actions are being done to bring contaminated sites back to an acceptable environmental standard. This includes large and small industrial sites, landfills and war remains. The ongoing work with remediation of marine sediments, are mainly based on gained experience from this project.

In Norway, almost 360 known sites are identified as heavily contaminated, many of which are due to seepage of contamination into ports and fjords along the coast. Further work with contaminated soil will focus on preventing hazardous substances to cause increased risk for human health as well as for preventing seepage from contaminated soil to affect areas prioritized for sediment remediation.

The Norwegian government's objective is that contaminated sites shall not pose a risk of pollution problems. When the polluter is known, the polluter is responsible for the remediation. If the polluter is difficult to obtain, the landowner is usually responsible for the cleanup operation. If any remnants left in the soil or isolation is the best solution of the problem, public restrictions for land use are established to prevent leakage, spreading or recontamination of land areas. The main goal is to restore these sites so they do not cause any harm to the environment and can be used with some or no restrictions for new purposes. The most severe sites are classified as impact level 3. Major remediation is necessary in order to bring the site to impact level 2 - recovery with some restrictions - or to impact level 1 – without any restrictions.

Every known site is located in a public database run by SFT in which information and status about sites are given in order to make information on polluted sites more easily accessible.

Strategy for contaminated sediments

Cleanup of contaminated sediments is part of an overall effort made by the environmental authorities to prevent hazardous substances and other harmful chemicals from damaging health and the environment. A long-term strategic objective aimed at «bringing the concentrations of the most dangerous chemicals in the environment down to the background level for naturally occurring substances and close to zero for man-made compounds» has been adopted (cf. White Paper No. 24 (2000-2001)). The EU Water Framework Directive also involves requirements relating to cleanup of the sediments. A strategy to ensure sufficient progress with cleanup operations in relation to the national result objective and to the requirements set out in the Water Framework Directive has already been adopted.

In Norway, the need to clean up contaminated sediments is assessed locally or regionally. Local priorities will be significant when deciding on the measures to be introduced and in what order. Detailed knowledge of the level of pollution, ongoing discharges and historical information about past discharges in the individual areas when undertaking this work is also

needed. Linking the work to a local level may also encourage active participation by local stakeholders with an interest in the cleanup operation.

Drawing up county action plans

To ensure an integrated approach in Norway as well as local involvement concerning contaminated sediments, county action plans have been drawn up during 2005 for the most polluted areas. For the remaining coastal and fjord areas, plans will be made by 2009.

The county action plans contain proposals as to what environmental quality level should be achieved for the entire fjord area. Furthermore they give an overview of the degree and extent of pollution, environmental problems and consumer interests, as well as sources of discharge. In addition, the plans describe effects and costs of different measures to handle the contaminated sediments.

County actions plans for 29 contaminated areas or fjords along the Norwegian coast have been made by the County Governor. Of these 29 areas, more detailed plans have been made for 17 areas. In some areas, cleanup operations have already been started, and SFT and/or the County Governor are working in order to start cleanup operations in other areas. The largest cleanup operation has started this year in the port of Oslo. Approximately 650,000 m³ of contaminated sediments from shallow areas in the port are being removed and placed in a deepwater disposal site nearby. Clean clay from another dredging project in the port will be used for capping of other contaminated areas within the port.

Challenges associated with cleanup operations

Several surveys have revealed a need for initiatives in polluted fjords and coastal areas. Cleanup operations, however, face a number of challenges such as gaps in knowledge, technological challenges, high costs and lack of clarity about the liability for conducting and financing the operation. In addition, preventing recontamination of the area is a major challenge. Carrying out cleanup operations in large areas with contaminated sediments is very expensive. Apart from the high cost, the situation in terms of liability is often far from clear. Dividing costs between the responsible is also difficult.

Cleanup projects often results in large amounts of polluted dredged material, and handling this is also demanding. It has often proved difficult to find suitable areas for disposal, and when such areas have been located, opposition by neighbours or NGOs has made it difficult to realize the project. Sediments in ports mainly consist of fine grained clay and silt, and treatment of this sediment is very demanding in technical terms. Hence, disposal is usually the chosen method for handling the contaminated sediments. So far, disposal sites have been placed both in deep- and shallow water at sea close to the cleanup area. A hazardous waste plant has also been used as a disposal site for sediments from time to time even though the levels of contaminants in the sediments are far below the levels for hazardous waste.

Legislation for contaminated sites

In Norway, excavating in contaminated soil is banned except when permission is given pursuant to the Pollution Regulations. Permission may be granted after an application process, but most often an order is issued either by the County Governor or SFT in order to conduct cleanup operations. In lesser cases where permission pursuant to the Planning and

Building Act is given by the municipality, permission under the Pollution Regulations is normally not needed. Contaminated soil is usually disposed of in authorized plants.

Dredging and dumping of sediments is also banned except when permission is given pursuant to the Pollution Regulations or Pollution Act. Normally, permission to dump uncontaminated sediments at sea is given at established disposal sites. Stakeholders have to apply for permission to dispose contaminated sediments. Permission may be given either by SFT or the County Governor after a public hearing. This is however, considered on a case- by case basis. If a new disposal site for contaminated sediments is to be established, permission pursuant to the Planning and Building Act is needed as well. This in turn requires a political decision.

Polluter pays principle

In Norway, the «polluter pays» principle is the basis for dealing with pollution in general. It is also the basis for the work on the cleanup of contaminated sediments and soil. Responsibility for preventing, identifying and repairing damage caused by pollution is a direct consequence of the Pollution Act. Anyone owning, doing or using something which causes pollution, or which is in any way associated with pollution, is to be regarded as responsible and may therefore be required to conduct cleanup operations. This is the main principle even though the discharges may have been known in the past and were legal at that time.

The use of orders pursuant to the Pollution Act on cleanup operations for contaminated sediments or soil does, however, give rise to a few specific problems. This is due to the facts that inputs of contaminants have taken place over a long period of time. They have often originated from many different sources and thereby many different polluters, some of which no longer exist or are difficult to identify and hold liable. The huge costs associated especially with remediation of contaminated sediments also represent a challenge when resorting to orders under the Pollution Act. So far, a cost sharing model between stakeholders and SFT has been tried out. SFT may share the costs of conducting cleanup operations if the stakeholders join the operation on a “voluntary” basis. In the sediment remediation project in the port of Oslo, SFT has paid for 30 % of the expenses. The rest has been shared between the Port Authority which is considered the main stakeholder, the municipality/city of Oslo, landowners and the Public Road Authority which is building an immersed tunnel through the port area. We believe that this kind of joint venture makes it possible to remediate larger areas compared to operations without such cooperation. It also reduces the costs for the problem owners. However, a joint venture process often requires a greater degree of cooperation, planning and a common understanding for the cleanup projects. Still, this is how SFT is planning to continue the work on contaminated sediments.

CONTAMINATED SITES IN DENMARK – STATUS OF INVESTIGATION AND REMEDIATION

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Abstract

In Denmark, the area of soil and groundwater contamination has been on the environmental agenda since the late 1970's. The aim of this paper is to give a brief overview of the status of contaminated sites in Denmark. The topics include organisation and legal framework, status of the investigation and remediation efforts, and current trends. These topics will be further detailed at the conference presentation.

Organisation and legal framework

The organisation of the area of contaminated sites can be described in terms of public authorities and private stakeholders. The public authorities include the municipalities, who are responsible for monitoring most of the possible polluting industries and activities. The counties, who are responsible for monitoring certain types of industries with an elevated risk of pollution, as well as for monitoring the groundwater. The counties also carry out the public financed investigations and remediation of contaminated sites that threaten human health or the environment when no person or company can be held responsible. The government through the Danish Environmental Agency makes the laws and regulations regarding the area, as well as manuals of e.g. how to carry out a risk assessment of a contaminated site. The Danish Environmental Agency also hosts a technology development program where new and innovative investigation and remediation technologies are tested on polluted sites in Denmark. This is typically done in cooperation with the counties.

The private stakeholders include the owners of contaminated sites, industries, and the public in general. Some of the stakeholders are joined in various NGO's such as the Danish Union of Preservation of Nature. In 1992, The Danish petroleum industry voluntarily undertook a set-up where a large number of contaminated filling stations are being investigated and remediated. This is done in cooperation with the municipalities, the counties and the Danish Environmental Protection Agency.

The legal framework consist of the Act on Protection of the Environment (Miljøbeskyttelsesloven from 1973, last revised in 2006), which aims at preventing pollution in general, and the Act on Soil Contamination (Jordforureningsloven from 1999, last revised in 2006), which aims at controlling the effects of a soil or groundwater pollution that has already taken place. The main principle of the legal framework is that a person or company who undertakes a possible contaminating activity is responsible for taking the appropriate measures to prevent any contamination of soil or groundwater, and for investigations and remediation if such pollution should take place. There are additionally a number of statutory orders regarding pollution preventing measures for specific industries or activities such as storage tanks for oil products. Other laws and statutory orders also influence this area, such as the Act on Water Supply and the EU Water Framework Directive.

Status of investigation and remediation

In 2004, a number of 9,036 sites were listed as possibly contaminated due to prior activities and 9,317 sites had been investigated and found to be contaminated. The typical contaminants include oil, petrol, chlorinated aliphatic compounds, heavy metals and creosote compounds. There has been some form of control measures or remediation activities at about 16,000 sites in the period 1994-2004. About 40% of the activities are financed by counties and the rest is financed by mainly the petroleum industry, The Danish Railroads, The Danish National Defense, plus various private companies, building project developers and land owners.

In 2005 it was estimated that additionally about 55,000 sites in Denmark might be contaminated due to former or present activities such as dry cleaning shops or service stations. It is assumed that investigations at these sites will reduce this number to 31,000 sites that are actually contaminated. 14,000 of these contaminated sites are estimated to be situated in areas of public priority, 4,000 due to risks of polluting groundwater used for water supply, and 10,000 due to risks of human health, e.g. vapors from the soil contaminating indoor climate or direct human contact with the contaminated soil. This means that a further 14,000 sites will in the future need some form of action to control or remediate the contamination. It is estimated that the total costs of investigations, control and remediation measures will be about 14.3 billion DKK, of which 10.4 billion DKK are assumed to be publicly financed and the rest privately financed.

Current trends

Recently, efforts have been made to simplify the administration of contaminated sites. Especially in the cities there are large areas with slightly elevated contamination levels due to diffuse sources such as air pollution from traffic or factories. These areas will now be classified as generally light polluted areas and the people living in these areas will be informed of simple precautions to take in order to avoid any risk of the pollution. This will make it easier for the authorities to focus on those contaminated sites that actually pose a risk to human health or the environment, and it will reduce the inconveniences for the owners of sites that are only slightly contaminated and not posing any significant risk and therefore not likely to be remediated anyway.

By the beginning of next year a change in the organisation of counties and municipalities will take place in Denmark. Many of the small municipalities will be joined to form larger municipalities. The present 14 counties will be closed down and instead 7 new regions will be formed. Most of the environmental tasks carried out by the counties will be divided between the municipalities and a number of new national centers under the Danish Environmental Agency. Public financed investigation and remediation of contaminated sites and the regulation of raw materials will be the only environmental tasks carried out by the regions.

The technology development within investigation and remediation in 2006 is concentrated on further development of methodologies for investigation and risk assessment, remediation with nano- and micro-size iron particles, remediation methods for pesticide contamination, ex situ soil treatment methods, and control measures for the transportation of soil.

References

Miljøstyrelsen 2005. Redegørelse om jordforurening 2004. Redegørelse nr. 4, 2005.

Miljøstyrelsen 2006. Teknologiprogram for jord- og grundvandsforurening 2006. Orientering nr. 3, 2006. Teknologiuudviklingsprogrammet for jord- og grundvandsforurening.



TITLE

SPEAKER

Sediment remediation in Norway

Jens Skei, Norwegian Institute for Water Research

The Svärtsjö project: environmental dredging and dewatering of pulp-and-paper sludges

Stany Pensaert, DEC NV (DEME Environmental Contractors)

Results from monitoring remediation of mercury contaminated sediments in lake Turingen

Andrew M. Peterson, WSP Environmental

Speciation and bioavailability of mercury in contaminated sediments in Sweden

Andreas Drott, Swedish University of Agricultural Sciences

Organic chemicals in sediment: novel methods for improved risk assessment and in situ remediation

Gerard Cornelissen, Stockholm University

Contamination of fjords by urban run-off

Arne Pettersen, Norwegian Geotechnical Institute

SEDIMENT REMEDIATION IN NORWAY

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Abstract

Several fjords and harbours in Norway have suffered extensive contamination of trace metals as well as organic pollutants, due to a long history of industrial activities. As a consequence fish and shellfish, at a large number of sites (> 30), contain levels of contaminants above what is considered recommendable for human consumption.

The Department of the Environment has launched a national remediation plan for contaminated marine sediments. A National Board was established in 2003 to give advices to the State Pollution Control Authority (SFT) regarding decision making, design of monitoring programmes, identification of gaps of knowledge, choice of remedial techniques etc.

In Norway to-day few full scale sediment remediation projects have been completed. We are still in the planning stage. There is one full scale capping project carried out near a zinc plant (Odda) in 1992 and a few minor capping projects. A series of pilot studies were carried out during 2001-2002, to investigate the effects of alternative remediation techniques and their implications on environmental risk. Additionally, a large scale dredging operation near a naval base, and the establishment of a near shore confined disposal site, was executed in 2003. Finally, there is a major ongoing dredging operation in Oslo harbour and deep water disposal in a fjord basin.

Contaminated sediments in Norway – state of the art

The typical coastline of Norway is intersected with narrow and deep fjords, with industry, harbors and population situated at the head of fjords (Syvitski et al, 1987). Traditionally, the fjords have been used as recipients of waste and as a result bottom sediments have been contaminated with a mixture of contaminants, organic as well as inorganic components. During the last decade the discharges of municipal and industrial wastes have decreased and questions are asked which role the contaminated bottom sediments play as a secondary source of pollution. Levels of contamination in some of the fjords are very high due to a low rate of natural sedimentation and minimal dilution. This is due to the effect of trapping of sediments in lakes upstream fjords (Syvitski et al., 1987).

The primary constituents of concern are dioxins, PCBs, PAHs, TBT, and mercury, cadmium and lead. Experimental work has been performed to assess the mobility and bioavailability of contaminants in sediments (Schanning et al, in press), and modelling has been performed to predict the effect of sediment remediation (Saloranta et al, 2006) and natural rehabilitation.

Some silled, fjord basins with restricted water circulation are oxygen depleted and anoxic with the redox boundary situated either in the water column, at the sediment-water interface or within the sediment (Syvitski et al., 1987). These conditions have considerable effects on the fate of contaminants in the sediments and implications related to remediation.

Considerable effort has been spent on mapping of contaminated sediments in Norway during the last decade. When it comes to risk evaluation related to the sediments less documentation exists.

Basic understanding of natural processes at the sediment-water interface

Decisions regarding sediment remediation should be based on best available technology as well as best available knowledge of natural processes. The knowledge of processes taking place at the sediment-water interface is of utmost importance when designing remediation projects. Knowledge of sedimentation, resuspension and sediment transport is necessary to evaluate environmental risks related to dispersal of contaminants and potential problems related to dredging or capping. Knowledge of advection, diffusion, adsorption and desorption of contaminants in sediments is also a basis of the risk assessment. This will give information on the mobility of contaminants and the sediment- water partitioning.

In addition to abiotic factors influencing the fate of contaminants in sediments, the biotic factors like bioturbation is a major issue. The extent of reworking of contaminated sediments plays a major role and adds another factor to the complexity (fig.1).



Fig.1. Bioturbation in the upper 10 cm of a fjord sediment (photo H.Nilsson, NIVA).

It appears that the need of basic understanding of natural processes taking place at the sediment-water interface is somewhat underestimated in most sediment remediation projects; in the planning phase as well in the implementation phase.

Contaminated sediments- risk or fantasy?

The objective of sediment management is to make sure that best available knowledge and technology are used to reduce the risks of environmental problems related to contaminated sediments, in a cost efficient and sustainable way (U.S EPA, 2005). Sediment remediation is complex, little predictable and very costly. Consequently, the decision making process has to be based on solid scientific knowledge, practical insight and socio-economic understanding.

At the preliminary stage of planning, the primary stakeholders and the decision makers should play an active role. The preliminary sediment management plan should describe the

environmental objectives, the risks and uncertainties, the expected timing in terms of achievement of environmental benefits and the order of actions taken. This is to avoid unrealistic expectations, which again is a source of mistrust. It should also be clearly communicated how the environmental benefits, as well as failure, are documented.

Due to the complexity of sediment management projects, effort should be made to communicate complicated technical facts, uncertainties, risk assessments and other tools used in the decision process to the public and stakeholders in an easily, understandable way (use of simplifications, but not oversimplifications). This is a challenge, but if ownership to a sediment remediation project is the objective, the parties involved will demand to understand all potential consequences and risks, environmentally as well as economically.

It is important to have a realistic view of the potential risk of contaminated sediments based on facts rather than emotion. Most sediments which are classified as contaminated are far from hazardous waste. Additionally, as the sediments are contaminated it is likely that the contaminants are strongly bound to the particulate face and it is not very probable that the contaminants are released to the water unless they are exposed to physio-chemical changes (i.e. physical disturbance, changes in redox conditions). This implies that in some cases the contaminated sediments may be considered as a potential source and in other cases as a sink. If documentation shows that the sediment may be considered as a sink, natural rehabilitation may be sufficient as a “remedy”. In cases where the rate of natural sedimentation is very low, thin layer displacement of 10 -15 cm of clean material may enhance the natural recovery of the sediment quality.

The main risk of concern is coupled to transfer of contaminants from the sediments to organism living in the sediments or at the sediment surface, and further transfer to fish and shellfish, which feed on sediment living organisms. The bioaccumulation of contaminants in these organisms will depend on the bioavailability of the contaminants in the sediments (Fig.2). In Norway health advices regarding consumption of seafood exist at more than 30 sites along the coast, due to elevated levels of contaminants. A majority of these sites also have contaminated bottom sediments, but a straight forward correlation between levels of contaminants in sediments and levels in seafood has not been demonstrated. However, contaminated sediments are considered as a contributor to the levels in fish and shellfish.

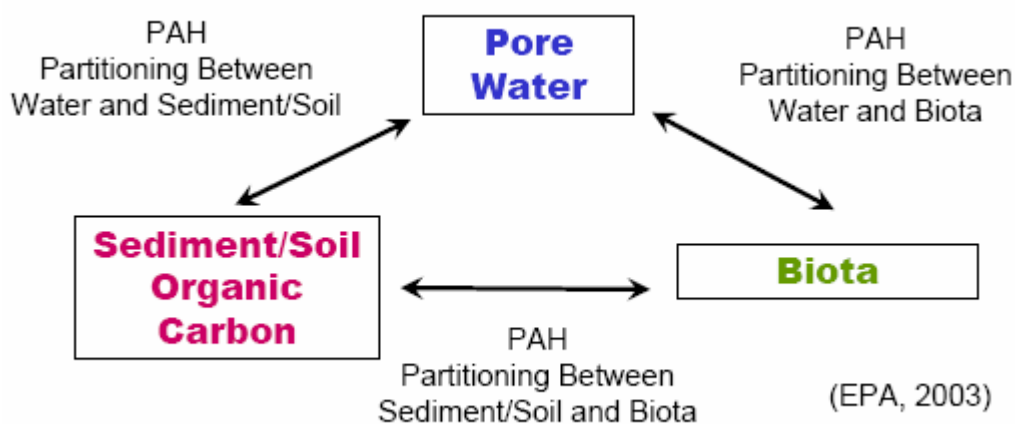


Fig. 2. Organic carbon controls bioavailability of PAHs.

Ecological risk imposed by contaminated sediments is even less documented. Research is needed to assess the levels of PCB, PAH, TBT, DDT and trace metals in the sediments which cause adverse effects on the ecosystem (effect on biodiversity etc.)

The public experience of risk and uncertainty is very often out of scale when it comes to contaminated sediments and remediation. To some extent this may be related to the quality of information given to the public (fig.3). Communication of risk is a big challenge and experience from the remediation project in Oslo harbor clearly indicates that this should not be underestimated.



Fig.3. The importance of the quality of information given about sediment remediation projects.

Sediment remediation in Norway – plans and implementation

In Norway a number of 17 counties along the coast have generated plans for sediment remediation. The priority of sites for remediation is based on a risk assessment guidance document, making sure that all sites may be compared in the same way. Furthermore, it is emphasized that site specific conditions should be taken into account as well as the local level of ambition with respect to sediment remediation. This will promote a stakeholder involvement and an ownership to the plans. At present the State Pollution Authority (SFT) has made a shortlist of 9 sites which will be subject to further planning and investigation. These are all marine sites related to harbors and fjords.

A challenge in these plans is to express environmental goals which are realistic and where the success (or failure) can be documented in a quantitative way. Development of suitable tools for decision making, which analyse the potential of achieving the objectives (and when) and the uncertainty and risk involved in the remediation project, is under way.

References.

- Saloranta, T.M., T. Andersen, and K. Næs. 2006. Flows of dioxins and furans in coastal food webs: Inverse modelling, sensitivity analysis and application of linear system theory. *Environ. Toxicol. Chem.*, 25: 253-264
- Schaanning, M., Breyholz, B. og Skei, J. (2005) Experimental results on effects of capping on fluxes of persistent organic pollutants (POPs) from historically contaminated sediments. *Mar. Chem.*, in press.
- Syvitski, J.P.M., Burrell, D.C. & Skei, J.M. 1987. *Fjords. Processes and Products*. Springer-Verlag, 379 pp. New York-Heidelberg-London-Paris-Tokyo.
- U.S.EPA .2005. Contaminated sediment remediation guidance for hazardous waste sites. EPA-540-R-05-12.

The Svärtsjö project: environmental dredging and dewatering of pulp-and-paper sludges.

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Abstract

The two Svärtsjö lakes near the city of Hultsfred (Sweden) have been contaminated in the past by pulp originating from a paper mill upstream the Pauliström. The pulp, which is highly organic and polluted with mercury, has settled down in both lakes, where it causes a serious threat to the marine fauna and flora. For this reason the Swedish EPA decided to remove about 300000 m³ of sediments in the course of 2006. The sediments have to be dewatered, and stored in a final storage area in the vicinity of the lakes.

The remote location of the lakes, combined with the environmental aspect of the dredging imposes quite some boundary conditions on the dredging operations:

- The dredger has to be completely dismountable and transportable over road as the nearest port is located at 200 km distance.
- In order to guarantee very accurate and low turbidity dredging, the positioning of the dredger has to be done based on an anchoring system at the shore of the lakes.

For this particular project a new dredger was designed and constructed, as shown in the picture below. It is based on 7 connectable and transportable pontoons. Positioning of the dredger in the lakes is based on four anchoring points along the shore, and a set of four GPS controlled winches with which a desired area can be dredged automatically.

At a rate of about 300 m³ per hour the dredger pumps the sediments via a 2 km long floating pipe system to the final storage area. Onshore, organic flocculant is injected controlled by continuous measurement of dry matter content flow in the pipe. Then the sediments are dewatered in geotubes, and the filtrate is treated in a biological water treatment unit.

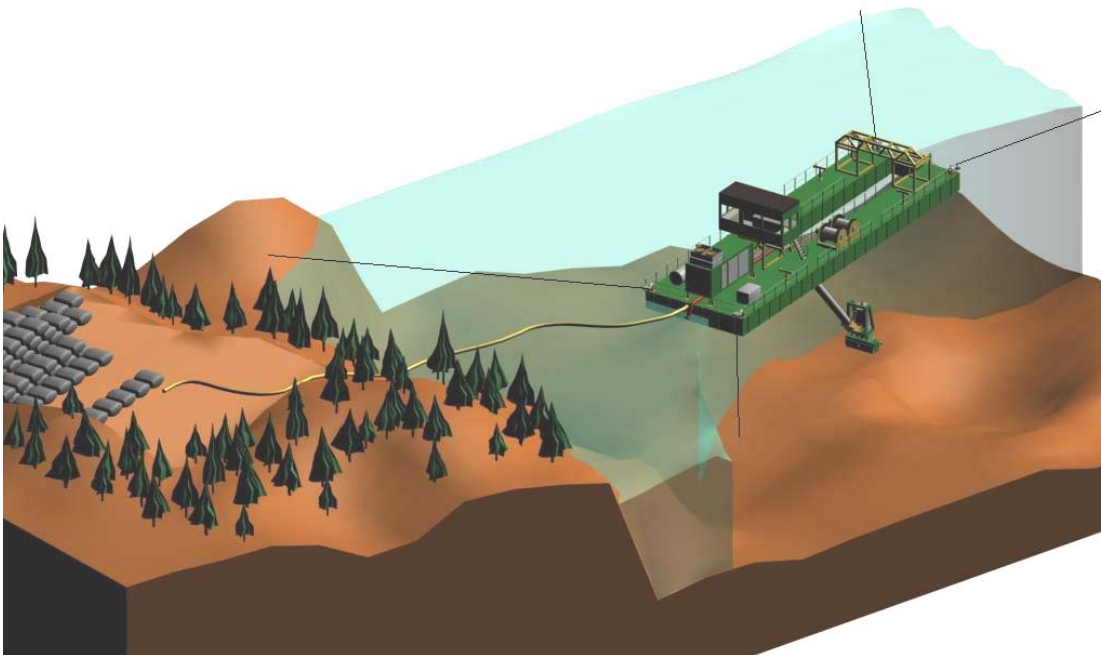
The advantage of the use of geotubes is that dewatering of sediment at this high flow rate is possible, and that the geotubes can be stacked allowing disposal in a sloped area. In addition, the geotubes textile gives a durable reinforcement and ensures stability of the dewatered and stored sediments.

As this project is ongoing in 2006, the presentation will outline and illustrate the whole project and in particular the challenges that were undertaken in synchronizing the dredging operations, sediment dewatering, and water treatment plant.

Nordrocs, 20-21 September 2006, Malmö Sweden



Picture of the environmental dredger.



Artist impression of the site.

RESULTS FROM MONITORING REMEDIATION OF MERCURY CONTAMINATED SEDIMENTS IN LAKE TURINGEN

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Abstract

Mercury contaminated bottom sediments in and upstream of Lake Turingen (Sweden) were remediated in four separate stages between 1995 and 2003, using both conventional methods and capping with artificial sediment. An extensive environmental monitoring program has been used to evaluate the project. The EM-program encompasses regular sampling of lake and river water, falling sediment and biological matrices, and core sampling. Results indicate that the remedial works isolated about 95% of all mercury in the lake. This has led to dramatic changes in the amount of mercury circulating in the system, in particular that which is bound to organic particulates. However, as of August 2006, no definite changes have yet been observed regarding the biological variables that correspond to project goals. The monitoring program is currently funded through 2008.

Introduction

Bottom sediments in Lake Turingen, a 1000 ha freshwater lake about 40 km southwest of Stockholm, Sweden (Fig. 1), are contaminated with mercury discharged between 1946 and 1966 from a decommissioned upstream paper mill. These were remediated in four stages, as separately contracted works. In the first stage (A), reservoir sediments upstream of the lake were isolated. The second stage (B) entailed a small amount of dredging of contaminated materials plus capping of 4 ha (near the mouth of the stream feeding the lake) with a geotextile, sand and crushed rock. In the third stage (C), about 80 ha were capped by artificial sediment based on aluminum hydroxide¹. Additional isolation of flood-plain sediments took place in 2004 (D). Further details about stages B and C can be found in Municipality of Nykvarn (2004) and Bergman et al (2005).

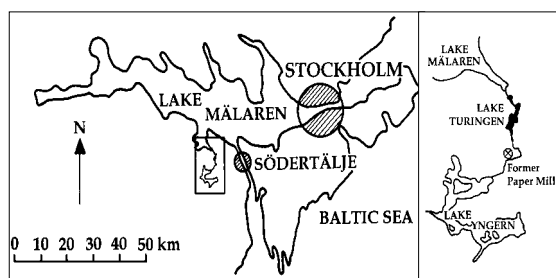


Figure 1. Location of Lake Turingen (Bergman and Petsonk, 1997).

An extensive environmental monitoring (EM) program has provided project management with indications of the short and long-term effects of remedial activities, as well as natural and other occurrences, which could impact on or be affected by remedial activities. The EM-program has been carried out in close cooperation with the construction quality control program used to ensure conformance with project specifications.

¹ The Cover Method (Vattenresurs AB).

Materials and methods

The system being monitored is extremely complex, and measurement of numerous variables in a variety of matrices is required in order to draw relevant conclusions. However, the EM-program has access to a large body of reference data and currently encompasses regular sampling of lake and river water, falling sediment, bottom sediment and biological matrices at numerous stations in and around the lake (see Fig. 2 and Table 1). These data have been important for decision-making during remediation, and currently provide preliminary indications as to how successful the various remedial stages have been at improving the environment in the lake. The program is currently funded through 2008.

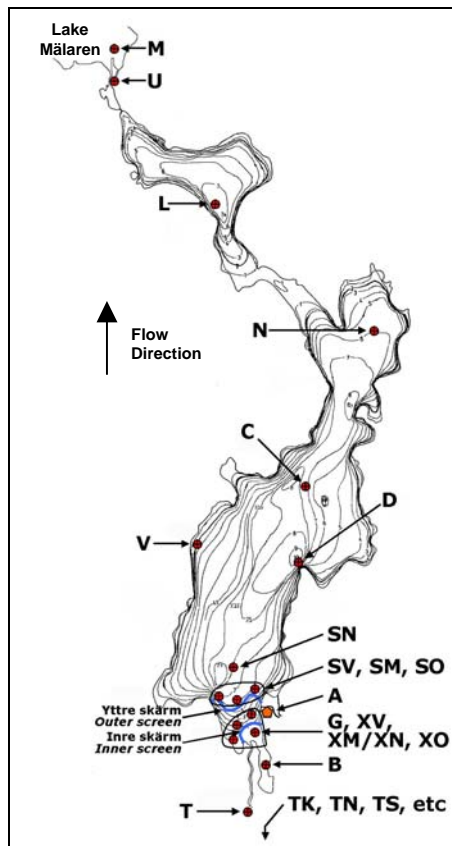


Figure 2. Fixed monitoring stations in and near Lake Turingen (Municipality of Nykvarn, 2004). Not all stations are currently active.

Table 1. Measurement parameters for environmental monitoring (Municipality of Nykvarn, 2004).

Medium	Parameters	Frequency
Noise*	<ul style="list-style-type: none"> Background noise Construction noise 	As required
Meteorological data*	<ul style="list-style-type: none"> Air temperature Precipitation Wind speed & direction 	Every 15 minutes
Hydrological data*	<ul style="list-style-type: none"> Lake water levels Stream flow 	See water
Lake and stream water	<ul style="list-style-type: none"> pH, alkalinity, color Turbidity, visibility Al, Cl, Fe, Mn, Hg, methyl-Hg Temp, O₂, EC, redox 	At least quarterly; weekly or daily during Stage B
Falling sediment	<ul style="list-style-type: none"> Al, pH4-Al, Fe, Mn, Hg, TOC Sedimentation rate 	At 8–12 week intervals
Bottom sediment	<ul style="list-style-type: none"> Al, pH4-Al, Fe, Mn, Hg, N, P, TOC 	As required
Zooplankton	<ul style="list-style-type: none"> Hg, methyl-Hg Species 	See water
Benthic & littoral fauna	<ul style="list-style-type: none"> Species Hg, methyl-Hg 	Annually
Perch, pike	<ul style="list-style-type: none"> Weight, length, sex, age Hg, methyl-Hg 	Annually

* Monitoring of these media was discontinued after completion of the remedial works.

Results

Core samples have been assessed both visually and chemically (with respect to mercury and/or pH4-aluminum²). Initial Hg-concentrations in the lake ranged up to 35 mg/kg dw. After being applied, artificial sediment was easy to distinguish from natural sediment; incremental treatment produced distinct layers, and concentration gradients

² Measurement of pH4-aluminum uses an analytical technique developed for the project (Kaj and Skarp, 1997) that differentiates between artificial and natural sediment by the leachability of aluminum at pH4.

were observed with increased sediment depth for both Hg and pH4-Al (Fig. 3). However, later results indicate both reduced gradients (mixing) and resuspension and/or recrystallization of artificial sediment.

Other measurements demonstrated that remedial works had only a limited, temporary effect on basic aquatic parameters such as color, turbidity, Secchi depth (visibility), conductivity and chloride concentrations.

Results from the EM-program can be found in Petsonk (2006). In the long term, mercury levels in water have decreased significantly and continue to decrease. Concentrations of most substances in water – including mercury – continue to vary cyclically, following the change of seasons and lake dynamics. During periods of stratification, levels of metals – especially mercury – are still much higher in deep water than in shallow water (Fig. 4).

Monitoring of falling sediment has been performed since 1994 at four stations, two in the lake itself and two downstream (Fig. 5). Stage A works had very little effect (if any) on the content of falling sediment. However, capping outside the mouth of the stream in 1999-2000 (Stage B) brought about a reduction of the mercury load, indicating that resuspension of contaminated sediment from that area had probably been eliminated. Additional reductions were achieved during Stage C. There was a disturbing spike in the data during autumn 2005, but it appears that this may have been an anomaly.

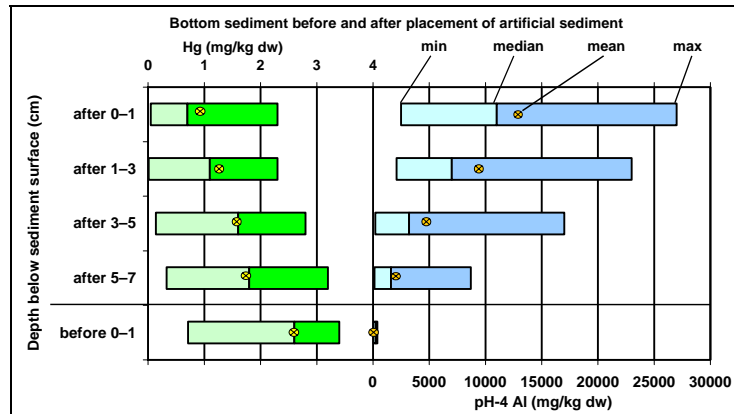


Figure 3. Hg and pH4-Al concentrations in sediment before and after application of artificial sediment. Higher pH4-Al values correspond well with lower Hg values.

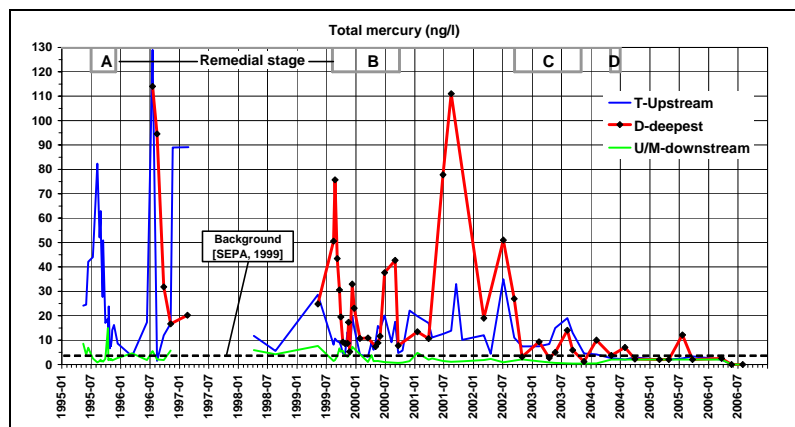


Figure 4. Mercury concentrations in water upstream, downstream and in the deepest part of the lake.

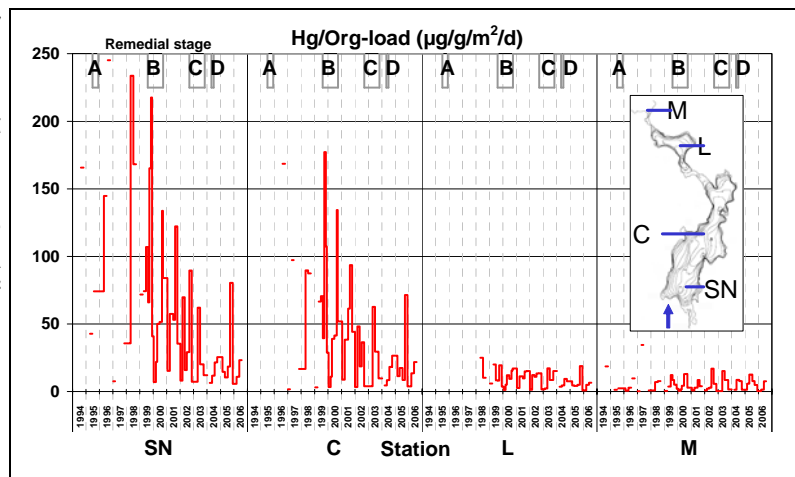


Figure 5. Flux of mercury bound to organic particles in falling sediment.

Levels of mercury and methyl mercury in biota have not yet shown any significant reductions, although concentrations in zooplankton do appear to be slowly decreasing (Fig. 6)³. Mercury concentrations in perch appear to be directly related to concentrations in zooplankton, and have not yet shown any significant influence from remedial activities. Concentrations in pike have also remained relatively constant for the past 35 years, approximately 2 mg/kg ww.

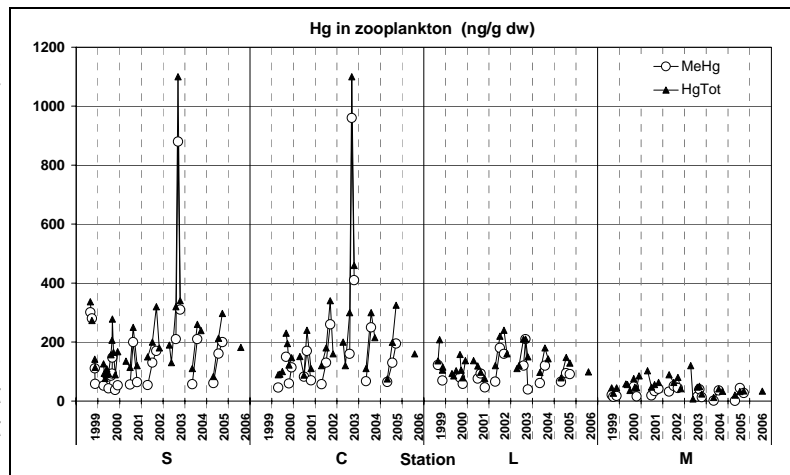


Figure 6. Concentrations of total and methyl mercury in zooplankton during Stages B and C.

Discussion

Results from this project illustrate the positive impact of capping contaminated sediment, both with natural materials and with artificial sediment. Mercury concentrations in water and falling sediment have decreased markedly. Even though positive biological effects have not yet been observed – i.e. the long-term effects of remediation are still uncertain – the ecological conditions for aquatic life have been substantially improved. As the monitoring program will continue through at least 2008, it will be extremely interesting to see what additional changes occur in the many parameters being observed.

There are indications that the artificial sediment cap may be eroding, changing character and/or mixing with underlying sediment. Levels of nutrients are also rising, perhaps indicating leakage into the water column. The underlying reason for these occurrences is also a matter of great interest, and should be further explored.

References

- Bergman, R., Carlsson, S-Å. & Petsonk, A. 2005. "In-Situ Mitigation of Mercury Contaminated Sediments in Lake Turingen". In Proc. Third International Conference on Remediation of Contaminated Sediments. Battelle Press, Columbus, OH.
- Bergman, R. and Petsonk, A. 1997. "Lake Turingen Remedial Project: Isolation of Mercury Contaminated Sediments". In Proc. Int'l Conference on Contaminated Sediments, Rotterdam. pp 25-32.
- Kaj, L., and Skarp, J. 1997. "pH4-Method for Leaching Aluminum from Sediment". Unpublished, in Swedish.
- Municipality of Nykvarn. 2004. "Lake Turingen Remedial Project - Final Project Report". http://www.turingen.se/bilder/slutrappport_final_project_report.pdf.
- Petsonk, A. 2006. "Lake Turingen Remedial Project: Status Report 1 November 2004 – 30 November 2005". Report to Municipality of Nykvarn, 54 pp, in Swedish. http://www.turingen.se/bilder/turingenrapportu2005_rev_060810.pdf.

³ It is believed that the drastic increase in mercury levels during 2002–2003 was an indirect result of the decrease in organic particulate present in the water column caused by production of artificial sediment.

SPECIATION AND NET METHYL MERCURY FORMATION IN MERCURY CONTAMINATED SEDIMENTS

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Abstract

The presentation will focus on results and conclusions from an ongoing research project on speciation and biogeochemistry in mercury (Hg) contaminated sediments. Some of the results are also presented in this short paper. The results support the hypothesis that the availability of Hg(II) to methylating microbes is controlled by the concentration of neutral Hg sulphides in the sediment pore water. Differences among sites show that the supply of energy-rich organic carbon also is of importance for Hg methylation rates. The solubility of MeHg in anoxic sediments is mainly controlled by the concentration of dissolved organic carbon (DOC) and inorganic sulphides in the sediment pore water.

Introduction

As it is mainly mono methyl mercury (MeHg) that biomagnifies in aquatic food webs, the risk of mercury (Hg) contaminated sediments is closely linked to the formation and solubility of this species. The concentration of MeHg in contaminated sediments is mainly determined by the net result of MeHg production (methylation) and MeHg degradation (demethylation) within the sediment. The solubility of produced MeHg is in turn largely determined by the composition of the sediment pore water, controlling solution MeHg speciation.

Results from an ongoing research project primarily aiming at identifying the main factors controlling net MeHg production and Hg and MeHg speciation in contaminated sediments will be presented. The intention of the project is to provide new knowledge in order to refine risk assessments. The presentation will mainly focus on the linkage between speciation and net methyl mercury formation, but other critical factors will be emphasised as well. This short paper presents some of the results and conclusions.

Material and methods

During 2004 and 2005, seven Hg contaminated sites in different parts of Sweden were sampled at various times of year. The sites differed with respect to environmental conditions, e.g. climate, salinity and primary production. All sites had been subjected to local Hg contamination either as Hg⁰(l) or as phenyl-Hg.

During sampling and sample pre-treatment, sediments and pore waters were kept under anaerobic conditions as much as possible. A comparison of filtration (after centrifugation) under oxic (ambient air) and anoxic (N₂) conditions showed that oxic filtration resulted in anomalously low pore water MeHg concentrations (as compared to anoxic filtration), pointing to the importance of anoxic handling during sample pre-treatment. (Drott et. al. 2006a).

Concentrations of total MeHg in sediments and total Hg and MeHg in sediment pore waters, as well as potential rates of methylation and demethylation during a 48-h incubation under laboratory conditions, were determined using species-specific isotope dilution gas chromatography inductively coupled plasma mass spectrometry (SSID-GC-ICPMS), after

ethylation with sodium tetraethylborate, or, in the case of pore water total Hg, reduction to Hg^0 (EPA method 1631). The concentration of sediment total Hg was determined on a Leco AMA 254 mercury analyzer. Concentrations of pore water dissolved organic carbon (DOC), dissolved inorganic carbon (DIC), Cl, Br, S, Fe, Mn and Ca, as well as total sediment, C, N, S and Fe were determined using standard methodology.

Thermodynamic equilibrium modelling of the chemical speciation of Hg(II) and MeHg in solution was accomplished by speciation calculations. Required equilibrium constants were taken from the scientific literature.

Results

In Figure 1, the calculated concentration of neutral, inorganic Hg sulphides in the sediment pore water is related to the experimentally determined specific potential methylation rate constant, K_m , for sites Köpmanholmen and Skutskär (filled circles), Karlshäll (open triangles), and Marnästjärn and Turingen (filled squares). Data for two surface sediments (0-5 cm) at Köpmanholmen, and average data for Övre Svartsjön and Nötöfjärden are included for comparison. As can be seen in the figure, the concentration of neutral inorganic Hg sulphides was significantly ($p < 0.001$, $p = 0.021$, and $p = 0.021$, respectively for the three sub-sets) positively correlated to K_m . The concentration of total Hg, organic thiol complexes, or total inorganic Hg sulphides in the pore water did not show any significant relationship to K_m (data not shown). Depending on site characteristics, three different models (A, B and C) were used to calculate the speciation of Hg in the sediment pore water.

The slopes of the relationships in Figure 1 differed between the three sub-sets of data. Also, C/N ratios differed substantially. The C/N ratio can be used to indicate differences in primary production and energy-rich organic matter availability among sites.

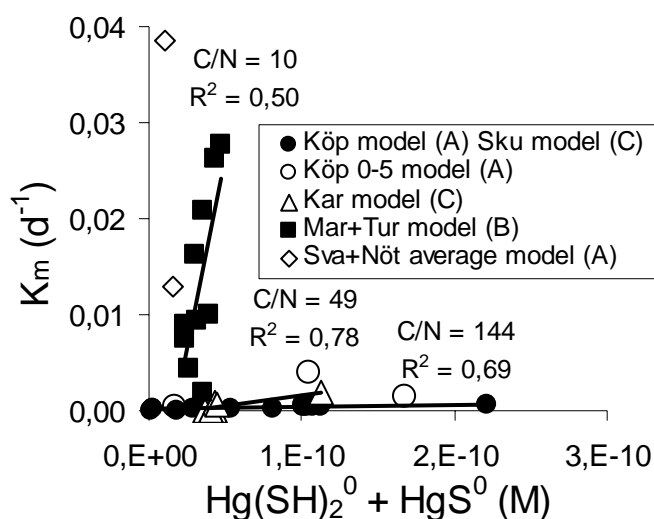


Figure 1. Relationships between neutral Hg sulfide concentrations and K_m for sub-sets consisting of Köpmanholmen and Skutskär, Karlshäll, and Marnästjärn and Turingen.

In Figure 2, the output of the speciation modeling for MeHg is shown for Köpmanholmen (a) and Skutskär (b). Note that the concentrations of MeHgCl , MeHgOH and MeHg^+ never exceeded 0.01 % of total pore water MeHg. Thus, the solubility of MeHg is mainly controlled by the concentration of DOC (forming MeHgSR) and inorganic sulphides (forming MeHgSH and MeHgS^-) in the sediment pore water.

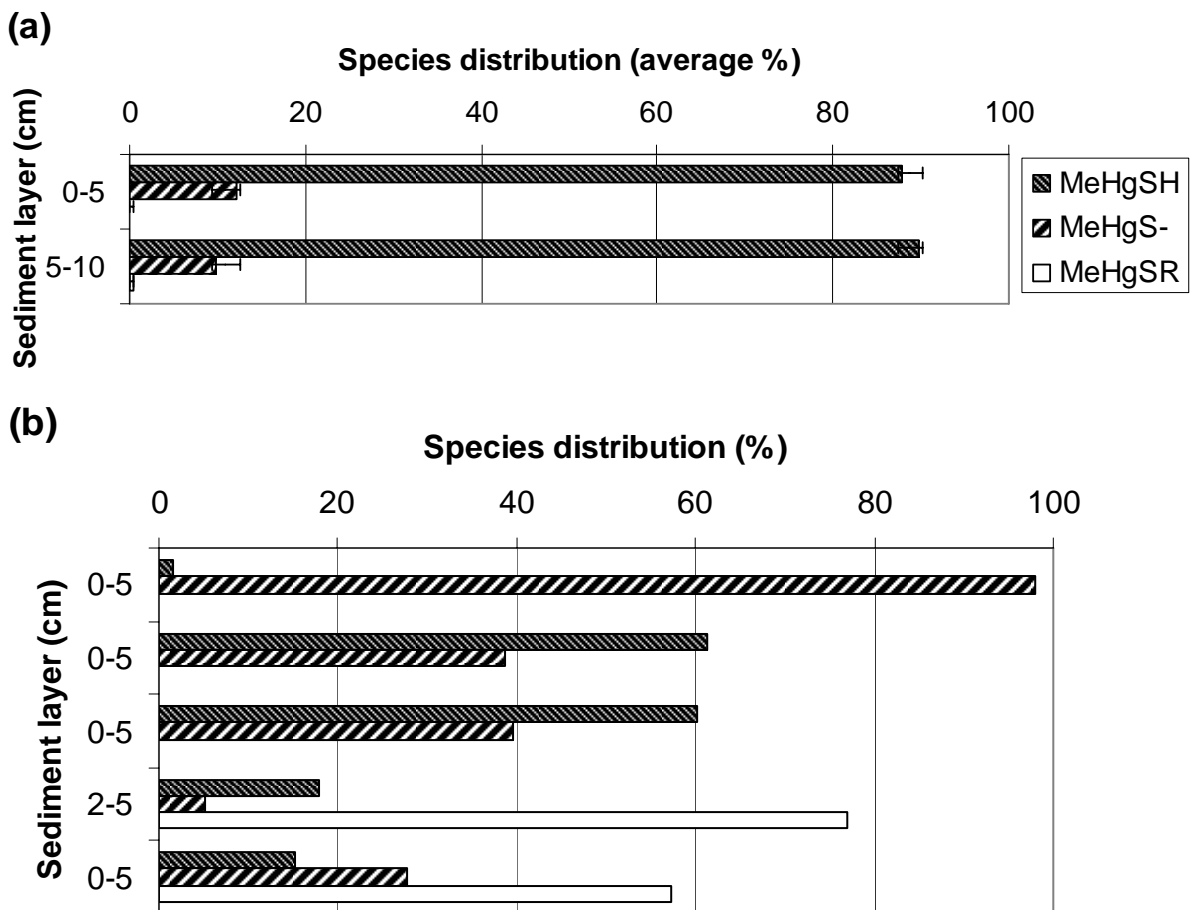


Figure 2. Dominant MeHg species distributions at (a) Köpmanholmen, (b) Skutskär. At Köpmanholmen (a), the distribution is displayed as average % for each sediment layer with error bars representing SD (n = 8).

Discussion

As Hg methylation in sediments mainly is performed by sulphate reducing bacteria (SRB), factors controlling the activity of SRB like: 1) sulfate (e^- -acceptor), 2) energy-rich organic matter (e^- -donor), and 3) temperature, together with: 4) availability of Hg(II) largely control Hg methylation rates, and subsequently sediment MeHg concentrations. The relationship between the concentration of neutral inorganic Hg sulphides and K_m displayed in Figure 1 support the hypothesis that the availability of Hg(II) to methylating microbes is controlled by the concentration of neutral species that easily can diffuse across biological membranes (e.g. Barkay et. al. 1997, Benoit et. al. 2001). We find similar relationships for total MeHg in sediments (Drott et. al. 2006b).

The differences in slopes and C/N ratios among sites likely reflect differences in primary production of pelagic plankton and algae, leading to differences in availability of energy-rich organic matter among sites. The low C/N ratio at the southern limnic sites (Marnästjärn and Turingen) likely reflects a high availability of energy-rich organic matter, leading to higher activity of SRB, and thus higher methylation rates, as compared to the brackish water sites (Köpmanholmen and Skutskär), and the northern limnic site (Karlshäll).

As shown in Figure 2, the solubility of MeHg in anoxic sediments is mainly controlled by the concentrations of inorganic sulphides and DOC in the sediment pore water. The solubility of

MeHg is critical for transport out of the sediment, uptake in biota, and probably also access to MeHg for demethylating microbes.

References

Barkay T., Gillman M., and Turner R.R. 1997. Effects of dissolved organic carbon and salinity on bioavailability of mercury. *Applied and environmental microbiology*. 63:4267-4271.

Benoit, J.M., Gilmour, C.C. and Mason, R.P., 2001. The influence of sulfide on solid-phase mercury bioavailability for methylation by pure cultures of *desulfobulbus propionicus* (1pr3). *Environmental Science & Technology*. 35:127-132.

Drott A., Lambertsson L., Björn E. and Skjllberg U. 2006a. Effects of oxic and anoxic filtration on determined methyl mercury concentrations in sediment pore waters. *Marine Chemistry*. In press.

Drott A., Lambertsson L., Björn E. and Skjllberg U. 2006b. Influence of mercury speciation in pore water on methyl mercury production in contaminated sediments. Submitted.

ORGANIC CHEMICALS IN SEDIMENT: NOVEL METHODS FOR IMPROVED RISK ASSESSMENT AND IN SITU REMEDIATION

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Abstract

Freely dissolved contaminant concentrations in soils and sediments are found to be much better measures of human and ecotoxicological risks than total concentrations. Such freely dissolved concentrations can be directly compared to toxicity-based water quality criteria to improve the assessment of actual risks.

For four priority Norwegian harbour locations, it appeared that the actual PAH risks were around 100 times lower than predicted on the basis of total concentrations, and that current risk assessment practice would thus overestimate PAH risk of these sediments by a factor of 100. Bioaccumulation tests with two organisms confirmed these findings.

In addition, we tested the addition of active carbon (AC) as an in situ remediation method for PAH contamination in sediments. It was shown that AC amendments of 2 wt-% reduced freely dissolved concentrations by a factor of 10-50; reductions of uptake in organisms were around a factor of 5. Thus, actual risks are reduced by AC amendment.

Introduction

Strong sorption to carbonaceous geosorbents

It has recently been shown that the presence of carbonaceous geosorbents (CG, including black carbon (BC; soot and charcoal), unburned coal and kerogen) can cause strong sorption of polycyclic aromatic hydrocarbons (PAHs), polychlorinated biphenyls (PCBs) and dioxins in sediments¹. We studied sorption of native PAHs in four Norwegian harbor sediments of which high fractions (21-56%) of the total organic carbon (TOC) consisted of carbonaceous materials, as shown by organic petrography. PAH sorption coefficients were 1-2 orders of magnitude above predictions based on amorphous organic carbon (AOC) partitioning alone⁵. Thus, strong sorption to carbonaceous geosorbents causes PAHs to be sorbed much more strongly to sediment than expected on the basis of K_{OC} -values, and than assumed in current risk assessment that is based on these K_{OC} values.

Improved risk assessment

In many countries, namely, human and ecotoxicological risk assessment of organic contaminants is based on quality criteria developed for the total contaminant content¹⁻⁴. However, freely dissolved contaminant concentrations in soils and sediments are found to be much better measures of human and ecotoxicological risks¹⁻⁴, because these take into account the abovementioned strong sorption. Such freely dissolved concentrations can therefore be directly compared to toxicity-based water quality criteria to improve the assessment of actual risks.

Activated carbon amendment

Luthy and coworkers were the first to recognize that the strong sorption to carbonaceous materials could be used in an engineering perspective⁸⁻¹¹: by the in situ addition of the carbonaceous material activated carbon (AC), one should be able to increase the sediment's sorption and therefore reduce the actual risk of present PAHs and other organic contaminants.

Methods

Passive samplers for Norwegian harbour sediments and dioxin-contaminated sites in Sweden

We used the passive sampler Polyoxymethylene (POM) to measure freely dissolved PAH concentrations for three priority Norwegian harbour locations^{5,6} (characteristics in Table 1).

At the moment we employ several types of passive samplers to monitor freely dissolved aqueous concentrations in the field. In addition, their applicability is tested for dioxins in Swedish dioxin-contaminated pentachlorophenol wood-doping sites.

Activated carbon amendment

In addition, we tested the addition of active carbon (AC) as an in-situ remediation method for PAH contamination in sediments⁵⁻¹¹. We studied this technique for three Norwegian harbor sediments, by measuring freely dissolved concentrations and bioaccumulation in two organisms in the presence and absence of AC.

TABLE 1: Sediment characteristics of Oslo (OS), Bergen (BG) and Tromsø (TR) sediments, partly from ref. 7.

	OS	BG	TR
Latitude	59° 54' 21" N	60° 23' 26" N	69° 39' 03" N
Longitude	10° 45' 01" E	05° 18' 15" E	18° 15' 42" E
Total Organic Carbon (TOC;%)	4.2	7.4	1.7
Total Organic Nitrogen (%)	0.26	0.27	0.13
Black Carbon (BC;%)^a	0.25	0.90	0.12
BC:TOC	6.1	12.1	6.7
CGC:TOC^b	50.9	55.8	27.9
Total-PAH (mg/kg dw^c)	31.1 ± 1.5	161 ± 31	9.0 ± 1.1

^a BC, Black Carbon, soot and charcoal that show strong sorption of PAHs, PCBs and dioxins.

^b CGC, Carbonaceous Geosorbent Carbon, i.e., the total carbonized organic matter + the unburned coal carbon, derived from organic micropetrography⁵.

^c dw, dry weight

Results and discussion

Freely dissolved concentrations and risk assessment

It appeared that the sediment-water distribution ratios for all PAHs were around 100 times higher than the values used to derive sediment quality criteria in risk assessment (Table 2). These high values could be explained on the basis of high soot and coal contents in the sediments⁵. Thus, risks were approximately as much as a factor of 100 lower than predicted on the basis of total sediment contents. As a consequence, current risk assessment practice would grossly overestimate PAH risk of these sediments.

TABLE 2: PAH distribution coefficients for four representative native PAHs in four Norwegian Harbour sediments. The K_{OC} used in current risk assessment is almost two orders of magnitude lower than the K_{OC} measured in all four field-contaminated harbour sediments. The reason is strong sorption to carbonaceous materials.

	Phenanthrene	Pyrene	Benzo[a]pyrene	Benzo[ghi]perylene
log K_{OC} risk assessment (L/kg)	4.2	4.7	5.7	6.1
log K_{TOC} field (L/kg)				
OS	6.19 ± 0.09	6.24 ± 0.09	7.45 ± 0.10	8.06 ± 0.05
BG	5.79 ± 0.17	6.09 ± 0.14	7.47 ± 0.04	8.06 ± 0.14
TR	6.13 ± 0.12	6.28 ± 0.16	6.78 ± 0.01	7.67 ± 0.01

Bioaccumulation tests with two organisms confirmed these findings: uptake by these organisms was very limited, with biota to sediment accumulation factors (BSAFs) as low as around 0.01.

On the basis of total bioavailability, the theoretical BSAF is around 1. Therefore uptake was also a factor of 100 lower than expected based on total PAH contents and total availability⁶ (Table 3). The reason is strong sorption to carbonaceous materials^{1,5,6,7}.

Thus, freely dissolved concentrations explained the organisms' contents well⁶. Therefore novel passive sampler methods will lead to a reduction of the uncertainty in risk assessments, a better prioritization of limited remediation resources, and improvements of the optimal design for monitoring.

Active Carbon addition to reduce bioavailability and actual risk

It was shown that AC amendments of 2 wt-% reduced freely dissolved concentrations by a factor of 10-50 (Figures 1,2). This implies that actual risks of PAHs in the three sediments were reduced by that factor. Reductions of uptake in organisms were around a factor of five⁶ (Table 4). Thus, actual risks are reduced by AC amendment^{5,6}. This technique has been shown to be effective for PCBs as well⁸⁻¹¹. It will be tested for dioxins in soil and sediment, as dioxins are strongly sorbed by AC.

TABLE 3: Biota sediment accumulation factors (BSAF) of original Oslo (OS), Bergen (BG) and Tromsø (TR) sediments (kg_{OC}/kg_{lipid}), as well as ratios between theoretical and empiric BSAFs ($BSAF_{theory} / BSAF_{measured}$) and between BSAFs calculated on the basis of freely dissolved concentrations and empiric ones ($BSAF_{free} / BSAF_{measured}$). It is observed that the freely dissolved concentrations much better predict the low BSAFs (within a factor of 2-6) than theoretical BSAFs based on total sediment contents (factor of 10-200 off).

		<i>Nereis diversicolor</i> (Polychaeta)			<i>Hinia reticulata</i> (Gastropoda)		
		BSAF _{measured}	Ratio BSAF _{theory} / BSAF _{measured}	Ratio BSAF _{free} / BSAF _{measured}	BSAF _{measured}	Ratio BSAF _{theory} / BSAF _{measured}	Ratio BSAF _{free} / BSAF _{measured}
OS	PYR	0.095 ± 0.010 ^c	10.5 ± 1.1	0.30 ± 0.02	0.077 ± 0.009	13 ± 2	0.40 ± 0.02
	BAP	0.0042 ± 0.0015	240 ± 86	3.4 ± 1.0	0.0064 ± 0.0037	156 ± 89	2.2 ± 1.1
	BGP	0.0036 ± 0.0014	274 ± 105	6 ± 2	0.012 ± 0.009	81 ± 59	1.8 ± 1.3
BG	PYR	0.11 ± 0.04	9 ± 4	0.36 ± 0.05	0.032 ± 0.012	31 ± 11	1.3 ± 0.1
	BAP	0.006 ± 0.003	167 ± 72	2.2 ± 0.7	0.006 ± 0.002	156 ± 46	2.1 ± 0.4
	BGP	0.009 ± 0.006	114 ± 72	2.5 ± 0.9	0.018 ± 0.007	56 ± 22	1.2 ± 0.2
TR	PYR	0.058 ± 0.013	17 ± 6	0.45 ± 0.03	0.015 ± 0.004	69 ± 21	1.8 ± 0.3
	BAP	0.011 ± 0.007	86 ± 50	6 ± 2	0.008 ± 0.007	120 ± 96	8 ± 5
	BGP	0.007 ± 0.004	146 ± 86	8 ± 4	0.010 ± 0.006	96 ± 56	5 ± 2

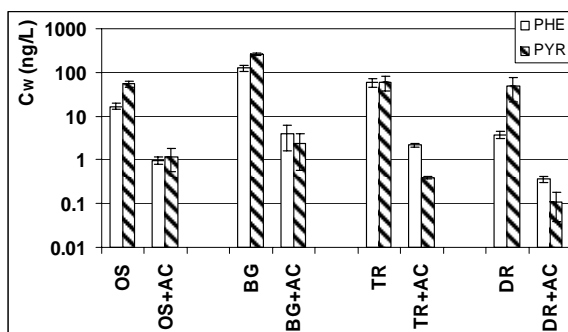


FIGURE 1: Effect on C_w of 2% AC in OS, BG, TR and DR sediments, for four PAHs.

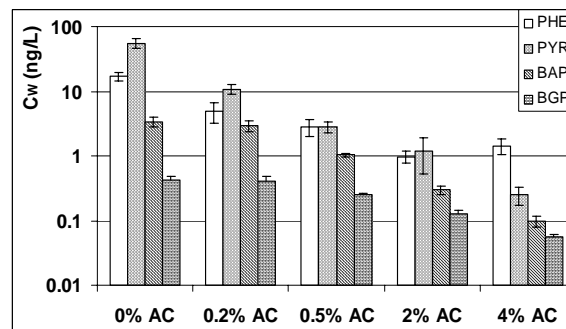


FIGURE 2: Effect on C_w of four dosages of AC in OS sediment.

Implications: AC as an in situ remediation technology

The advantages of the new environmental technology of AC amendment over many other remediation methods include i) it can be used as an in situ risk reduction method, ii) the price is low, and iii) it overcomes significant controversies associated with disposal of dredged materials.

Also, unlike dredging, AC amendment of sediment need only treat the upper sediment layer comprising the biologically active zone. Much money can be saved if the cost of cleaning up contaminated soils and sediments is reduced by a cost-effective, innovative technique such as AC amendment⁵⁻¹¹.

TABLE 4: Biota sediment accumulation values (BSAF; kg OC/kg lipid) after activated carbon (AC) addition (2%), and reduction factors of lipid-normalized organism contents ($C_{\text{biota, lipid}}$) due to 2% AC amendment, for Oslo (OS), Bergen (BG) and Tromsø (TR) sediments. For example, $C_{\text{biota, lipid}}$ of pyrene (PYR) in OS sediment was reduced 29-fold by the addition of 2% AC⁶.

		<i>N. diversicolor</i>		<i>H. reticulata</i>	
		BSAF _{AC} ^a	Reduction factor $C_{\text{biota, lipid}}$ ^b	BSAF _{AC}	Reduction factor $C_{\text{biota, lipid}}$
OS	PYR	0.0048 ± 0.0033 ^f	29 ± 20	0.010 ± 0.002	11 ± 2
	BAP	0.0023 ± 0.0016	2.5 ± 2	0.007 ± 0.002	1.3 ± 0.8
	BGP	0.0006 ± 0.0002	9 ± 6	0.014 ± 0.003	1.3 ± 1.1
BG	PYR	0.008 ± 0.006	19 ± 14	0.009 ± 0.008	4 ± 3
	BAP	0.007 ± 0.006	1.1 ± 1.2	0.007 ± 0.005	1.1 ± 0.8
	BGP	0.014 ± 0.012	0.8 ± 0.7	0.018 ± 0.014	1.2 ± 0.7
TR	PYR	0.003 ± 0.002	43 ± 33	0.02 ± 0.03	2 ± 2
	BAP	0.010 ± 0.004	2.0 ± 1.2	0.05 ± 0.06	0.4 ± 0.6
	BGP	0.009 ± 0.001	1.5 ± 0.7	0.04 ± 0.04	0.6 ± 0.8

^a BSAF in the presence of 2% AC.

^b Reduction factor in concentrations in biota on a lipid basis.

References

- Cornelissen G, Gustafsson Ö, Bucheli TD, Jonker MTO, Koelmans AA, Van Noort PCM. 2005. Extensive Sorption of Organic Compounds to Black Carbon, Coal, and Kerogen in Sediments and Soils: Mechanisms and Consequences for Distribution, Bioaccumulation, and Biodegradation. *Environ Sci Technol* 39: 6881.
- Escher BI, Hermens JLM. 2004. Internal Exposure: Linking Bioavailability to Effects. *Environ Sci Technol* 38:455A-462A.
- National Research Council. 2002. *Bioavailability of contaminants in soils and sediments: processes, tools and applications*; National Academies Press: Washington DC, USA.
- Ehlers LJ, Luthy RG. 2003. Contaminant bioavailability in soil and sediment. *Environ Sci Technol* 37:295A.
- Cornelissen G, Breedveld GD, Kalaitzidis S, Christanis K, Kibsgaard A, Oen AMP. 2006. Strong Sorption of Native PAHs to Pyrogenic and Unburned Carbonaceous Geosorbents in Sediments. *Environ Sci Technol*, 40: 1197-1203.
- Cornelissen G, Breedveld GD, Naes K, Oen AMP, Ruus A. Bioaccumulation of native PAHs from sediment by a polychaete and a gastropod: Freely dissolved concentrations and activated carbon amendment. *Environ. Toxicol. Chem., in press*.
- Oen AMP, Cornelissen G, Breedveld GD. 2006. Relation between PAH and black carbon contents in size fractions of Norwegian harbor sediments. *Environ Pollut* 141:370-380.
- Werner D, Higgins CP, Luthy RG. 2005. The sequestration of PCBs in Lake Hartwell sediment with activated carbon. *Wat Res* 39:2105-2113.
- Zimmerman JR, Ghosh U, Luthy RG, Millward RN, Bridges TS. 2004. Addition of Carbon Sorbents to Reduce PCB and PAH Bioavailability in Marine Sediments. Physicochemical Tests. *Environ Sci Technol*, 38:5458.
- Millward RN, Bridges TS, Ghosh U, Zimmerman JR, Luthy RG. 2005. Addition of Activated Carbon to Sediments to Reduce PCB Bioaccumulation by a Polychaete (*Neanthes arenaceodentata*) and an Amphipod (*Leptocheirus plumulosus*). *Environ Sci Technol* 39:2880.
- Zimmerman JR, Werner D, Ghosh U, Millward RN, Bridges TS, Luthy RG. 2005. Effects of dose and particle size on activated carbon treatment to sequester polychlorinated biphenyls and polycyclic aromatic hydrocarbons in marine sediments. *Environ Toxicol Chem* 24:1594-1601.

CONTAMINATION OF FJORDS BY URBAN RUN-OFF

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Abstract

Many active industrial point sources in Norway have been strongly reduced or eliminated the last decennia. Despite this improvement contaminant levels in sediments and marine biota have not be reduced to a similar extend. This might indicate that diffuse sources are an important source of contaminants. In the present study the contribution of run-off from urban areas to the contaminant load in fjords has been characterised.

As a result of the large seasonal variation in the run-off from paved urban areas, sampling and quantification is challenging. A time integrated sampling method has been developed using the "sedimentary record" of particles deposited in the urban drainage system. This method has been applied in studies in several towns around the Oslofjord. Results from the town of Drammen show surprisingly high levels of TBT, practically in all samples, whereas PCB, DDT, PAH and heavy metal levels showed a stronger local variation. The studies clearly emphasize the need for local source identification before remediation plans are put into force. They make it also possible to determine the level of improvement which can be achieved by remediation given the present active sources of contamination.

Introduction

In Norway serious contamination of marine sediments has been found in fjords and coastal areas. The Ministry of the Environment has identified the abatement of contaminated coastal, fjord and harbour sediments as a top priority for the coming years. Studies have been performed to develop remediation plans for highly contaminated fjords. To significantly reduce environmental risk, remediation involves large areas and consequently high costs. An important part of these studies has been to characterise active sources still contributing to the contamination of the fjord sediments. This study focuses on the characterisation of urban run-off in the town of Drammen as input for the remedial action plan for the Drammensfjord (Pettersen and Eek, 2005).

Materials and methods

The town of Drammen is situated where the Drammen river runs out into the fjord (Figure 1). This area was earlier highly industrialised, but today industrial activity has been strongly reduced. Despite this fact the sediments in the Drammensfjord are highly contaminated and there are consumption restrictions for cod-liver from this area as a result of high PCB levels. Diffuse sources are believed to be the main source of active contaminant transport to the fjord. The purpose of this study was to estimate the contaminant contribution from urban areas to the Drammensfjord.

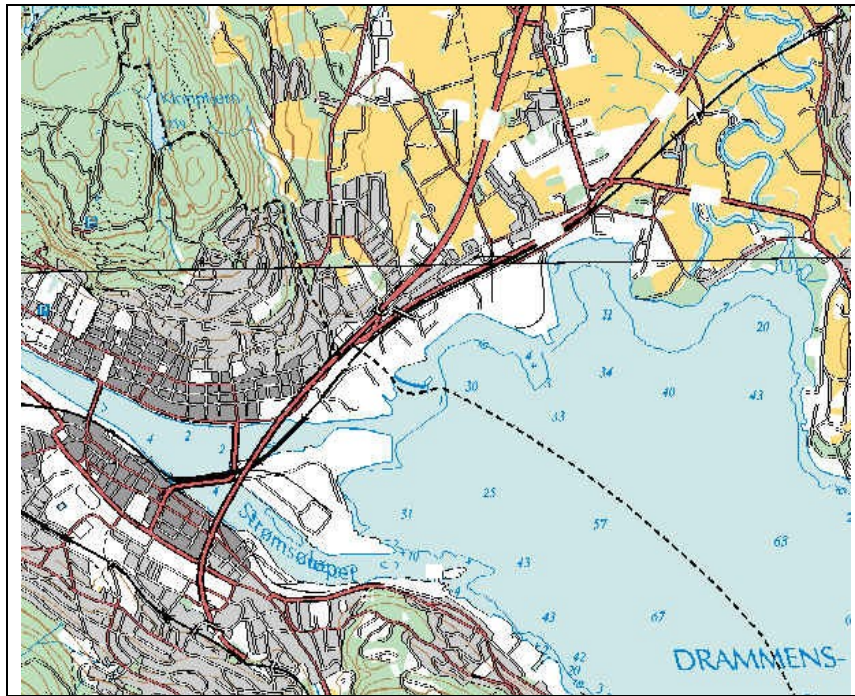


Figure 1. Map of the Drammen region showing coastal zone along the river and fjord.

Sampling methods

As a result of the large seasonal variation in the run-off from paved urban areas, sampling and quantification is challenging (Jantsch et al. 2006). A time integrated sampling method has been developed using the “sedimentary record” of particles deposited in the urban drainage system. 12 sediment samples have been collected from man-holes and pumping stations for surface run-off. Sampling locations close to the waterfront were selected in cooperation with the local technical division in Drammen to cover the expected variation in contaminant levels (Nesse, 2005).

Sediments in the drainage system were sampled by a small handheld grab sampler. The sampled material was described and transferred to pre-washed sample containers. An accredited analytical laboratory was used to determine the total content of metals, tin-organic compounds (TBT), polycyclic aromatic hydrocarbons (PAH) polychlorinated biphenyls (PCB), DDT, mineral oil and several brominated flame retardants as well as chlorinated paraffins. All results were normalised to the dry matter content of the sample.

Results

The results show large local variations in the concentration of the analysed components (Table 1). The concentrations of heavy metals are generally at a level corresponding to little to moderately contaminated according to the Norwegian classification system for sediment quality (SFT, 1997). PAH, PCB and DDT were found at significant contamination levels in only one to three samples. However all samples contained significant to high contamination levels of TBT. All analysed brominated flame retardants (not included in classification system) with the exception of HBCDD were detected in the surface run-off (Pettersen and Fjeld, 2005). Chlorinated paraffins were found in all samples at mean levels of 0.5 to 4 mg/kg for short chain and long chained paraffins respectively. Of the perfluorinated alkylated substances (PFAS) only PFOS (Perfluoro octane sulfonate) was detected at relatively low levels.

Table 1. Mean concentrations of organic and inorganic contaminants in sediment samples from the drainage system of the city of Drammen.

Compound	Units	Mean	Min.	Max.	n
TOC	% TS	3,38	0,54	16,6	13
As	mg/kg TS	5,0	1,8	8,1	12
Cd	mg/kg TS	0,48	<0,2	1,9	12
Cr	mg/kg TS	34	14	72	12
Cu	mg/kg TS	67	11	230	12
Hg	mg/kg TS	0,16	<0,05	0,53	11
Ni	mg/kg TS	27	11	43	12
Pb	mg/kg TS	39	16	91	12
Zn	mg/kg TS	184	84	540	12
Sn	mg/kg TS	6,9	<5	21	12
Mineral oil	mg/kg TS	1727	150	7300	12
PAH-16	µg/kg TS	777	50	2800	12
PCB-7	µg/kg TS	38	<3	260	12
TBT	µg/kg TS	20,6	4,7	57	12
DDT	µg/kg TS	2,4	<0,5	10,2	12
SCCP	µg/kg TS	552	215	1789	6
MCCP	µg/kg TS	3905	306	20500	6
Bisfenol A	µg/kg TS	16	<1	62	6
TBBPA	µg/kg TS	3,7	<1	10,6	6
HBCDD	µg/kg TS	<15	<15	<15	6
SUM BDE	µg/kg TS	156	0,75	593	6
PFAS	µg/kg TS	0,7	<0,2	1,2	2

TBT is Tributyl tinn

SCCP is short chained chlorinated paraffins

MCCP is medium chained chlorinated paraffins

TBBPA is tetrabrombisfenol A

HBCDD is hexabromocyclododecan

Sum BDE is the sum of tri-, tetra-,penta-, hexa-, hepta-, okta-, nona- and deca brominated difenyleters.

The detected component of PFAS is PFOS

Discussion

The quality of the sediment transported to the fjord system by surface run-off shows that heavy metal levels are not in conflict with the remediation objectives for the inner Drammensfjord (Pettersen and Eek, 2005). However substances earlier assumed to have distinct industrial sources like TBT, brominated flame retardants, and chlorinated paraffins are found at several locations in the drainage system and therefore seem to have diffuse sources related to urban activity. This can have implication for the outcome of remediation efforts in the fjord. The potential environmental impact of these compounds is poorly known. This limits the possibilities to include remediation objectives for sediment management plans.

The method used in this study allows quantifying and localising of potential sources of pollution based on a single sampling event. The samples give a time integrated value thereby reducing the uncertainty related to aqueous phase sampling. The method is especially applicable for compounds with a low aqueous solubility. These compounds are generally the main focus for remedial action to limit negative long term environmental effects.

References

Jantsch TG, Lindholm O, Hult F, and Strand KR. 2006. Presence of organic contaminants in runoff (in Norwegian). *Vann*. 41 (1): 41-55.

Nesse E. 2005. Urban run-off as potential pollution source from Drammen and Lier to the Drammensfjord (in Norwegian). MSc thesis, NTNU, Trondheim 80 p.

Pettersen A, and Eek E. 2005. Remedial plan for contaminated sediments in the Drammensfjord (in Norwegian). NGI-report: 20041208-3, Norwegian Geotechnical Institute, Oslo. 71 p.

Pettersen A. and Fjeld E. 2005. Environmental contaminants in the Drammen river. SFT report: TA2120, State Pollution Control Authority, Oslo. 24 p.

SFT. 1997. Classification of environmental quality in fjords and coastal water. SFT report TA-1467. State Pollution Control Authority, Oslo.



TITLE

Comparison of leaching methods for the estimation of bioavailability in two CCA-contaminated soils

Standardised leaching tests for soil testing
– results of work within ISO

Comparison of leaching of PAHs from two contaminated soils under varying hydraulic retention times using a column leaching test

Leaching tests in risk assessments of contaminated areas – a project within the Environmental Protection Agency's programme on "Sustainable Remediation" ("Hållbar Sanering")

Landfill leachate treatment in cold climate. Examples from Troms County, North Norway

SPEAKER

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Comparison of leaching methods for the estimation of bioavailability in two CCA-contaminated soils.

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Risk assessment of large areas of contaminated land is a delicate task and numerous methods are proposed for this purpose. The concept of the contaminant bioavailability has gained increasing importance in risk assessment compared to insufficient risk estimations based on total concentrations of pollutants. Site-specific bioavailability can be measured by rigorous chemical analysis in combination with batteries of ecotoxicological assays and mesocosm-studies. However, these are expensive and time- and personal demanding. Tiered approaches have been suggested and preferably in a first tier, cheaper screening methods such as leaching tests and single toxicological methods of less complexity and ecological relevance could be used. The methods in the present paper are all first tier methods aiming to define highlighted zones where elevated risk can be expected and where more efforts need to be put. That in terms of both qualitative and quantitative measures, using the more sophisticated methods of higher complexity and ecological relevance, in a second and possibly third tier.

The present study is a comparison between different methods aiming to estimate soil contaminants bioavailability. Two soils, one sandy and one organic soil, contaminated by chromium, copper and arsenic (CCA) from wood impregnation are used in the study. A series of leaching/extraction methods are compared for metal yield and biocompatible extracts are also characterized by two cytotoxicological bioassays, Neutral red incorporation assay (NRI) and screened for genotoxicity using the DRAG-assay. The following leaching/extraction methods are part of the study; Pressurized liquid extraction (PLE) with three different settings in temperature (50°C, 100°C and 150°C), batch leaching EN 12457-2, authentic pore water from lysimeters, leaching in 0.01M CaCl₂ and an in vitro gastrointestinal extraction relevant for oral intake of soil (IVG-E).

Complete chemical analysis are conducted of both soils and soil extracts including arsenic speciation since arsenic often is the limiting risk factor in CCA-contaminated soils. For further comparison with “true bioavailability”, a bioaccumulation study in earth worms is performed in glass test vessels filled the CCA-soils. The results will be compared and discussed in terms of methodology relevance, speed, costs and use practicality.

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STANDARDISED LEACHING TESTS FOR SOIL TESTING - RESULTS OF THE WORK WITHIN ISO

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Abstract

Leaching tests for soil are often used to assess the release of contaminants into water. Therefore ISO (International Organisation for Standardization) started the work for standardisation of leaching tests for soil and soil like materials for both inorganic and organic material. The work within the ISO Scientific committee includes at the moment 5 work items. There are two different batch leaching procedures as well as one column test and one batch test at different pH values. A guidance document for the choice of test method is also under development.

The aim has switched and is now to establish technical specifications (TS) instead of standards. The future procedure for the different work items will be discussed at the next meeting of the scientific committee in London in October 2006.

Introduction

Leaching tests are performed to investigate the release of compounds from solid matter into a liquid phase. Leaching tests for soil are often used to assess the release of contaminants into water.

Standardised test methods for leaching tests increase the trustworthiness of the results as well as they make results better comparable. ISO started the work for standardisation of leaching tests for soil and soil like materials for both inorganic and organic material.

Materials and methods

The establishment of new international standards is based on the scientific work of experts in the particular field. Scientific committees within ISO work out the draft standards that are circulated for comments. The comments are discussed within the scientific committee and necessary changes are made.

Results

The work within the Scientific committee includes at the moment 5 work items. There are two different batch leaching procedures as well as one column test and one batch test at different pH values. The work with test 21268-4 has started later, therefore only a first draft is available. Also the guidance document is at a draft stage.

ISO/TS 21268-1	Soil quality - Leaching procedures for subsequent chemical and ecotoxicological testing - Part 1: Batch test using a liquid to solid ratio of 2 l to 1 kg
ISO/TS 21268-2	Soil quality - Leaching procedures for subsequent chemical and ecotoxicological testing of soil and soil materials - Part 2: Batch test using a liquid to solid ratio of 10 l/kg dry matter
ISO/TS 21268-3	Soil quality - Leaching procedures for subsequent chemical and ecotoxicological testing of soil and soil materials - Part 3: Up-flow percolation test
ISO/TS 21268-4	Soil quality - Leaching procedures for subsequent chemical and ecotoxicological testing of soil and soil materials - Part 4: Influence of pH on leaching with initial acid/base addition
ISO/DIS 18772	Soil quality - Guidance on leaching procedures for subsequent chemical and ecotoxicological testing of soils and soil materials

The main difference between the batch tests is the liquid-solid (LS) ratio, which is 2 and 10, respectively. The procedure assumes that equilibrium or near-equilibrium is achieved between the liquid and solid phase during the test period. The third batch test is performed at different pH values. Important steps in the batch test procedure as proposed now are:

- sample pre-treatment and dry weight determination
- leaching step: water with low Calcium Chloride content, usually 24 hours
- Separation step: centrifuge or filter
- Determination of pH, redox, turbidity (or DOC and conductivity)
- Analysis of leachate and reporting.

The proposed column test procedure also includes several steps:

- sample pre-treatment and dry weight determination
- packing of the cleaned column and connection of column to the system with filters and pump
- pumping of water at about 15cm/d
- collecting of leachate at 7 different LS from 0.1 to 10
- solid liquid separation of leachate: centrifuge and filter
- Analysis of leachate and reporting.

Discussion

The work in the scientific committee aimed at first in establishing ISO standards. As the methods evolved and agreed on in the scientific committee have not been validated, the aim is now to establish technical specifications (TS). The future procedure for the different work items will be discussed at the next meeting of the scientific committee in London in October 2006.

Although technical specifications do not have the same status as ISO standards, they are helpful as they contain important information on how to perform tests. Following specifications reduces the risk for inaccurate results due to mistakes in the testing procedure. As the work with most of the standards has resulted in well evolved procedures, it should be possible to establish technical specifications within reasonable time, although some questions like the solid-liquid separation have to be addressed more.

References

ISO/FDIS TS 21268-1:2006 (E), ISO copyright office, Geneva, Switzerland
ISO/FDIS TS 21268-2:2006 (E), ISO copyright office, Geneva, Switzerland
ISO/FDIS TS 21268-3:2006 (E), ISO copyright office, Geneva, Switzerland
ISO/FDIS TS 21268-4:2006 (E), ISO copyright office, Geneva, Switzerland
ISO/DIS 18772, ISO copyright office, Geneva, Switzerland

COMPARISON OF LEACHING OF PAHS FROM TWO CONTAMINATED SOILS UNDER VARYING HYDRAULIC RETENTION TIME USING A COLUMN LEACHING TEST

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Abstract

Leaching of polycyclic aromatic hydrocarbons from two soils, collected at a gasworks site and a former wood impregnation site, was studied using a column leaching method with on-line solid phase extraction. The time period required to reach equilibrium in leaching experiments ranges from minutes to days, or even years, depending on what processes that control the release of the contaminants and the type of sorption domain from which the contaminants are being released. By varying the hydraulic retention time (HRT) of the system, the time needed to obtain equilibrium concentrations in leachates was investigated. The purposes of this study were i) to investigate if leachate concentrations from field contaminated soil, obtained by column leaching, can reach equilibrium concentrations at very short contact times and ii) to demonstrate that leaching of PAHs from aged contaminated soils can show significant differences in leaching behaviour and leachable concentrations.

Introduction

Polycyclic aromatic hydrocarbons (PAHs) can exhibit a wide range of carcinogenic and toxic effects and are hence of great concern with respect to both human health and the environment. High concentrations of PAHs can be found at former industrial sites, particularly those associated with petroleum products, gas production and wood impregnation. At risk assessment of contaminated soils it is crucial to correctly estimate the mobility of the contaminants and their potential to leach to groundwater and surface water. Leaching tests can be an important tool for this matter, however, agitated batch leaching experiments may be unsuitable for determining the leaching of PAHs, due to colloid- and particle-facilitated release of PAHs caused by the abrasion that occurs when the soil sample is shaken or swirled (Bergendahl and Grasso, 1998; Hansen et al., 2004). Hence, column leaching tests may provide a more suitable method of assessing the leaching of PAHs. Furthermore, leaching data from column leaching experiments can provide information about which processes that control the release rates of the contaminants.

An important parameter when conducting leaching experiments is the contact time between the material that is undergoing leaching and the leachant. When batch experiments are used to determine leaching, steady-state concentrations can only be achieved if equilibrium is reached between the solute and the sorbed fraction of contaminants. However, when leaching is assessed with column experiments, stable, constant leaching concentrations can be obtained both when distribution equilibrium processes control leaching as well as when leaching is limited by mass transfer or any other process. To evaluate whether steady-state leaching is due to equilibrium conditions or mass transfer limitations, the hydraulic retention time (HRT) may be varied. During non-equilibrium conditions the concentration in the leachate will increase as the HRT is prolonged. However, the effluent concentration will be independent of the HRT if the soil column, from bottom to top, is subject to equilibrium conditions.

The objective of this study was to investigate the HRT needed for column leaching to obtain distribution equilibrium concentrations of PAHs in the leachate from two PAH contaminated soils. In addition, this study also aims to demonstrate that heavily contaminated soils, that have been aged, may show significantly differences in leaching behaviour and the leached concentrations can differ in order of magnitudes.

Materials and Methods

Field contaminated soil was collected from a former wood impregnation facility in the northern part of Sweden and from an old gasworks plant located in central Stockholm. The sites are heavily contaminated with PAHs from spills of creosote and coal tar. Soil characteristics are given in Table 1 for the two soils studied. The leaching experiments were conducted with a column leaching method where a flow of sterile water was pumped through a soil column. Quantification of the contaminants was enabled by solid phase extraction, (SPE). For a more detailed description of the experimental set-up and the analytical methods used, see Enell et al. (2004). Three columns were filled with the creosote contaminated soil (approx. 0.5 kg) to study leaching at three hydraulic retention times (HRTs); 0.3, 0.5 and 1.0 h. For the coal tar contaminated soil, collected at the gasworks site, five columns were used for the leaching with the same HRTs applied as for the creosote contaminated soil; experiments conducted at 0.3 and 0.5 h were performed in duplicates, while one column was used to conduct the leaching corresponding to a HRT of 1.0 h. Flow rates were in the range of 0.1 to 0.5 L h⁻¹. The leaching behaviour of fluorene, phenanthrene, anthracene, fluoranthene and pyrene was investigated. Initial amounts (shown in Table 1) were determined by an extraction method using acetone and ultrasonication described in details in Soares et al. (2003).

Table 1. Characteristics of the two soils studied together with total extractable amounts (as mean values and standard deviations, n = 2) of the PAH studied in coal tar and creosote contaminated soil.

Parameter	Coal tar cont. soil	Creosote cont. soil	PAH	mg kg ⁻¹	
				Coal tar cont. soil	Creosote cont. soil
Fraction of organic carbon, f _{oc}	0.16	0.03	FLU	7 ± 1	110 ± 12
CEC (mmol _e /kg)	62.5	24.4	PHE	61 ± 3	240 ± 43
pH water (1:10)	7.62	5.64	ANT	60 ± 3	226 ± 101
Water content (% w/w)	6.7	8.8	FLA	221 ± 43	1066 ± 36
Density kg/L (d.w.)	1.36	1.06	PYR	177 ± 34	521 ± 24
Composition of the soils	(%)	(%)			
Coarse sand >0.2 mm	54	62			
Fine sand (0.02-0.2 mm)	20	23			
Silt (0.002-0.02 mm)	21	10			
Clay (<0.002 mm)	5	5			

Results

The two soil sample studied showed very different leaching behaviour. The leached concentrations of all studied PAHs, from the soil contaminated with coal tar, showed rapid establishment of steady-state leaching with accumulated liquid to solid (L/S) ratio, while leached concentrations, from the soil contaminated with creosote, decreased linearly with increasing L/S ratio. As an example, leached concentrations of fluorene, as a function of accumulated L/S ratios for the two soils studied are shown in Figure 1. Mean values of leached steady-state concentrations from the coal tar contaminated soil at the different HRTs studied were calculated. Mean concentrations for fluorene, anthracene and pyrene vs. HRT are shown in Figure 2.

During the course of the experiment, less than 1 % of the total initial amount of the studied PAHs in the coal tar contaminated soil was leached, while considerably higher amounts (e.g. approx. 80 % of the lighter PAHs) were leached during the same time from the soil collected from the former impregnation facility (Table 2).

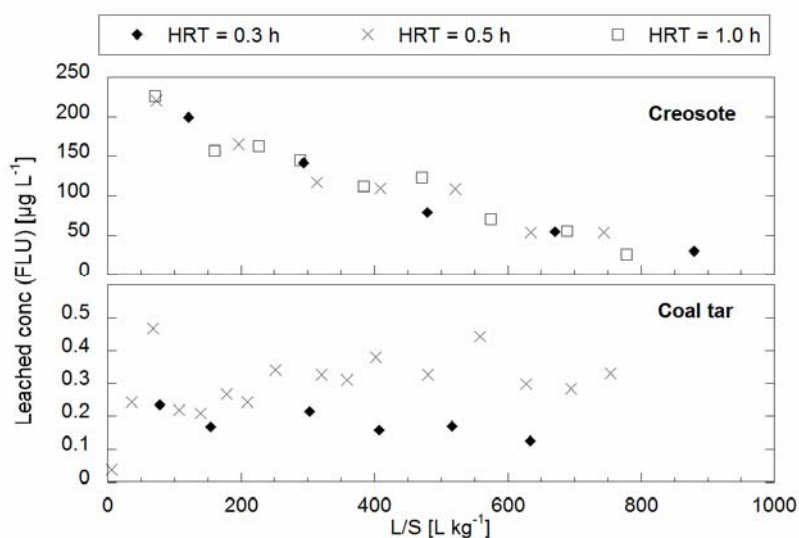


Figure 1. Leached concentrations of fluorene (FLU), at different hydraulic retention times (HRT) for the two studied soils vs. accumulated liquid to solid (L/S) ratios.

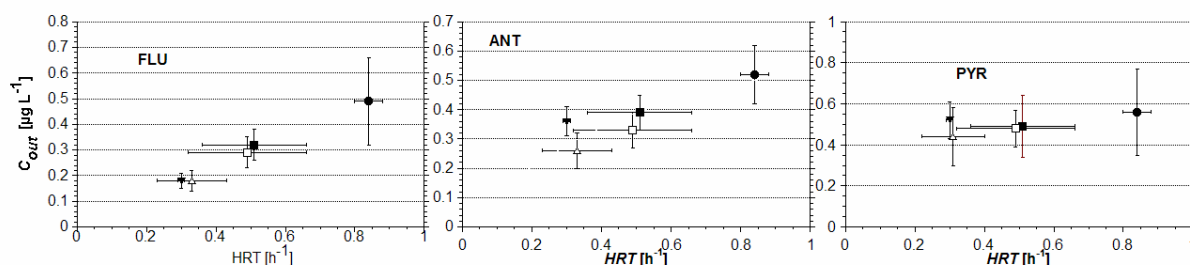


Figure 2. Calculated mean values of leached concentrations at various *HRTs* (standard deviations of leached concentrations and retention times are shown as error bars).

Table 2. Leached accumulated amount/kg soil at approx. L/S-ratio 700 L/kg and leached fractions as percentage of initial amounts in soil together with Mean values of leached steady state PAH concentrations, for gasworks soil C_{mean} , calculated using leached concentrations during the L/S period of ≈ 70 -700 L/kg, and initial leached concentrations, C_{max} , for the creosote contaminated soil. For comparison solubility equilibrium concentrations, S_{eq} , are given for coal tar and creosote, calculated by Raoult's law (DIN, 1998).

PAH	Coal tar contaminated soil				Creosote contaminated soil			
	Leached acc. amount	% leached of total	C_{mean}	S_{eq}	Leached acc. amount	% leached of total	C_{max}	S_{eq}
	(HRT=0.5 h at L/S=700)	(acc. L/S=700)	(HRT=0.5 h)	Coal tar MW of 253 g mol ⁻¹	(HRT=0.5 h at L/S=700)	(acc. L/S=700)	(L/S=90 kg L ⁻¹)	Creosote MW of 137 g mol ⁻¹
	[mg/kg]		[µg/L]	[µg/L]	[mg/kg]		[µg/L]	[µg/L]
FLU	0.220	1.84	0.31 ± 0.07	220	85	77	215 ± 14	440
PHE	0.955	0.46	1.36 ± 0.34	190	78	32	166 ± 8	870
ANT	0.246	0.64	0.35 ± 0.08	53	14	6	26 ± 2	51
FLA	0.381	0.18	0.59 ± 0.19	18	46	4	85 ± 4	68
PYR	0.271	0.16	0.42 ± 0.15	15	19	4	35 ± 2	47

Discussion

Initially leached concentrations from the creosote contaminated soil were very high and the increased contact time had no effect on leachable concentrations. When compared with calculated solubility equilibrium concentrations for creosote, using Raoult's law and an average molecular weight of 137 g mol^{-1} for creosote (DIN, 1998) the obtained leached concentrations of PAHs were found to be in the same range (Table 2). Hence, it was concluded that the leaching of PAHs from the creosote contaminated soil was controlled by solubility rather than soil/water distribution equilibrium or desorption kinetics.

On the other hand, leached concentrations of PAHs from the coal tar contaminated soil, were rather low and distinctively lower than estimated solubility equilibrium concentrations for all of the PAHs studied (Table 2). Consequently, the leaching was most likely governed by desorption equilibrium conditions or mass transfer limitations. The mean concentrations calculated from the leached steady state concentrations, increased with increasing HRT for fluorene, phenanthrene and anthracene. For fluoranthene the concentration approached a constant value at HRTs longer than 0.5 h, while for pyrene no increase in concentration was seen. This indicates that the leaching of the lighter PAHs was controlled by mass transfer limitations under the prevailing conditions, while the effluent concentrations of the heavier PAHs were close to their respective distribution equilibrium concentrations despite the very short contact time applied.

In conclusion, the effluent concentrations of several of the studied PAHs were surprisingly found to be close to distribution equilibrium concentrations, despite rather high pore water velocities which generated short HRTs. The two studied soils differed significantly in their leaching behaviour although initial concentrations were similar and both soils were aged. Only a few percent of the initial amount of PAHs could be leached from the soil collected from the old gasworks site while considerably larger fractions of the PAHs were leached during the same time from the soil collected from the former impregnation facility. This supports the conclusion that risk assessment should be based on actual leaching concentrations rather than total soil residual concentrations.

References

- Bergendahl, J., Grasso, D., 1998. Colloid generation during batch leaching tests: mechanics of disaggregation. *Colloids and Surfaces a-Physicochemical and Engineering Aspects* 135, 193-205.
- DIN 1998. DIN V 19736. Soil quality -Derivation of concentrations of organic pollutants in soil water. *Bodenbeschaffenheit, Ableitung von Konzentrationen organischer Stoffe im Bodenwasser*, Deutsches Institut für Normungen.
- Enell, A., Reichenberg, F., Warfvinge, P., Ewald, G., 2004. A column method for determination of leaching of polycyclic aromatic hydrocarbons from aged contaminated soil. *Chemosphere* 54, 707-715.
- Hansen, J. B., Grøn, C., Hjelmar, O., Asmussen, O., Klem, S., Mizutani, S., Gamst, J., Wahlström, M., Håkkanson, K., Breedweld, G. 2004. Leaching tests for non-volatile organic compounds - development and testing. Hørsholm, Nordtest, DHI Water & Environment.
- Soares, A., Guieysse, B., Mattiasson, B., 2003. Biodegradation of nonylphenol in a continuous packed-bed bioreactor. *Biotechnology Letters* 25, 927-933.

**Leaching tests in risk assessments of contaminated areas
A project within the Environmental Protection Agency's programme on
"Sustainable Remediation" ("Hållbar Sanering")**

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Background

Risk assessments of contaminated sites include both the direct impact upon human health when exposed to contaminated soil and the risk for leaching of contaminants to ground water and surface water. Conventionally, decisions are based upon measurements of the total content of contaminants in soils, usually combined with analysis of groundwater or surface water. However, it is well known that for both inorganic and organic compounds in soils only part of the total content of contaminants may be available for leaching to groundwater or surface water, as well as for uptake by humans.

During the last 15 years, leaching tests for inorganic compounds have been developed and standardized for assessing the risk for leaching to ground and surface water. These leaching methods have primarily been developed for waste materials, but they have to some extent been adjusted for use on contaminated soils. Similarly, test methods have been developed for assessment of the availability of soil contaminants for impact upon humans and the soil environment. However, until now the use of leaching and availability tests for risk assessments of contaminated sites has been limited.

In this project a preliminary methodology for the use and interpretation of leaching tests in risk assessments for contaminated sites is presented. Focus is on inorganic compounds although the applicability of different tests is discussed for both inorganic and organic compounds.

The methodology is based on an evaluation of a set of leaching tests on Swedish and Danish soils including modelling of leaching processes. The results show that there are uncertainties as how results from standardized leaching tests should be used in risk assessments. To develop the methodology proposed in the project into a more practical advice in a detailed guidance-document, more work is required on leaching tests on contaminated soils and their interpretation in risk assessments of contaminated areas.

In the follow-up project, which started in August 2006, different methods to evaluate leaching will be tested by applying them to contaminated sites in a case study. The aim is to verify the applicability of different methods for both inorganic and organic compounds and to evaluate the practical use with respect to the complexity of the methods, the data requirement and the uncertainty associated with the results.

LANDFILL LEACHATE TREATMENT IN COLD CLIMATE - EXAMPLES FROM TROMS COUNTY, NORTH NORWAY

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Abstract

Leachate from municipal landfills and composting of wet organic waste contributes to the pollution of watercourses in North Norway. The experience on low-cost natural systems for leachate treatment in the northernmost counties is limited. At Buktamoen municipal landfill the leachate is treated in an aerated lagoon followed by infiltration in a glaciofluvial deposit. Water emerging in the front of the deposit is at last treated in an open constructed wetland. Since this is a low loaded multistage system, we expect considerable removal for the total systems also during wintertime. Probably this treatment will not be sufficient for leachate from composting of organic waste with very high content of organic materials.

Introduction

In Norway many of the municipal sanitary waste (MSW) landfills were located without a liner in closed gravel pits in large glaciofluvial or fluvial deposits. These deposits with thickness of 10-40 m are located in many of the main valleys. Up to a few decades ago, such landfills were established without concern of the leachate. Although natural attenuation processes in sandy aquifers may reduce or retain many of the constituents in leachate, it is often observed that polluted groundwater may cause an environmental hazard when it reaches open water, especially downstream small aquifers. At some sites, the leachate is even collected and deliberately infiltrated, hoping the infiltration to break down the organic material in the leachate. The consequence is in many cases that the leachate, depleted in oxygen, emerges at the base of the deposit or in a river without any reduction of organic matter, but with a substantial content of dissolved iron and other undesirable components. Some times a quite unexpected pollution is seen several hundred meters away from the source of contamination.

New regulations require that landfills must be built with a liner to control diffuse losses of leachate; and leachate treatment is compulsory. On-site "high-tech" leachate treatment facilities are often avoided due to high construction and operation costs. There are more than 50 on-site systems using natural systems for leachate treatment in Norway, but experiences with such technology in the northernmost counties are limited. In Troms County there are several examples of these problems, two of them are presented in this paper, and a few more will be mentioned in the oral presentation. Even though Troms County is sparsely populated, the pollutions can influence local groundwater supply and reduce the quality in vulnerable ecosystems as e.g. salmon rivers. Thus, some action should be taken to reduce the problem. However, these municipalities are often small, and their economy does not allow large investments in abandoned landfills. In addition, the cold climate in the inland of North Norway does not favour extensive low-cost solutions as biodegradation in open ponds or constructed wetlands. The main task is therefore to optimize a solution to an acceptable cost and environmental standard based on local resources. A "perfect" leachate treatment plant can hardly be achieved.

Materials and methods

Buktamoen landfill in Målselv municipality

This 45 da landfill was established in 1969 and is situated in a glaciofluvial terrace of >30 m thickness. Leachate from the newest part of the landfill (1996) with liner was collected and initially infiltrated in a pond, on the top of the deposit. After a few years, the water emerged at the front of the terrace with a bad odour and a lot of precipitated iron. Seepage contaminated the marshy land at the front of the terrace and probably also the river Målselv a few hundred metres away. Due to this, a collection ditch was dug, and the leachate was pumped back to the infiltration pond. Consequently, the leachate was sent into an everlasting loop that was continuously supplied with new leachate from the landfill. As a result, the amount of infiltrated leachate increased for each year, and in 2002 was nearly 100,000 m³/y. The amount of leachate directly from the landfill was only 5,000 m³, and this amount is about the same as can be calculated based on annual precipitation, expected evaporation, and supposed consumption of water in the landfill.

The chemical composition of leachate shows high variation, but average content found in 2002 for instance, was as follow:

COD: 2,600 mg/l Tot-N: 390 mg/l
BOD₍₅₎: 1,450 mg/l^{*} Tot-P: 3.6 mg/l

In 2004 a new on-site treatment systems was designed based on local conditions and available area. To solve the pollution problem at the front of the terrace, the first step was to break the loop of pumped water. A ditch was dug from the pumping station in front of the terrace to an old riverbed. This riverbed was slightly widened and supplied with 3 thresholds as shown in figure 1, and thus transformed into a 300 m long surface flow treatment wetland.

To solve the anaerobic situation, the leachate has to be aerated before infiltration. One of the two old infiltration ponds has been transformed to a 2.5 m deep aerating lagoon with a volume slightly less than 600 m³. The aeration is based on 2 units of 5 HP propeller aerators, each mounted on a polyethylene pontoon floating on the surface.

Two new open infiltration basins in parallel, each 600 m², have been built closer to the edge of the terrace to avoid the area of anaerobic conditions from the old infiltration system.

The new system was finished by August 2006. The total cost for this treatment system amounted to about €140,000.

Skibotn landfill and composting plant in Storfjord municipality

The abandoned Skibotn municipal landfill established in a shallow gravel pit in a glaciofluvial terrace was closed more then 25 years ago. Leachate seeps to the groundwater in a very large area without any collection system. In this terrace, there are only a few meters of coarse-grained gravel over some silty layers. No contamination was reported as long as the landfill was used, and the deposit was supposed to give the necessary groundwater protection.

About five years ago, a plant for composting wet organic waste was established upon the old landfill site. The leachate form the compost together with some storm water run-off was infiltrated in a basin in the coarse-grained layer of the deposit. Then outflow of water coloured by precipitated iron was observed at the base of the small terrace close to the landfill site, as well as in the Skibotn River several hundred meters away. No contamination was observed in between the terrace and the river. Using a mechanical

digger, however, leachate was found in coarse-grained fluvial material a few meters below the surface in the alluvial plain in this area. The transport to the river probably takes place in old subducted riverbeds in the plain.



Figure 1. Wetland and threshold under construction. The threshold has an impermeable core.

Analyses of the leachate emerging at the base of the terrace strongly indicate that the pollution originates from both the landfill and the infiltration basin for compost leachate. Only the seepage from the compost can be easily collected, and the company running this plant aims at cleansing this part of the contaminant. About 6,000 m³ of water are infiltrated per year, but the amount is expected to increase to 10,000 m³/year. The content of organic matter is very high, but also the content of phosphorus is quite high compared to leachate from municipal landfills.

COD _{Cr} : 65,600 mg/l	Tot-N: 2,370 mg/l
BOD ₍₅₎ : 42,400 mg/l	Tot-P: 408 mg/l

Oxidation of biological matter by aeration in ponds with air temperatures as low as -20 °C is hardly possible, and thus other possibilities were searched for. The possibility of utilizing the heat from the composting process was discussed. The compost heaps are covered with sheets of plastic and supplied with air from pipes in the concrete floor. The heated air could be used to warm up a tent over aeration ponds, but as the air will be depleted in oxygen a heat exchanger had to be installed. The total cost, the technical complexity, and some uncertainty of the whole effect finally stopped this project.

The use of an SBR reactor before the infiltration was also planned. An offer for an SBR reactor for 10 m³/d and BOD₍₅₎ 25 000 mg/l was approx. NOK 2,1 million (€240,000) at the

end of 2005. However, when the higher concentrations of BOD₍₅₎ and COD was presented neither the company that first favoured this solution nor any other companies offered a SBR reactor as an answer of the tender inquiry.

Only one offer for treatment of leachate was given in the competitive tendering in February 2006. A Finnish company proposed a treatment plant based on a patented technology called *Closed Electro Flocculation*. This technology is said to reduce the main pollution components of this leachate by 99.9 %, but the cost for the complete installation will be as much as €500,000, and some additional costs will probably accrue. The owner of the compost plant hesitates to use this amount of money.

The possibility of using a membrane filter (reverse osmosis) could also be discussed, but the high organic content will probably tend to block the tiny holes and result in a large amount of dilute sludge that should be taken care of.

Consequently, the final solution of the leachate treatment is still not found.

Results and discussion

There are not available data on treatment performance on the multistage system at Buktaemoen so far. We can anticipate that the low winter temperature significantly reduces the removal efficiency of each of the 4 treatment stages. From the literature, there are a lot of data showing that extended aerated lagoons can cope with a wide range of leachate flows and strengths, including shock loads due to the large lagoon volume. Lagoons with similar equipment and loading are widespread in Norway. Monitoring data from some of these systems show that low winter temperatures of the leachate in the lagoon (<5°C) reduces the efficiency for removal of BOD, COD and NH₄-N. High nitrification and BOD removal (>75%) can be achieved in periods with water temperature >10°C if lack of phosphorus does not limit the process. Slow rate soil infiltration of wastewater in open ponds has performed well in cold climate provided adequate pre-treatment and loading rates. Free water flow constructed wetlands have been used successfully for secondary and tertiary treatment of effluent from aerated lagoons in Sweden, UK and USA. Although each treatment stage can be influenced by low temperature, we do not expect freezing to cause hydraulic malfunction. Since this is a low loaded multistage system, we expect considerable removal for the total systems also during wintertime.

The substitution of the aerated lagoon by a sequential batch reactor (SBR) at Buktaemoen probably would double the costs, and the cost for a membrane filter is stipulated to NOK 3 million (approx. €360,000). During winter, the aeration would probably be more effective by use of an SBR plant with an insulated reactor, but based on the cost differences the environmental county authorities have accepted the solution presented above at Buktaemoen.

Conclusion

Use of multistage leachate treatment systems consisting of aeration lagoons succeeded by sedimentation, soil infiltration, and use of polishing ponds, wetlands or peat infiltration can be alternative low cost options for small landfills in cold climate regions if area and porous soil types are available. The loading rates must be low since cold winter climate will restrict the biological removal processes. Such MSW leachate treatment systems have been established in the north of Norway, but so far experiences are limited regarding performance of removal and the need of operation and maintenance.



TITLE

SPEAKER

Remediaiton of TBT contamination at Vuosaari harbour in Helsinki

Ari Püspanen, Rambøll Finland

Guidance manual for screening and selecting technologies for remediation of chlorinated solvents in Sweden

Peter Englöf, SWECO VIAK AB

Remediation due to creosote-contamination in wetlands in Råde, Norway

Gunnar Broenstad, Multiconsult AS

Environmental solutions for military shooting ranges in Norway and Finland

Sara Kajander, Construction Establishment of Finnish Defence Administration

Risk communication concerning contaminated land

Johan Asplund, WSP Environmental

REMEDICATION OF TBT CONTAMINATION AT VUOSAARI HARBOUR IN HELSINKI

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Abstract

In the spring of 2003, when the dredging of clay masses of the becoming Vuosaari Harbour was about to start, elevated concentrations of tributyltin (TBT) were detected in the seabed. Masses planned to be dredged could not be dumped in the sea dumping area and the dredging operation could not proceed. To continue the work a method for reconditioning the seabed had to be developed. The overall solution consists of three main phases. First, the contaminated area was isolated from the surrounding sea area by protective embankments and a silt curtain structure. Next, the contaminated sediment was removed from the seabed by decontamination dredging carried out by accurate dredging techniques. Finally, the contaminated sediment transferred to the filling area will be mass stabilized and utilized as the harbour's field structures. By this procedure TBT will be permanently isolated from the aqueous environment. The removal of TBT from the seabed has been a seamless part of the Vuosaari Harbour's construction work.

Introduction

The Vuosaari Harbour

The becoming Vuosaari Harbour is situated in the Southern Finland on the coast of the Gulf of Finland. The Vuosaari Harbour Project aims to replace the cargo operations in two existing main harbours in the Helsinki metropolitan area. The existing harbours are located near the centre of Helsinki. The new Vuosaari Harbour will be built on the site of a former shipyard, near to the residential area of Vuosaari, which is the most eastern suburb of Helsinki. The new Vuosaari Harbour will be the largest cargo harbour in Finland. The total quay length will be 3600 metres, with 20 berths (Fig. 1). More information can be found from the Vuosaari Harbour Project's internet pages, www.vuosaarensatama.fi.



Figure 1. The becoming Vuosaari Harbour.

The problem

The construction work for the Vuosaari Harbour began in January 2003. Of the area comprising Vuosaari Harbour, about 90 hectares will be filled from the sea. Before filling, the clay (the soft soil) of the sea bottom has to be removed by dredging. In addition, water depth adjacent to the quays have to be increased by dredging to fulfil the rated safe clearance depth (Fig. 2).

In the spring of 2003, when the dredging was about to start, elevated concentrations of tributyltin (TBT) were detected in the seabed of the Vuosaari harbour area. The finding launched comprehensive sediment studies, which showed that TBT concentrations in the surface sediments outside the former Vuosaari shipyard, and in a small area also in a deeper sediment layer, were so high that sediment to be dredged could not be dumped in the sea dumping area and the dredging operation could not proceed in this area. The concentration of TBT was highest in a 20 hectares area outside the former Vuosaari shipyard (areas TBT1 and TBT2 in Fig. 3). The highest TBT concentration analysed was about 15000 µg/kg dw (normalised). Elevated concentrations of polychlorinated biphenyls, PCBs, were also found in sediment of the most TBT-contaminated area. On an average the concentration of triphenyltin (TPhT), another organic tin compound used for same purposes as TBT, was only about 15 % of the concentration of TBT in the sediment samples analysed. The sediments of the north-eastern part of the becoming harbour area proved to be practically clean and the sediments of the new navigation channel to Vuosaari Harbour proved to be uncontaminated.

To enable continuation of the construction work the environmental authority, Uusimaa Regional Environment Centre, allowed dredging to be continued outside the contaminated area. The TBT-containing surface sediment of the Niinilahti Bay was peeled by careful grab bucket dredging and the peeled sediment was transferred to a temporary storage basin in Käärmeniemi (Fig. 3).

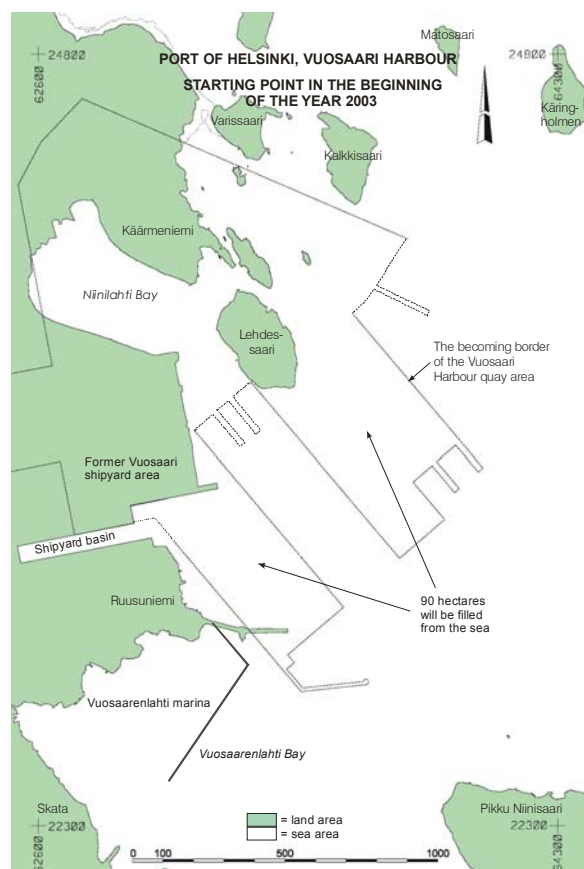


Figure 2. The Vuosaari Harbour area. The starting point in the beginning of the year 2003. Land areas are green and sea areas white.

After this operation the clean clay masses were dredged from the Niinilahti Bay and these masses could be dumped in the sea dumping area. But, based on the data collected, in the area of 75 hectares there was still left surface sediments which were considered not suitable for sea dumping because of the high TBT concentrations (mainly over 200 µg/kg dw (normalised)). The sediment analyses indicated that there was 97,6 kilograms of TBT in the seabed in this area of 75 hectares.

Materials and methods

The Port of Helsinki, in co-operation with experts from various fields, developed a method for reconditioning the seabed outside the former Vuosaari shipyard, which allowed TBT to be removed from the bottom sediments in an environmentally safe manner. The overall solution consists of three main phases.

The first phase

First, the contaminated area was isolated from the surrounding sea area by permanent protective embankments and a temporary silt curtain structure (Fig. 3). The function of the protective embankments and the silt curtain structure was to prevent escaping and spreading of suspended solids and TBT bound to these solids outside the isolated area.

The slightly TBT-containing surface sediment of the area needed for building the protective embankments was again peeled by careful grab bucket dredging and the peeled sediment was transferred to the same temporary storage basin in Käärmeniemi as the surface sediments of the Niinilahti Bay. The protective embankments were built of combined blasted rock. The silt curtain material was woven geotextile (polypropylene) with the strength of $\geq 70/120$ kN to each direction. The length of the silt curtain was 650 meters and the height 10-12 meters. The silt curtain structure was anchored to its position at 25 meters intervals. There was a ship gate (width 50 m) in the silt curtain structure.

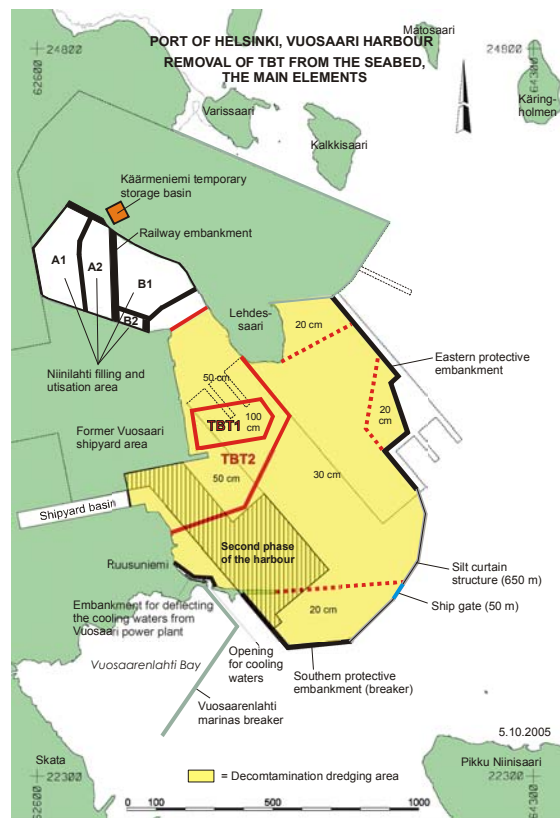


Figure 3. Removal of TBT from the seabed, the main elements of the process. The yellow area is the decontamination dredging area. The numbers on the yellow area indicate the minimum decontamination dredging depth.

The cooling water discharge from the Vuosaari gas power plants was deflected by an additional embankment outside the isolated area. The Niinilahti Bay, which was emptied from the clay masses, was prepared to receive the contaminated sediment to be dredged from the isolated area. The Niinilahti Bay was filled by sea sand to the level of -4,0 m. The embankments of the Niinilahti filling and utilization area were built of combined blasted rock and the embankments were covered by a sealing layer of moraine. The first phase was completed in the end of April 2005.

The second phase

Secondly, the TBT-contaminated sediment, about 450000 m³ (as consolidated sediment), was removed from the seabed by decontamination dredging carried out by accurate dredging techniques and transferred to the Niinilahti filling and utilization area (Fig. 4). The initial decontamination dredging was done by a special bucket designed for this task. These masses were transferred to the Niinilahti area by barges. For finishing up the work suction dredging was used to remove the loosened, possibly TBT-containing matter from the seabed. The suction dredged material was pumped through a pipeline to the Niinilahti area. Excess water originating from the suction dredging and pumping was removed from the Niinilahti area through a water box, which allowed only the clear surface effluent to flow out to the basin B2, and then through the moraine covered outermost embankment of the basin B2 to the isolated sea area. The suction dredging work had to be done at intervals because of the slow infiltration through the outermost embankment. According to the water quantity and quality studies the loss of TBT from the Niinilahti area was calculated to be very small (in any case < 0,1 kg).

The result of the decontamination dredging was controlled according to specific quality verification (i.e. sediment sampling and analysing) program approved by the environmental authorities. The second phase was carried out in May - October 2005.



Figure 4. Decontamination dredging in progress (June 2005).

The third phase

Finally, the TBT-contaminated sediment will be mass stabilized and utilized as the harbour's field structures in the Niinilahti filling and utilization area. The TBT-contaminated sediment is deposited as a layer of about five metres in thickness on the bearing fill layer of sea sand (Fig. 5). The sediment is mass stabilized by mixing Portland cement in it (Fig. 6). The amount of cement used is 130 kg/m³ of sediment. The stabilized sediment is covered first by a one meter high work embankment and later by a higher pre-loading embankment. In this way, a load-carrying harbour structure can be obtained. The quality of the stabilized structure is monitored according to quality control program approved by the environmental authorities.

The stabilized layer will be covered with a drainage layer and a surface layer, with an asphalt layer on top. The stabilized structure will be covered by a system of drainage pipes to minimize water pressure to the structure. The becoming harbour area between the Niinilahti area and the sea area will be filled by sea sand and the heavy background embankments of the quays. The outermost quay wall will be a concrete structure. Thus the Niinilahti filling and utilization area will be thoroughly encapsulated and permanently isolated from the sea area. The stabilized TBT-contaminated sediments will replace sea sand or blasted rock that otherwise would be needed in constructing the Niinilahti area.

The mass stabilization work started in December 2005 and the work will continue until March 2007. The drainage and surface layers as well as the drainage pipe system will be built after the pre-loading embankment is removed.

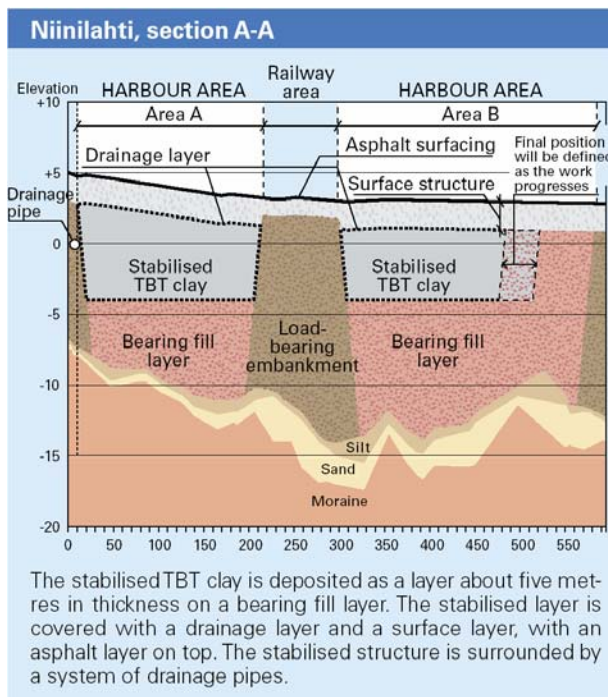


Figure 5. Principle of the structures of the Niinilahti filling and utilization area. Sea sand was used as the bearing fill layer. All the embankments were covered by a sealing layer of moraine.

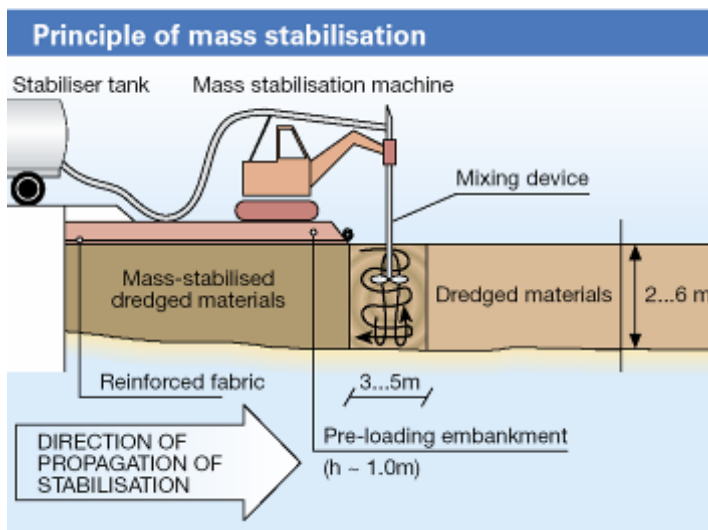


Figure 6. Principle of mass stabilization.

Permissions required

The TBT removal process required three separate permissions, the first for the construction work of the protective embankments, the second for the decontamination dredging, and the third for the utilization of the TBT-contaminated sediment in the harbour's field structures.

Results and discussion

With the TBT removal process 98,8 % (96,5 kg) of the TBT of the decontamination dredging area has been removed from the sea and will be permanently isolated from the aqueous environment.

The influences of the construction work and the TBT removal process on the sea environment have been monitored according to very extensive monitoring program approved by the environmental authorities. During the TBT removal process there has been no indications of such detrimental effects, what differ from the effects of conventional dredging operation. In fact, thanks to the protective structures, spreading of suspended solids was restricted to a smaller area than without the TBT removal process. A very significant result of the monitoring is, that the concentration of TBT and other organotin compounds in sediments, molluscs and fish in the sea area surrounding the harbour building site have decreased from the year 2003 to the present. The extensive monitoring of the sea environment continues until the Vuosaari Harbour's construction work is finished.

The removal of TBT from the seabed has been a seamless part of the Vuosaari Harbour's construction work. At the moment it is apparent, that the Vuosaari Harbour will be operative as scheduled, i.e. by the end of the year 2008.

Groundwater pollution of PCE in Hagfors, Sweden

Abstract for oral presentation in section: State-of-the-art projects from the Nordic countries

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Backgrounds

Hagfors municipality is situated in Värmlands county in Sweden. In this small town there has been a large dry cleaner facility, once the largest dry cleaner facility in Northern Europe causing extensive impacts to soil, soil vapor and groundwater. There have been investigations going on in Hagfors since 1993. A result from these investigations is that two soil remediation projects have been conducted underneath the dry cleaner facility. The main reason to remediate was the risk of intrusion of PCE into the manufacturing industry hall where they conduct welding. PCE and welding may cause a poisonous gas called Phosgene at very low levels of PCE.

Purpose

The scope of this presentation is to present the contamination extent in soil, pore gas and groundwater in Hagfors. Furthermore I will briefly inform about the remediation conducted with SVE and ISSS, the results and lessons learned and propose a remediation for groundwater.

Project

In Hagfors we first remediated in 1996 with Soil Vapor Extractions (SVE), continued with sub-slab venting 1996 - 2003 and finally used In Situ Steam Stripping (ISSS) combined with SVE 2003 - 2004. The ISSS remediation is the only one of its kind in Sweden and one of few in Scandinavia. Totally approx. 7 000 kg of PCE have been remediated, 1 500 kg 1996, 500 kg 1996 - 2003 and 5 000 kg 2003- 2004. The groundwater analyses conducted in Hagfors started up 1994 with testing the groundwater quality in 5 new and 4 old wells. The number of wells and the number of filters vertically has increased during the years up to 32 different locations/levels 2003. There have been eight rounds of analyses in the period 1994 to 2005. The parameter analyzed was at first PCE and TCE. This program has gradually increased to PCE and its degradation products and redox parameters. In the ground water zone there has been no remediation. In SWECO's project called "Manual for remediation of chlorinated solvents in soil and groundwater", a Hållbar Sanerings project, we use Hagfors as an example to evaluate how to remediate this particular site, both with aspect to the soil and the groundwater. This project is performed by SWECO in cooperation with the Danish company COWI A/S and the US company GeoSyntec.

Conclusions

The investigation shows that there is a large PCE pollution in both in the unsaturated and the saturated zone. In our Hållbar Sanerings project we learned from GeoSyntec that we had not realized the width of groundwater pollution. In fact the amount of PCE transported off site in groundwater during a 30 year period of time is in the same order as the amount of PCE remediated, several tons. We now clearly see that it is imperative to set up proper RAO's (Remedial Action Objectives). That we did not see ahead of the remediation 1996. We have in Hållbar Sanering proposed RAO for PCE/Phosgene in indoor air, PCE in soil and in the river Örbäcken. We also have discussed (but not proposed) on RAO's for groundwater (if Hagfors should be interested in groundwater for water supply) and for indoor climate in houses on top of the plume. I will briefly present the recommended RAO's and remediation (-s) for soil and groundwater. Especially the RAO concerning PCE/Phosgene is of interest because the dangerous combination PCE and welding might occur at many sites and the knowledge of the risk is not well known.

REMEDIATION DUE TO CREOSOTE CONTAMINATION IN WETLANDS IN RÅDE, NORWAY

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Abstract

Råde Impregneringsverk is located 80 km southeast of Oslo. The site is located very close to the southern end of the Auberghølen lake and the original water channel out of the lake, leading to Skinnerflo nature reserve. Between the impregnation site and this channel there is wetlands. Investigations as early as 1992 had proven the site and in particular the wetlands between the site and channel to be strongly creosote-contaminated.

The remediation project started in 2003, firstly by preparing a risk assessment and remediation plan according to instruction from SFT. The remediation plan had to consider the fact that there were strong local and environmental interests in avoiding any permanent damage or alteration of the wetlands, and that any water level change in the lake could damage surrounding farm land. The remediation was carried out in 2005. Strongly contaminated soil (323 ton) was removed from the plant site and a clean soil cover established on this site (6.300 m²). A new water channel (length 270 m) with a wier (to stabilise the water level in the lake) and a fish ladder in a culvert was constructed through the wetlands.

Introduction

Råde Impregneringsverk is located 80 km southeast of Oslo. The plant was operated by NSB from ca. 1900 to 1943, mainly impregnating railway beams. Investigations as early as 1992 had proven the site and in particular the wetlands between the site and water channel to be strongly creosote-contaminated. The contaminated wetlands and surrounding areas were regularly flooded, as the water level in Auberghølen varied considerably pending on the weather.

The site is registered in SFTs list of the most important contaminated sites in Norway. Research was carried out in the 1990s to establish the environmental effect of the wetlands contamination, and different remediation alternatives were evaluated and rejected. In 2003 the site owner Jernbaneverket (the Norwegian National Rail Administration) was instructed by SFT to prepare a risk assessment and remediation plan for the site. The remediation plan was presented by Jernbaneverket to SFT early 2004. SFT instructed in April 2004 Jernbaneverket to carry out the remediation according to the remediation plan with some minor adjustments, so that the contamination would not constitute any future health or environmental risk.



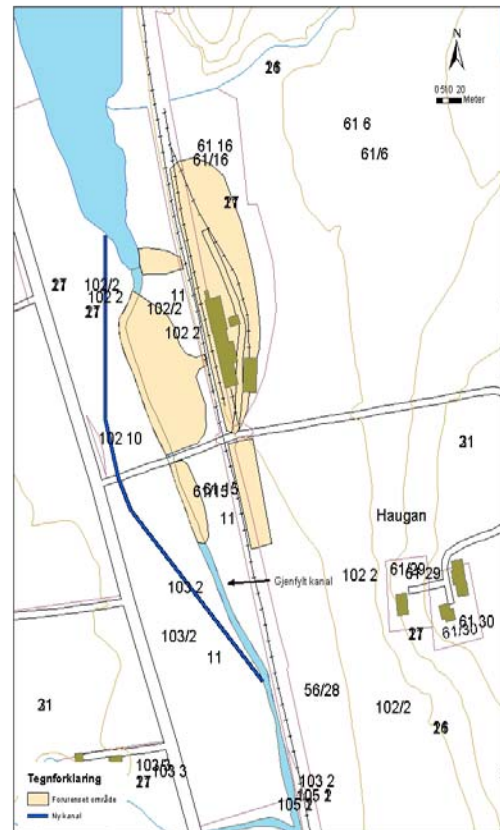
Materials and methods

Site description

The site is located very close to the southern end of the Auberghølen lake and the original water channel out of the lake, leading to Skinnerflo nature reserve. Between the impregnation site and this channel there is a railway line and part of a larger wetlands area. The wetlands are considered important for the biological diversity in the area, with a rich life and varied flora.

Except for a thin fill layer on the plant site the ground consists of a fat, silty clay with very varying depths to bedrock. The creosote contamination situation is:

- Plant site: ca. 4.200 m² is contaminated with contaminated soil volume ca. 4.500 m³.
- Wetland: ca. 4.400 m² is contaminated with contaminated soil volume very roughly estimated to ca. 16.000 m³. Free phase creosote registered in the 1990s in cracks in clay and along plant roots to several m depth. Strongly contaminated sediments in the old water channel through the wetlands.



The main cause of the wetlands contamination was a buried pipe leading from the plant site underneath the railway to a partly buried barrel at the edge of the wetlands. The barrel was supposedly used to collect surplus creosote from the plant.

Risk assessment and remediation plan

A health related risk assessment was performed based on registered soil concentrations and assumed future recreational area usage. An environmental related risk assessment was performed based on a series of biological effect studies in the 1990s (microtox, plants, fish, earthworms, mouse, snails) and pore water sampling. The risk assessment concluded that:

- Health: both the plant site and the wetland is contaminated above calculated accept criteria for top soil. A new top soil layer is uncomplicated on the plant site, but in the wetland this may cause permanent damage to existing plant life.
- Environment: the monitoring of the water in the channel has not documented any unacceptable migration of PAH / oil towards downstream areas, but sudden flush of contaminants may be possible during heavy rains. The effect studies are not conclusive. There is however no doubt that soil in some parts of the wetlands is poisonous, may locally affect certain plants and cause increased PAH-concentrations in earth worms (being food for birds and small mammals). The ecosystem in the wetlands is however probably not in general negatively affected.

The remediation plan had to consider the fact that there were strong local and environmental interests in avoiding any permanent damage or alteration of the wetlands, and that any water level change in the lake could damage surrounding farm land. The most important element of the risk assessment was therefore to balance the need of preventing future environmental risk and not altering the wetlands.

The remediation evaluation concluded that a clean top soil layer should be established on the plant site after removing strongly contaminated soil (PAH > 2.100 mg/kg). The existing water channel passing the contaminated wetlands should be relocated (preventive remediation). At the same time the water level in the Auberghølen lake should be stabilised at a normal level.

Performed remediation

After SFT accepted the remediation plan with minor adjustments, detailed design and tender preparation was carried out. SFT instructed that the possibility of establishing a thin top soil layer in the strongly contaminated parts of the wetlands should also be evaluated. The remediation was then carried out from May – August 2005. There was a strong focus during the remediation of not damaging the wetlands permanently. An emergency plan and a monitoring plan was prepared.

The remediation on the plant site included:

- Demolishing all existing buildings (ca 750 m²)
- Removing 113 m³ contaminated water from three creosote tanks in bedrock underneath the impregnating plant. The tanks afterwards filled with clay
- Removing strongly contaminated soil with average PAH-concentration 3.500 mg/kg (323 tonn)
- Establishing a clean soil cover of 0,35 m clean soil and 0,15 m gardening mould on the whole site (6.300 m²) and sow seeds of local grass
- Establishing a cut off trench for surface water upstream the site



Plant site before remediation



Plant site with new soil cover (2005)

The remediation of the wetlands included:

- A new water channel (length 270 m)
- Widening the down stream part of the existing channel to improve the water flow capacity of the channel
- A new culvert on the water channel under a local road with an adjustable wier (to stabilise the water level in the lake) and a fish ladder to make it possible for fish > ca. 7 cm to bypass the wier
- Backfilling the part of the existing channel (partly contaminated) that had been replaced (230 m)
- Covering the strongest contaminated part of the wetlands (1.700 m²) with 20 cm soil with high organic content, after cutting down but not removing existing vegetation of common reed. The purpose was to establish a clean top soil layer that would not alter the plant life on the covered part
- Planting black alder and willow along the new channel



Northern part of new channel (2005)



Part of culvert with wier (2005)

Results

Event though construction work was carried out within partly strongly contaminated wetlands, no unforeseen contamination occurred. The water level in the lake has been stabilised, fish is observed to easily move past the water ladder (and people fishing there) and the continued water monitoring show no contaminant migration. The ground underneath parts of the wetlands will however also in the future be strongly contaminated.

In late August 2006 SFT and Jernbaneverket will make a site visit to inspect the effect of the remediation, including how well the wetlands have vegetated along the new channel and where the thin soil cover was established, and whether the stabilised water level in the Auberghølen lake should be adjusted.

References

Multiconsult AS. 2004. Risk assessment and remediation plan. Report no. 111242-1.

Multiconsult AS. 2005. Final report documenting performed remediation. Report no. 111242-4.

SFT. 2004. Instruction to perform remediation at Råde Impregneringsverk. Letter dated 27. April 2004.

ENVIRONMENTAL SOLUTIONS FOR MILITARY SHOOTING RANGES IN NORWAY AND FINLAND

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Abstract

The environmental impacts of heavily used military shooting ranges are a current issue in Norway and Finland, hence the two countries cooperate to find environmentally sound solutions. Both countries use mainly traditional sand-berms as bullet traps on outdoor shooting ranges, and tightened environmental legislation requires new measures to prevent the spreading of lead and other metals. As there are 200 and 800 military shooting ranges in Finland and Norway, respectively, and more than 3000 civilian shooting ranges, the issue is of great economical importance. The presented project is carried out in unofficial co-operation between Norwegian and Finnish armed forces. The purpose is to: 1) find the distribution of metals in shooting range soil, 2) understand processes important for leaching of metals and 3) find the technically and economically best way to prevent the migration of lead (Pb), copper (Cu), antimony (Sb), and other metals from shooting ranges into soil, surface- and groundwater. The results show that there is no single solution for preventing leaching of metals from shooting ranges.

Material and Methods

The project is still ongoing and consists of six parts:

1) Surveys of the content and distribution of lead and other metals in shooting range soil and impacts on surface- and groundwater.

A survey on heavy metal contamination in different parts of shooting ranges was carried out in Finland in 2004. The concentration and horizontal and vertical spreading of Pb, Cu, Sb, Zn and Ni was surveyed on nine military outdoor shooting ranges with varying natural settings and rate of use. The goal was to establish a possible contamination pattern. The survey also included sampling of groundwater, surface water and plants, as well as leaching tests of soil from impact berms and a study on air-quality during firing, i.e. amount and lead concentration of dust in the air.

In Norway concentrations of lead and copper in creeks and rivers draining shooting ranges have been monitored using Aquatic Moss (*Fontinalis darlica*) and water samples. Twenty five shooting fields have been monitored for up to 15 years (Rognerud 2006).

2) Studies on metals; species of metals and behaviour during episodic events.

The amount and variation in species of metals in a creek draining a shooting range, were studied during episodic discharge of water. The first study (2001) was carried out during periods of snow melting followed by periods of rain. Samples of water were taken

almost every second day from a creek during snow melting. During the rain events, water samples were size fractionated and analyzed for Pb, Cu and Sb (Strømseng and Ljønes 2003). In a second study (2003), water samples were taken from the creek during periods of rain. Samples were size and charge fractionated *in situ*, using filtration, ultra filtration and ion exchange, and analyzed for Pb, Cu, Sb and Zn (Heier *et al* 2004).

3) Collection of background data on current remediation techniques and commercially available environmental shooting range solutions.

4) Testing efficiency of various soil amendment products and filters.

Soil additives are used to stabilize soluble metal species in shooting range embankment soils to prevent metals from leaching from berms or contaminated soil. Contaminated soil from berms was mixed with respectively bone char, iron (Fe⁰) powder and alginate and filled into columns. The columns were placed outside to receive natural precipitation. Samples from the outlet were collected after periods of precipitation, approximately every month, for one and a half year.

Several studies to test different filter media have been conducted in Norway in order to remove Pb, Cu and Sb from polluted water. In 2004 FFI established a large-scale filter system consisting of two down-flow filters and one up-flow filter at Steinsjøen shooting range (Strømseng *et al* 2005). In 2005 FFI established a mobile field laboratory for testing filter media *in situ* at various shooting ranges. Contaminated water collected automatically from the creek is led through water quality control, and through 9 columns.

5) Pilot-scale testing of 3-4 shooting range structures chosen or developed on basis of the results of the other studies.

6) Recommendations for shooting range structures.

Results

In study number 1 (Construction Establishment of Finnish Defence Administration 2005), results from Finland show that the distribution of heavy metals on shooting ranges has a clear pattern. On basis of this information, a “theoretical model shooting range” has been developed to use for risk assessment and economical comparison between possible future shooting range structures and remediation/stabilisation solutions. The theoretical model shooting range is a 150 m rifle range with 25 firing places and a use rate of 150 000 shots/year. Based on the use and dimensions, the distribution on heavy metal contamination on the range can be summarized as follows:

	Pb (mg/kg)	Pb(t)	Pb(%)	Soil (t)	Soil (%)
Front of firing line	300	0,3	6	950	18
Field	100	0,3	6	3100	60
Side berms	50	0	0	0	0
Target area	500	0,01	0	18	0
Berm	1500	0,9	17	580	11
Bullet pocket	100 000	3,5	67	34	0,7
Slope behind berm	500	0,2	5	470	9
Total		5,2	100	5200	100

The study also showed, as predicted, that in the neutral conditions that prevail on most Finnish shooting ranges, there is very little migration of contaminants into the ground or surface water. In acidic conditions migration of metals into surface waters could be observed. The effect of the sample pH could also be seen in the leaching tests that were performed on impact berm soil. In addition to the pH, the grain size also affects the solubility of the metals. At the firing line more soluble fine particles of metal are found as the firing event produces dust from the bullet. No traces of lead were found in the groundwater of the shooting ranges, even though most are situated on permeable, sandy soil. The depth of the groundwater surface varied from 1-10 m. An increased amount of dust was detected in the air by the firing line and the impact berm during firing. The lead content of the dust was considerably higher by the firing line, but not as high as to cause health risk.

The monitoring program conducted in Norway also shows that there is usually only problems with leaching of metals in acidic areas. In general, where Ca concentrations are high (above 10 mg/l), concentrations of Pb and Cu in creeks can be expected to be below 2 and 8 µg/l, respectively. In creeks where Ca concentrations are low and TOC concentrations are high, concentrations can be up to 200 µg/l Pb and 300 µg/l Cu. There is no such correlation for the anionic Sb. Typical concentrations vary from 1 to 20 µg Sb/l (Rognerud 2006). Acidic bogs that have been used for combat ranges have often been limed. This method can reduce leaching of metals, but the monitoring program has shown that liming in some cases increase leaching of metals.

In study number 2 (Strømseng and Ljønes 2003), results show that during the short snow melting period the amount of metal run-off is estimated to 30 % of total yearly amount. No correlation was found between the concentrations of metals and the flow. However, in the following period, where several rainfall events occurred, the concentration of metals correlated positively with the flow. The correlation (R^2) between Pb, Cu, Sb and flow were 0,95, 0,83 and 0,74, respectively. The results from 2003 confirmed that the concentration of all elements increased when the discharge increased. Pb was mainly associated to particles and colloids. Cu was mainly found in the colloid fraction (on average 60 %) and the < 10 kDa fraction (35 %). Zn and Sb were mainly found as low molecular species, on average 60 % and 84 % respectively (Heier *et al* 2004). A large fraction of Zn was present as positive reactive species, while Sb was mainly present as negative reactive species. FFI is currently conducting an experiment to measure bioavailability of Pb, Sb and Cu in fish.

In study number 3, data on remediation techniques and commercially available environmental shooting range solutions was collected and compared using the theoretical model shooting range (Finnish Defence Forces 2005a, 2005b). The study showed that although there are number of commercially available shooting range solutions, there is generally very little data available on the building, maintenance and close-down costs. There are also very few solutions using chemical stabilization of contaminants. The conclusion was that the most promising structures would have to be pilot tested and possibly further developed to obtain the environmentally sound and cost-efficient solution that is sought. It also seems that there will be different solutions for different environments and rates of use. In the case of remediation techniques, the solution will have to be built on risk assessment and minimizing the lead migration, rather than removal of all the contaminated soil. The theoretical shooting range model shows that most of the lead in shooting ranges is concentrated in a small amount of soil.

By removing this and the soil polluted by fine-grained metals at the firing line, the risk is already decreased considerably and an in-situ treatment of the less contaminated areas will perhaps form the rest of the solution.

In study number 4, FFI found that granulated peat from northern Sweden have potential for removing metals from water in up-flow filter system. Further experiments are needed to make final conclusions on long-term effects. Tests were performed in the field laboratory with granulated peat, granulated peat coated with micro algae, active coal, olivine, iron coated olivine and light expanded clay aggregates (leca) coated with olivine (2005/2006). The results are not yet evaluated and reported.

The soil additive study conducted by FFI show best results using 2,5 % (w/w) iron added, where the leakage of lead is reduced with approximately 90 %, even after 1,5 years. Iron coated olivine and olivine are currently being tested.

Study number 5 is still being planned, and will probably be carried out in 2007-2009. In study number 6 Finland and Norway will work together to make a guideline for recommendations of shooting range structures that are environmentally sound.

Discussion

The results from the contamination studies show that there is usually no immediate need for renovation of shooting ranges. From an environmental and health point of view there is still time for surveys for the Best Available Technology of renovation. An acute need for renovation or other local measures usually arises from change of land use (closed-down ranges) or exceptionally high environmental risk caused by for example acidic conditions or sensitive ground- or surface water use. Closed-down ranges are currently being investigated for contamination and leaching, whereas all active ranges (Norway) are being investigated to determine how much contamination is leaching from the ranges. The knowledge obtained from our studies will be used to test and conduct remediation methods to prevent leaching of metals. Different remediation methods are probably needed for different parts of a closed-down range for optimum results. Soil amendment shows great potential, especially for use on the less contaminated parts of a range and ranges that are less used. For renovation of ranges in use, there seems to be no single solution. Structure recommendations will be developed for various environments (soil type, surface- and groundwater conditions, pH etc.) and rates of use.

References

Heier L S, A Strømseng and M Ljønes. 2004. Speciation of heavy metals in runoff streams in shooting ranges. FFI report 2004/02971. In Norwegian.

Rognerud S. 2006. Overvåking av metallforurensning fra militære skytefelt og demoleringsplasser. Resultater fra 15 års overvåking. NIVA/Forsvarsbygg report 5162.

Strømseng A E and M Ljønes. 2003. Heavy metal run-off during episodic events – Field study at Steinsjøen shooting range. FFI report 2003/00715. In Norwegian.

Strømseng A E, M Ljønes and M Christiansen. 2005. Full scale filter tests for cleaning heavy metal polluted water with thermal treated peat in Steinsjøen shooting area. In Norwegian.

Construction Establishment of Finnish Defence Administration (Ramboll Finland Oy). 2005. Environmental Impacts of Heavy Metals on Armed Forces' Pistol- and Rifle Shooting Ranges. Summary Report. In Finnish.

Finnish Defence Forces (Ramboll Finland Oy). 2005a. Remediation of Lead Contaminated Soil from Shooting Ranges; Need of Remediation and Technologies. In Finnish.

Finnish Defence Forces (Ramboll Finland Oy). 2005b. Development of Shooting Range Structures. In Finnish.

Who can you trust? To communicate risks concerning contaminated land

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Abstract

Soil investigations and clean-up projects may go on for many years and may create anxiety to neighbors and other interested parties. Efficient and credible communication of risks is therefore an important element of a soil remediation project. On behalf of the Swedish EPA (Naturvårdsverket), WSP Environmental carried out a study on risk communication in association with contaminated land.

The study identified the following components as being of special interest concerning risk communication of contaminated land.

- To plan the risk communication by identification of target groups, identification of means for communication and preparedness to manage potential conflicts.
- To create an open and transparent process where there is focus on the dialogue between the interested parties.
- To realize that facts and rationality does not necessarily go hand in hand. Risk perception is a mixture of technical, psychological, sociological and cultural considerations.
- To prioritize communication issues and prepare a communication plan. It is too late to deal with the communication issues in a systematic way when the public and mass media already are knocking on the door.

Introduction

Soil investigations and clean-up projects may go on for many years and may create anxiety to neighbors and other interested parties. Efficient and credible communication of risks is therefore an important element of a soil remediation project. On behalf of the Swedish EPA (Naturvårdsverket), WSP Environmental carried out a study on risk communication in association with contaminated land. The study provides advice to project management and authorities on how to deal with risk communication. The overall aim of the study was to collect ex-

periences and advice to minimize the risk for conflicts between interested parties. The study was carried out in 2005-2006.

Materials and methods

The study consisted of a literature review and a case study of five Swedish contaminated land projects where risk communication had played an important role.

The purpose of the literature review was to give an overview of the concept of risk communication in general and in particular literature concerning contaminated land. The literature studied had an emphasis on reports, books and guidebooks dealing with risk communication in a comprehensive way. The case study is a study of five Swedish projects where risk communication concerning contaminated land has played an important role (in a positive or negative way). The process of identifying suitable projects to study involved contacting all County administrative boards in Sweden. The five projects chosen represent the three major project categories that exist in contaminated land projects, i.e.:

- Development project e.g. when building projects involve dealing with contaminated land.
- Supervision projects, e.g. when the supervisory authority/environmental authority forces a company to investigate or clean-up a contaminated site where they currently or previously carried out activities.
- Subsidy project, e.g. when the Swedish EPA finance investigation or treatment of contaminated land.

The case study consisted of interviews with three categories of people who had been involved with/in contact with the risk communication that had been carried out in the different projects. The three categories were, the project manager, a representative of the environmental authority responsible for supervising the project and a member of the public who in some way became affected by the project. The interviews questions where designed in cooperation with a psychologist at the Section of Behavioral Medicine, Lund University Hospital.

The project organization consisted of a working group and a reference group. The working group consisted of Johan Asplund, Torbjörn Brorson and Fredrik Lundgren, all three from WSP Environmental. The reference group where composed of people who deal with risk communication in their ordinary line of work. The reference group consisted of StenInge Arnesson, Kemira Kemi AB. Tomas Nilson, City of Malmö, Björn Karlsson, Lund University and Marie Arnér, WSP Environmental.

Results

The study showed that conflicts concerning communication of risks are common in contaminated land projects. The study also showed that sometimes unexpected issues triggered a communication conflict. It is therefore important that the communicator is flexible and prepared to adjust to questions and demands that may emerge unexpectedly.

The conflicts are usually about facts, trust or values. It is important to recognize that risk communication is not a “one way street” where the experts’ role is to inform the public and others of the facts of the projects. Risk communication is a process that should be built on the participation of all parties involved or affected by the project. It is also very important to plan the risk communication in advance and try to identify the information and communication demands of the concerned parties.

Trust is a very important aspect in all risk communication. To establish and maintain trust is therefore crucial. Sometimes an external expert may help in increasing the public's trust and confidence in a specific project.

Discussion

It is not easy to communicate risks in projects where the target group is heterogeneous and where there are differences in knowledge and experience. When it comes to contaminated land it is important to realize that experts do not have preferential right of interpretation of the risks. If members of the public experience that there is a risk involved with a certain project, that experience is real to them and has to be dealt with accordingly. To ignore it, or write it off as ignorance, is to invite problems.

The study identified a number of factors that should be considered when planning a risk communication strategy. Some of the factors are:

- Urban planning can cause conflicts in relation to contaminated land projects.
- Contaminated land affects property value and therefore has the potential to affect people's private economy.
- The risk communication process needs to be project specific and planned in advance.
- Make an effort to build confidence and trust early in a project, you'll probably need it later!
- Prior to risk communication - Try to identify all parties that may be affected by the project.
- Inform all concerned parties in advance and invite them to a dialogue.
- If possible - Let an external party review all information material before it is used.

- Media is an important information carrier. Realize that almost every contaminated land project has the potential to make the headlines.
- Be flexible and adapt to the situation and peoples demands.
- Risk comparisons can cause conflict. Use them with great care.
- Don't forget that risk communication takes time.
- Be very clear about who is responsible for what. Misunderstanding about responsibilities cause conflict. What is the responsibility of the site owner? What is the responsibility of the environmental authorities?
- Don't forget to follow-up your communication effort.

References

Af Wåhlberg och A & Sjöberg, 2000, L. **Risk perception and the media**, Journal of Risk Research **3** (1), 31-50.

Breck, T., 2002, **Risikommunikation dialog om det osäkra**, Natur och kultur. Stockholm.

Burningham, K. & Thrush, D., 2004, **Pollution concerns in context: a comparison of local perceptions of the risks associated with living close to a road and a chemical factory**, Journal of Risk Research **7** (2), March 2004, 213-232.

Drew, C. & Nyerges, T., 2004, **Transparency of environmental decision making: a case study of soil cleanup inside the Hanford 100 area**, Journal of Risk Research **7** (1) January 2004, 33-71.

Hedman, L., 1999, **Att kommunicera risker**, Räddningsverket . Karlstad.

Hunt, J. & Wynne, B. (red), 2002, **Social assumptions in remediation strategies**, Lancaster university. Lancaster.

Hunt, J. & Wynne, B. (red), 2002, **Social assumptions in remediation strategies**, Lancaster university. Lancaster.

Lidskog, R., Nohrstedt, S-A & Warg, L-E (red), 2000, **Tillit och trovärdighet i risikommunikation. Risker, kommunikation och medier En Forskarantologi**, Studentlitteratur. Lund.

National Research council, 1989, **Improving risk communication**, National Academy press. Washington DC.

Naturvårdsverket, 2006, **Efterbehandling av förorenade områden Kvalitetsmanual för användning och hantering av bidrag till efterbehandling och sanering**, Utgåva 2, Naturvårdsverket. Stockholm.

Nicole, 2004, **Communication on contaminated land**, www.NICOLE.org.

SNIFFER, 1999, **Communicating understanding of contaminated land risks**. Rapport SR97(11)F.

Weber, O., Schoz, R., Buhlmann, R. & Grasmuck, D., 2001, **Risk Perception of Heavy Metal Soil Contamination and Attitudes toward Decontamination Strategies**, Risk Analysis, Vol. 21, no. 5, 2001.

Wester-Herber, M, Warg, L-E, 2004, **Did they get it? Examining the goals of risk communication within the Seveso II Directive in a Swedish context**, Journal of Risk Research **7** (5) July 2004, 495-506.



TITLE

SPEAKER

Source zone remediation with innovative technologies

Tom Heron, NIRAS

Advances in biological in-situ remediation:
Full scale applications in the Netherlands

Maurice Henssen, Bioclear

Experiences of applied remediation technologies
and solutions in Sweden - 1995-2005

Berith Juwonen, Tyréns

Full scale testing of sediment remediation
technologies for Nordic conditions

Jens Laugesen, DNV Consulting

Demonstration of the use of monitored natural attenuation
(MNA) as a remediation technology – DEMO-MNA

K.S. Jorgensen, Finnish Environment Institute (SYKE)

Recent developments for fungal bioremediation
technologies of soil

Marja Tuomela, University of Helsinki

Field Demonstration of Biostimulation and
Bioaugmentation for Remediation of Chlorinated
Solvents in a Sand Aquifer

Neal D. Durant, GeoSyntec Consultants

Remediation of the residual sludge from soil washing

Pernille E. Jensen, Technical University of Denmark

SOURCE ZONE REMEDIATION WITH INNOVATIVE TECHNOLOGIES

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Definition of the source zone

The source zone is defined as the volume of soil that contains mobile, droplets or ganglia of free phase contaminants as well as soil where free phase has been present, but has undergone a phase transition by sorption, diffusion and/or dissolution and can now be detected by analyses as high soil concentrations.

The challenges of source zone remediation

The ability to remediate source zones by in situ technologies is mainly hampered by flow by-pass, diffusion limitations and dissolution kinetics.

Flow by-pass

In almost any heterogeneous setting the residual capacity of the soil (the retention capacity) is larger in the fine grained soils than in the more permeable sediment. Thus, after a spill of free phase and drainage through the soil profile, more droplets or ganglia of free phase will be trapped in the lower permeable parts. Furthermore, pools of non aqueous phase liquid (NAPL) with NAPL saturations of more than 50 % can accumulate on top low permeable layers.

These droplets or pools of free phase dramatically reduce the permeability of the soil for the flow of water. Reduction in permeability for water of 60 – 80 % is common. The effect of this reduced relative permeability is that the natural flow of air and water or the flow induced during remediation efforts will be diverted around the most contaminated parts of the soil. This effect is mostly pronounced in NAPL bearing fine grained layers in a more permeable setting.

The described flow by-pass severely limits the ability to flush out residual NAPL/pools and the ability to deliver reactive remediation agents to the NAPL zones. This is one of the key explanations why Pump & Treat, Soil Vapor Extraction, Air Sparging, Surfactant Flushing etc. often fail to reach remedial goals.

Diffusion limitations

When free phase has leached through the soil a following spill, sorption, dissolution and diffusion will slowly redistribute the contaminants. Particularly in silty and clayey sediment diffusion will spread contaminants from the more permeable parts of the soil into the silt/clay matrix. During remediation the more high permeable zones can possibly be flushed by water, steam, air, ozone etc. whereas the low permeable parts of the soil are not affected by convective flow. Thus the

removal of contaminants from the low permeable soils is largely governed by the diffusion of contaminants out of the matrix or the diffusion of reactants into the matrix. Both processes are very slow (time scale of years to decades) and result in tailing and rebound phenomenon during remediation campaigns.

Dissolution kinetics

NAPL in the subsurface can be removed by dissolution into infiltrating rain water or to ground water. One effect of the flow by-pass described above is, that the contact between NAPL droplets and the water is reduced. Furthermore in the contact zone between NAPL and water a concentration gradient (profile) will develop in the laminar flow zone between the two separate phase liquids. The removal efficiency of the dissolution is thus limited by the diffusion of contaminants from the NAPL, through the laminar flow zone and into a more turbulent water flow in the soil pores. This limits the efficiency of technologies based on removal of contaminants in the dissolved phase.

Can source zones be remediated to MCL's (Maximum Concentration Levels)/very low concentrations?

Before roughly 1990 it was not considered feasible to remediate DNAPL source zones by in situ technologies to anywhere near MCL's. During the 90's technologies were developed to effectively address the source zones and with the evolution of the thermal methods and the most aggressive free radical based oxidation technologies source zone remediation became possible under some (high permeable) circumstances. In the late 90's and during the following years further technology development has enabled complete remediation of sites with different geology and different NAPL composition. As an example more DNAPL source zone in dense clay has successfully been treated with thermal methods.

What can we achieve with in situ technologies?

With most available in situ remediation technologies removal efficiencies of more than 60 – 70 % of the total mass in highly contaminated source zones are not likely to be achieved. The limiting factors are described above. Due to the limitations described above, clayey and silty soils pose the biggest challenges to source zone remediation. As of now, only with well designed and operated thermal technologies as In Situ Thermal Desorption (ISTD) and Electrical Resistivity Heating (ERH) almost complete source zone removal can be achieved. It is urgent to stress, that although the technologies are very promising for many sites, project success can be severely hampered by poor design, operation and/or monitoring.

In sandy soils more technologies offer the potential for almost complete source zone remediation: In Situ Chemical Oxidation based on free radical oxidation (Fentons, ozone, Regen-OX, Persulfate), the thermal methods ISTD, ERH and steam stripping have been developed and commercialised. Assisted by hydraulic or pneumatic fracturing, Enhanced Reductive Dehalogenation and the injection of micro- or nano-scale iron into source zones are technologies that are being developed for possible future use for source zone remediation.

ADVANCES IN BIOLOGICAL IN-SITU REMEDIATION: FULL SCALE APPLICATIONS IN THE NETHERLANDS

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Abstract

Bioremediation techniques have shown to be very powerful in the remediation of various sites in the Netherlands contaminated with chlorinated solvents and aromatics. Techniques have evolved and have become more proven technologies. Also the trust in these techniques has grown and bioremediation has become a realistic and recognized alternative for the more traditional excavation and pump and treat remediations.

Introduction

In the Netherlands groundwater is frequently used for production of drinking water. Therefore natural resources of groundwater have to be protected from contamination. Remedial activities are often initiated due to this need for protection. The densely population in the Netherlands also result in high amount of contaminated areas. At this moment more than 600,000 contaminated sites are known, of which at least 10% is highly contaminated and should be remediated as soon as possible. The high cost involved with this action – estimated at 18 billion euro - procured the need for cost effective remediation technologies.

In the past 10 years various new techniques with focus on biological in-situ remediation have been developed and implemented in the Netherlands. With national funding programs like NOBIS and SKB these developments and pilot testing were possible. Nowadays, these techniques are used full scale at contaminated sites, with good results.

Based on several cases the background of techniques is explained and results are shown. But also possible problems that had to be solved will be described. Besides, process management is important in these projects for successful communication and implementation.

Technologies and monitoring

One of the most interesting challenges is the anaerobic degradation of various contaminants. Great improvements have been realized in the past decade, e.g. in the field of reductive dechlorination of chlorinated solvents like per- and trichloroethylene (PCE and TCE). These contaminants can be degraded under strictly anaerobic, methanogenic conditions. In this process PCE/TCE is transferred to less chlorinated ethylene intermediates (c-DCE and VC) and eventually into harmless ethylene and ethane.

Degradation of these chlorinated ethylenes can be enhanced by addition of carbon sources to the soil and groundwater. This may be carbon source like sugars, alcohols or fatty acids.

Also aromatics may be degraded under anaerobic conditions, thereby acting as electron donors. In this case addition of an electron acceptor like nitrate or sulphate may be used for injection.

It should be emphasized that in all cases a reliable pre-investigation – e.g. consisting of groundwater characterization, degradation tests or pilot push-pull tests - is very worthwhile to overcome problems in the full scale application and to determine upfront the real possibilities and risks of an in-situ bioremediation. Based on the results of pre-investigations various designs and implementations of full scale bioremediation projects have taken place in the Netherlands.

Results

Hoogeveen and Almelo: Anaerobic degradation of chlorinated ethylenes contamination with the help of bioaugmentation, the injection of specific microorganisms

At the Hoogeveen site per-, tri- and cis-dichloroethylene (PCE, TCE and c-DCE) are present in an area of 250,000 m³ mostly sandy soil, down to a depth of 50 meters. The occurrence of c-DCE indicates only partial degradation. Non-chlorinated ethylene – the end product of reductive dechlorination – is not formed. Also feasibility tests showed no natural degradation capacity in the native soil to ethylene, also not with only carbon source addition. Despite lack of specific organisms, economic evaluation of the remediation cost showed that even with high initial cost for bioremediation the total cost still would be less than for conventional techniques for just isolation of the contaminants. At this site the new TCE concept was used, consisting of inoculation of the soil with dechlorinating bacteria from a bioreactor at full scale (figure 1).



Figure 1: Bioreactor system Hoogeveen



Figure 2: Mobile bioreactor

This first system encountered various problems, especially with the re-infiltration of groundwater and addition of carbon sources. Clogging of the wells occurred within several days, making it impossible to infiltrate biomass and carbon sources. Therefore a new filtration system, adaptation of the wells and the aboveground system was executed. After technical adaptation of the process, remediation proceeded well and within 12 months after optimisation the PCE contamination (starting with 50,000 ug/l) was degraded to ethylene, with PCE and less chlorinated ethylenes concentrations less than 1 ug/l.

At the Almelo site chlorinated solvents are present and remediation has taken place for about 2 years using air sparging, without the expected result. Based on the screening of redox parameters and determination of specific organisms (DNA analyses) the remediation technique was changed into an in situ bioremediation. Since molecular screening showed the absence of contaminant degrading organisms in the soil, the remediation plan consisted of injection of these organisms. In this way, with minimal investigation actions and costs the remediation plan was altered and results from the biological monitoring tools were used to design the new remediation.

Based on very fast remediation time and the huge capacity of the reactors in the Hoogeveen project, the reactor was downscaled to 8 m³ mobile system (figure 2). Pre-investigation showed that the needed reduction of the soil and groundwater for bioremediation was possible. Problems with extraction and infiltration were prevented using the experiences from the Hoogeveen site and a proper periodic cleaning procedure.

Monitoring indicated fast reduction of the soil due to the injection of the carbon sources acetate and lactate, removing the present sulphate and feeding the soil with sufficient organic substance for reductive dechlorination. Within 8 months the total site of 12,000 m³ and down to a depth of 9 mbgl was remediated, PCE concentrations decreased from 15,000 to <10 µg/l and ending up with harmless ethylene.

	February 10, 2004	April 8, 2004	June 2, 2004	October 5, 2004	July 7, 2005	November 3, 2005
PCE (µg/l)	15.000	7.000	< 0,1	0,2	< 0,1	< 0,1
TCE (µg/l)	< 40	1.300	< 0,1	< 0,1	< 0,1	< 0,1
C-DCE (µg/l)	< 40	1.300	10.000	45	0,76	0,3
VC (µg/l)	< 40	NA	NA	23	0,8	0,13
Ethene (µg/l)	0,2	4	57,9	2.100	NA	NA
Ethane (µg/l)	1,3	2	13,4	6,0	NA	NA

Results at the Almelo site, indicating quick and effective degradation, without residual concentrations of harmful intermediates.

Appingedam: Anaerobic biodegradation of high concentrations in the source zone

In the Appingedam case chlorinated ethylenes are partly present underneath old and fragile infrastructures. Soil contamination is present in the soil layer up to 9 mbgl, consisting of clay and peat layers that are difficult to treat. Excavation or pump and treat are impossible to carry out due to the soil type and possible unwanted settling of the soil. The volume of the source zone is approximately 1,000 m³ and total mass of perchloroethylene (PCE) is estimated at 500 kg. Due to vertical transport of the contaminants the chlorinated ethylenes have reached a sandy layer at 9 mbgl, in which a plume of approximately 150 meters has developed. The migration of the contaminants is the reason for remediation. The former remediation plan consisted of an everlasting and costly pump and treat of the plume area.

An alternative option is to use the already occurring degradation capacity - in particularly - in the core zone and to make sure that transport to the sandy layer is stopped. However, rates of degradation, especially for cis-dichloroethylene are too low to rely completely on NA. Batch experiments were performed with soil from the source zone to obtain information about the possibilities to enhance the rates of degradation by addition of carbon sources.

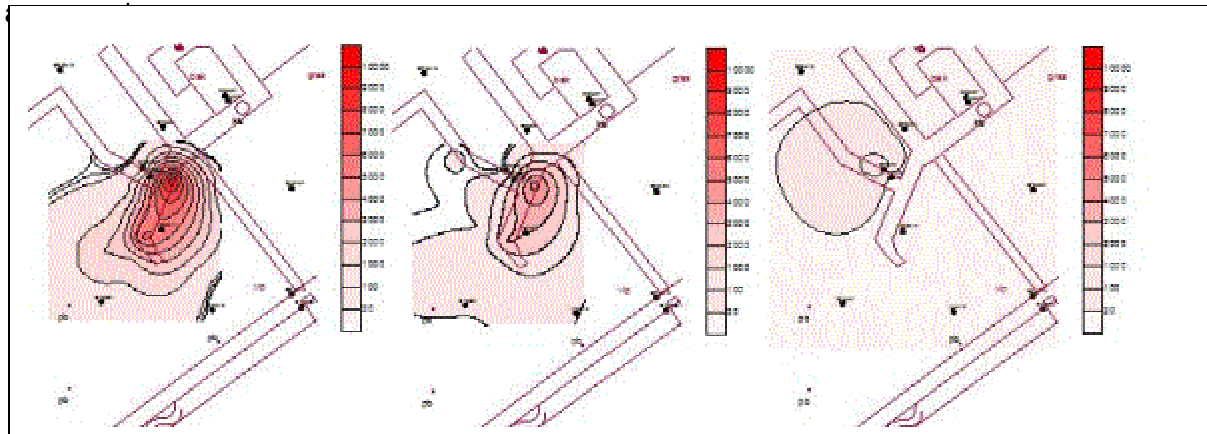
Despite the high contaminant concentrations (hundreds of mg/kg dw and groundwater concentrations up to 150,000 µg/l) in the core zone degradation rates could be increased by a factor 5-7 by adding carbon source. Based on these results the former remediation plan - installing a biobarrier at the end of the plume - was changed into a biological in situ treatment in the source zone. Furthermore, the expected remediation time is decreased to 10-12 years. Results of the monitoring end 2005 show that the plume area has vanished completely and that only the original source zone is contaminated. This zone has to be kept biological active. This is reached by injection of carbon source once every 2 - 5 years.



Source zone (red) and plume area (black) before remediation started (2002) and after 3 years.

Vries: Anaerobic degradation of aromatics using sulphate

Monitoring of the redox parameters at various sites that are contaminated with BTEX aromatics often show a consumption of different electron acceptors, indicating that natural biodegradation processes



Results of anaerobic benzene degradation using injections of sulphate to stimulate degradation

In the case of Vries-4 the consumption of sulphate/nitrate and the presence of intermediate products like benzoates and alkyl phenols indicated clearly the occurrence of natural attenuation. But it also proved that the biodegradation of BTEX aromatics was limited because of a lack of sulphate in the source zone. Combining these results, new and cost effective remediation strategies were introduced. Injection of electron acceptors like nitrate or sulphate resulted in a relatively easy and effective remediation. The proposed strategy could be tested using degradation tests. At the Vries site a one-time injection with high concentration of sulphate was used to treat the residual benzene contamination with concentrations up to 10,000 ug/l. Within 2.5 years the benzene concentrations decreased to about 10 ug/l.

Role of process management

First it should be emphasized that in-situ remediation projects are difficult to visualize: There are no clear remediation activities going on at a site, like extracted groundwater which is smelling or excavated soil in which mineral oil is visible. Progress in in-situ remediation projects relies on monitoring of specific parameters and exchanging a clear expression of the expected course of the process to e.g. governments is necessary. Therefore, in-situ projects require a lot of expertise to be able to obtain a good remediation. Besides, details are often very important in the whole process, making sometimes the difference between success or failure. Communication of all of this plays an important role in these projects and therefore project teams (with the consultant, contractor and government included) are a good collaboration platform.

Conclusions

For successful implementation of (biological) in situ techniques a proper pre-investigation and process management is essential. Once experienced with these techniques, implementation of biological in-situ techniques can result in more cost effective solutions. And in some cases these techniques can be very fast, resulting in remediation times of approximately 1-2 year, being therefore a good and safe alternative remediation technology.

Experiences of Applied Remediation Technologies and Solutions in Sweden 1994–2005

Berith Juvonen, Tyréns

This study has been financed by the Swedish Environmental Protection Agency and Tyréns and was conducted between august 2005 until april 2006. Participants were Ulf Wiklund and Berith Juvonen/Tyréns in Umeå/Skellefteå , Thomas Liljedahl/Environmental Chemistry University of Umeå, Johan Helldén/Johan Helldén AB and Sandra Broms/SPIMFAB.

The project

- Reviews and summaries 226 remediation projects that has been conducted in the country, approximately 15 % of the total amount of remediation projects.
- Describes and forms an opinion from the experiences made about the available remediation technologies in Sweden. International views are also made.
- Summaries how polluted soils are treated on Swedish waste treatment facilities on the basis of a questionnaire answered by 19 different treatment-companies.
- Summaries experiences of SPIMFABs work with remediation of closed up gas stations from the beginning in 1998 until recent years. The summary includes the number of completed remediations, the technologies used, the cost of the remediation, and further more.
- Summaries experiences made in 17 case studies and attempts to find key factors for the success of different technologies on basis of studies of the remediation reports of the different project and interviews with people involved.

The presentation at NordRocs places the discussion of the total benefit of remediation projects on the agenda. The project has been trying to analyse the compact on the environment that the different technologies and different projects in the case studies have contributed with. It is important to remember that the whole chain is to be counted with. But instead of complicated calculations that comes with Life Cycle Assessments where every in put in the project, for example the metal raw material in the treatment machine is counted with, this study takes a wider but yet simpler grip of the question. However, it is important to remember that no general conclusions about different technologies can be made when the basis of the study is far too small.

In this simplified calculation factors transport and energy consumption have been calculated with as negative contributes and remediated tonnes of soil as the positive contribution. The energy consumption in transports and the use of machines in the remediation projects has been recalculated to kg discharged CO₂ with help from a model made of the Swedish Environmental Protection Agency. The kg of discharged CO₂ is then related to the amount of tonnes remediated soil in the different projects. The presentation will give he result of that comparison.

FULL SCALE TESTING OF SEDIMENT REMEDIATION TECHNOLOGIES FOR NORDIC CONDITIONS

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Abstract

Cleaning up contaminated sediments in the Nordic countries has mainly been done with dredging and disposal or with capping. There are however several other remediation technologies which are of interest for Nordic conditions. In the Norwegian Pilot Project programme, full scale testing has been done on stabilisation of sediments and on sediment washing (separation) technologies. The purpose of the project was to investigate if these technologies were suitable and cost-effective for Nordic conditions. The full scale tests were done in Trondheim harbour on typical contaminated harbour sediments. The stabilisation of sediments was done with different methods and totally 25 000 m³ were stabilised. The sediment washing was done on site in a mobile washing and separation unit that treated 2 100 m³ sediments. Both technologies proved to be suitable for Nordic conditions. Analyses of the costs for the technologies showed that stabilisation can be a cost-effective solution today. Sediment washing is expected to be more cost-effective in the future when disposal costs are anticipated to be increasing.

Introduction

When contaminated sediments have to be remediated they are either capped in situ (a clean layer of sediments is placed on top of the contaminated layer) or dredged and disposed. The disposal of dredged material is either in a land/nearshore confined disposal or an open water disposal which normally has to be capped. The reason for choosing these "conventional" methods is the technically low complexity and the corresponding relatively low costs.

In the Norwegian Pilot Project programme alternative technologies have been tested for handling contaminated sediments with special emphasis on technologies suitable for Nordic conditions. The Norwegian Pilot Project programme has been initiated by the Ministry of the Environment and administrated by the Norwegian Pollution Control Authority. The project has been performed as a joint venture project between The Port of Trondheim, Rambøll, Skanska and DNV Consulting.

In Trondheim harbour two technologies have undergone a full scale pilot test:

- Stabilisation of contaminated sediments with cement + different additives
- Washing of contaminated sediments (separation in clean and contaminated fractions)

Stabilisation has been done in a couple of Nordic projects (Sörnäinen/Sörnäs in Helsinki, Hammarby Sjöstad in Stockholm and the ongoing project at Vuosaari/Nordsjö harbour close to Helsinki). Sediment washing has not been done before in the Nordic countries.

Materials and methods

Stabilisation of contaminated sediments

The stabilisation of contaminated sediments had the following tasks:

- Select useful binders
- Find a suitable mixing technology
- Perform a cost-benefit analysis of the method

Several binders were tested in the laboratory (cement, filter ash, silicate and lime) with respect to strength and leaching of contaminants. The most promising was a mixture of cement and filter ash (50/50) with 60 kg cement/m³ sediment + 60 kg filter ash/m³ sediment. In a later phase, mixing was also done with only cement (100 kg cement/m³ sediment) to reduce the mixing time. Approximately 25 000 m³ of contaminated sediments were stabilised.

For the stabilisation two different techniques were used:

1. Stabilisation in a separate basin. After 24 hours of hardening the sediments were moved to their final destination, the confined disposal facility.
2. The sediments were directly placed in the confined disposal facility where they were stabilised.

The latter technology was more efficient and did also give less disturbance of the stabilised material.

Washing of contaminated sediments

Washing of the contaminated sediments was done in a slightly modified mobile washing unit which normally is used for contaminated soil, see Figure 1. The unit had a maximum capacity of 5 metric tons/hour. The real capacity is normally lower because the equipment is set up for an average grain distribution and if this distribution is different, the capacity will be reduced. In this case the average capacity was 2,5 metric tons/hour.



Figure 1. The washing unit in Trondheim harbour.

The unit separated the sediments in clean and contaminated fractions. As a rule of thumb the amount of sand (and larger) in the sediments should be 60 % or more if washing should be considered. Furthermore analyses should confirm that the major part of the contaminants is bound to the fine and organic fraction. Figure 2 shows the flow chart for the washing unit.

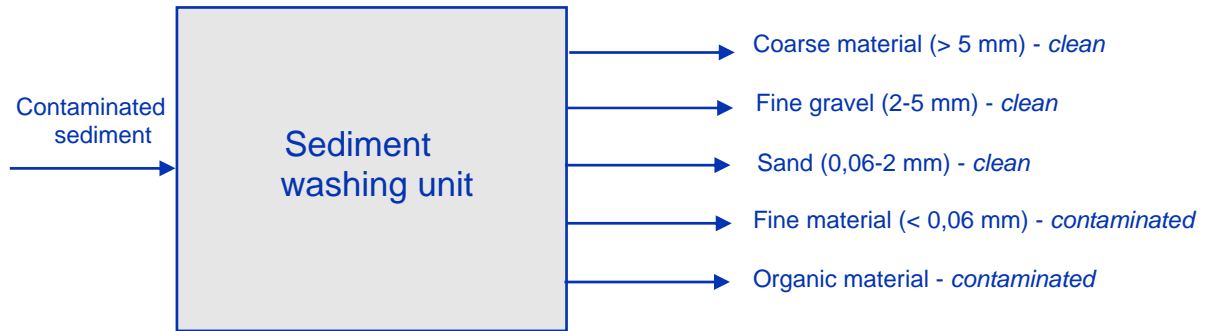


Figure 2. Flow chart for the sediment washing unit.

Results

Stabilisation of contaminated sediments

The stabilisation of contaminated sediments was successful. The stabilisation was performed during winter time without any negative influence on the result. The uniaxial strength of the material which was obtained in the field satisfied the criteria given for the project (100 kPa) based on the laboratory tests.

The cost for stabilisation in the pilot project was approximately 50 €/m³ sediment. Based on the costs from the pilot project, a cost estimate of 20 €/m³ was made for a larger project (100 000 m³). Figure 3 shows the unit costs for some of the Nordic projects where contaminated sediments have been stabilised. It is obvious that these kinds of projects can achieve a substantially lower unit cost when large volumes of sediments are treated.

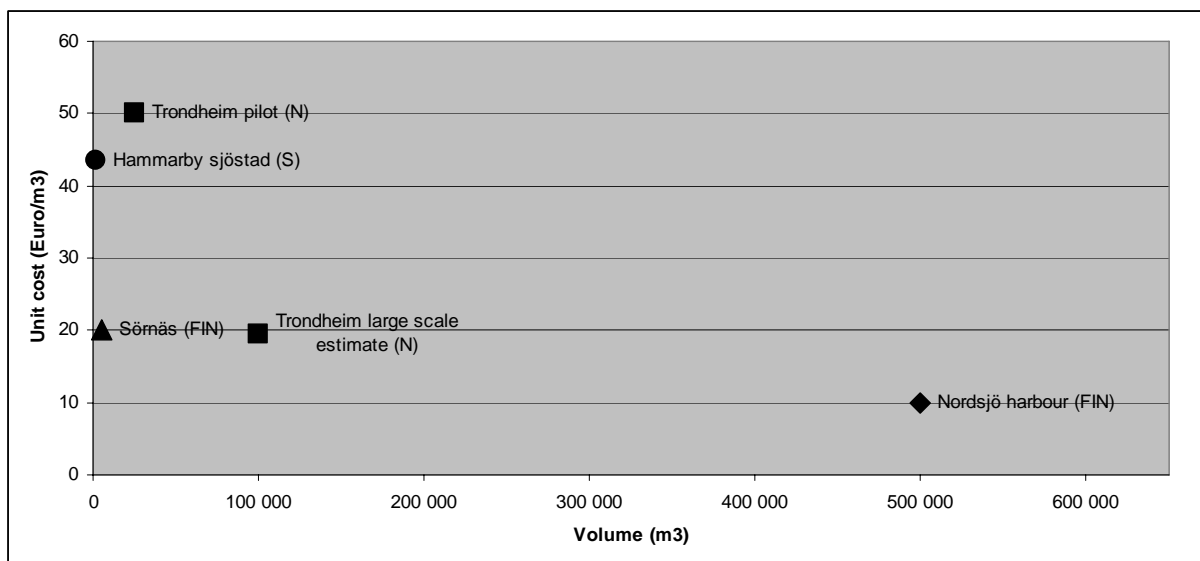


Figure 3. Unit costs for stabilisation of contaminated sediments in Nordic projects. Data for Trondheim is from Brånås and Laugesen (2005), data for Sörnäs and Nordsjö

harbour is from Leppänen (2006) and data for Hammarby Sjöstad is from Cementa (2002).

Washing of contaminated sediments

The washing/separation of the contaminated sediments worked well after some initial problems with organic particles (wood chippings) in the fine gravel fraction. This was solved by adding an air blower which removed the lightweight organic particles.

The washing/separation of the contaminated sediments reduced the amount of contaminated sediments by 70 %. Analyses showed that the resulting 30 % (the fine and the organic fraction) contained 60-70 % of all the heavy metals and 85-95 % of the organic contaminants (PAH and PCB). This contaminated fraction (30 %) was the remaining volume that had to be placed in a disposal facility.

The main benefit of the washing/separation is that the amount of sediments which have to be placed in a disposal facility can be substantially reduced. The 70 % which was the "clean" fraction was close to fulfilling the Norwegian criteria for free use of soil (SFT, 1999) or for dumping in the sea (SFT, 2004). It is anticipated that with minor developments of the washing unit, the criteria for free disposal of treated material should be possible to fulfil. The unit cost for the pilot test was very high because only a relatively small amount of contaminated sediments were treated (2 100 m³). An estimate for a larger commercial washing/separation project (100 000 m³) indicated a cost of 80 €/m³ (45 €/metric ton) including disposal of a 30 % contaminated residual fraction.

Discussion

Both technologies proved to be suitable for Nordic conditions. Analyses of the costs for the technologies showed that stabilisation can be a cost-effective solution today. Sediment washing is expected to be cost-effective in the future when disposal costs are anticipated to be increasing.

References

Brånås, M. and Laugesen J. (2005). Pilotprosjektet i Trondheim havn. Kost-nytte vurdering for mudring og stabilisering. Rapport 2003-016.

Cementa. (2002). <http://horse.merox.se/Aktuellt/Luma.html>

Grini, R.S., Emaus, K. and Kristiansen, B. (2005). Pilotprosjektet i Trondheim havn. Utførte arbeider 2003 og 2004. Rapport 2004-014.

Grini, R.S. and Kristiansen, B. (2006). Pilotprosjektet i Trondheim havn. Utførte arbeider renseprosjektet. Rapport 2006-021.

Laugesen, J. and Brånås, M. (2006). Pilotprosjektet i Trondheim havn. Kost-nytte vurdering for våtmekanisk rensing av forurensede sedimenter. Rapport 2006-023

Leppänen, M. (2006). Personal communication.

SFT (Statens forurensningstilsyn). (1999). Risikovurdering av forurenset grunn. Veiledning 99:01A.

SFT (Statens forurensningstilsyn). (2004). Veileder for håndtering av forurensede sedimenter. TA-1979/2004.

DEMONSTRATION OF THE USE OF MONITORED NATURAL ATTENUATION (MNA) AS A REMEDIATION TECHNOLOGY – DEMO-MNA

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Abstract

The aim is to demonstrate the feasibility of monitored natural attenuation (MNA) as a remediation technique for oil-contaminated sites in Finland. We use a combination of different disciplines such as geochemistry, hydrogeology, microbiology, ecotoxicology, toxicology, chemistry, and modelling based on the results from these studies. It was shown that a thorough geological and hydrogeological characterization is needed to assess for the understanding and modelling of the fate of the contaminants. Strong evidence for both aerobic and anaerobic biodegradation of oil hydrocarbons were found in laboratory tests at near *in situ* conditions, even though the observed rates may be overestimated. Field observations of electron acceptor concentrations and elevated CO₂ and HCO₃⁻ concentrations reflecting in high electric conductivity values within the plume support these findings. Electrical resistivity soundings in the area confirmed highly conductive zones around the hot spot. Biotests showed that the deeper layers were toxic to seed germination and worm reproduction. However, biomonitoring of the soil arthropods showed that a normal invertebrate community existed in the topsoil. The preliminary estimated cleanup time is several decades and the land-use scenario for the site is recreational, and this will be taken into account in the final risk assessment.

Introduction

Monitored Natural Attenuation (MNA) is a remediation technique in which a contaminated site is cleaned by natural processes (Chapelle, 1999). When considering MNA as a possible remediation option for a particular site, proof for the actual removal of the contaminants has to be shown and a proper risk assessment at the site has to be carried out (ASTM, 1998). To become acceptable remediation practice in Northern Europe, the feasibility of the method needs to be demonstrated under (hydro)geological and climatic conditions prevailing in northern part of Europe. The aim of this EU-Life-funded project (DEMO-MNA, 2003-2006, LIFE03/ENV/FIN/00250) is to demonstrate the feasibility of MNA as a remediation technique for oil-contaminated sites in Finland. The most important questions to be answered are the time frame of remediation and the acceptable risk level during and after the MNA-based remediation. Furthermore, the site characterization and monitoring required for the use of MNA will be evaluated.

Materials and methods

Site description. The study site, Trollberget, is an abandoned dumpsite outside the City of Hanko in Southern Finland. The dumpsite has been used for disposal of oily wastes. The biodegradation potential of oil contaminants has been verified previously at the demonstration site (Salminen et al., 2004; Tuomi et al. 2004). At the site, some heavy metal contamination has also been found; this is taken into account in the assessment of environmental risks at the site. Both the saturated and the unsaturated zones were investigated.

Geophysical measurements. Ground penetration radar system (GPRS) soundings were performed by a RAMAC/GPR ground radar equipped with a 100 and 200 MHz antennas in order to study the internal structures and thickness of the Quaternary sediment cover. Electrical resistivity soundings were performed by a an automatic Sting/swinft multielectrode instrument in May 2004 in order to map the extent of glacialacustrine silt and clay unit typically having poor results with GPRS sounding.

Soil, groundwater and soil gas sampling. The complex stratigraphy was verified with 11 test pit sections dug by a tractor excavator in May 2004. The excavator was also applied in order to determine the hydrological properties of the sediment units with standard infiltration tests. A gravimetric survey was performed in order to determine the depth and surface topography of the bedrock surface underneath the soil cover. Groundwater monitoring wells were installed at 7 locations in May 2004 in addition to the already existing 8 wells. Groundwater was sampled in Sept 2004, June 2005 and Oct 2005 with disposable bailers (Waterra).

On-line monitoring. A soil monitoring station (DEMSOIL) was installed in the oil plume in order to obtain on-line measurements of soil and groundwater temperature and electrical conductivity. Sensors for temperature, electrical conductivity, and water content in soil were installed at 110 cm, 190 cm, and 230 cm depth. Sensors for pH, redox, electric conductivity and the groundwater level were installed in the groundwater well DEMOGW20. Data loggers (Cambell Scientific) were used for automatic data collection.

Chemical analysis. Groundwater inorganic groundwater parameters were determined according to standard procedures. Element concentrations (cations) were determined with ICP-MS, PO_4^{3-} with titrimetric determination, anions with potentiometric determination. Furthermore field measurements were performed immediately after sampling, including measurements of pH, temperature, electrical conductivity, O_2 , Fe^{2+} , CO_2 and CH_4 using a portable WTW multimeter. Fe^{2+} was determined using the ferrozine (Stookey, 1970) method and a portable spectrophotometer (Hach). CO_2 and CH_4 was measured after acidification in closed bottles by GC-FID upon return to the lab. Total hydrocarbons in groundwater were analyzed according to ISO 9377:2 (2000) by GC-FID and volatile organic compounds according to ISO 10301(1997) by GC-MS. Total oil hydrocarbons in soil and sediment were analyzed according to ISO 16703 (2004) by GC-FID.

Microbiological assays. Soil samples were obtained from two or three depths from 3 sampling points. Each sample was sieved in the field through an 8-mm sieve and 30-g subsamples were distributed into 21 parallel 120 ml serum bottles. Bottles from the anoxic zone were immediately flushed with N_2 in the field. Samples from the oxic layers were incubated aerobically for 3 months, and samples from the anoxic layers were incubated anaerobically for 1 year in the laboratory at 7°C without any nutrient amendments. Triplicate bottles were sacrificed for hydrocarbon analysis at regular intervals. Furthermore CO_2 production was determined in aerobic bottles and CH_4 and CO_2 production was monitored in anaerobic bottles according to Salminen at al. 2004. In 2005 CO_2 production rates in bottle tests were compared rates obtained in intact, undisturbed soil cores.

Ecotoxicological measurements. Germination tests were carried out for four plants species, ryegrass, red clover, lettuce and cress according to ISO 11269-2 and ISO 17126. Soil animal tests, earthworm and potworm acute and reproduction tests (ISO 16387, 2004), were also performed. Furthermore the population structure of soil invertebrates in the topsoil was determined by identifying and counting individuals.

Modelling of the contaminant transport. The GMS program (EMRL) is used for the integration of geological structure and groundwater flow and contaminant transport. Since the spread of petroleum spill is essentially spreading along the capillary fringe, the feasibility of

simple biodegradation models (such as BIOSCREEN or BIOPLUMEIII by USEPA) that assume homogeneous hydraulic properties need to be assessed carefully.

Risk assessment. For the risk assessment, results from new soil and groundwater analyses were studied. Critical contaminants, which will be taken into account in the site-specific risk assessment were identified by using a scoring system taking into account the environmental concentration and (eco)toxicity of analyzed substances. Preliminary conceptual models for critical substance groups were prepared on the basis of their properties, environmental conditions on site and relevant land use scenarios.

Results

Based on the results of ground penetration radar, drilling and test pit studies, a schematic geological model of the study site has been compiled. The total thickness of the Quaternary sediment cover is approximately 4-8 m comprising of sediment units gently dipping to the northwest. The lower part of the strata comprises coarse-grained matrix supported diamicton. A discontinuous clay unit overlies the diamicton. A diverse littoral unit several meters in thickness overlies the clay. The littoral deposit consists of coarse, medium and fine sand typically representing horizontal lamination. The results of the resistivity measurements showed that an area of low resistivity could be located close to the hot spot. This indicates that there is high conductivity in this area.

Strong evidence of both aerobic and anaerobic biodegradation of oil-derived hydrocarbons were found in laboratory tests at near in situ conditions. Field observations of electron acceptor concentration and elevated CO_2 and HCO_3^- concentrations reflecting high electric conductivity values within the plume support these findings. The degradation rates roughly depended on the initial mineral oil concentrations and the rates did not correlate with the total soil respiration rate due to the presence of other organic matter than oil. In aerobic conditions, the degradation rates were $34 \text{ mg (kg x day)}^{-1}$ in soils from a highly polluted point ($15\,000 \text{ mg kg}^{-1}$) and $2 \text{ mg (kg x day)}^{-1}$ in soils from a less polluted point ($1\,300 \text{ mg kg}^{-1}$). During four months 60 % of the mineral oil disappeared in aerobic microcosms. Based on comparison of CO_2 production rates in microcosm bottle tests and intact, undisturbed soil cores it seems that the bottle tests overestimate the general microbial activity 5-50 times.

Seed germination tests showed that 2 samples below the clean topsoil (from the hot spot and from the slag deposition area) were fairly toxic. One sample that contained only heavy metals and no oil was not toxic. Lettuce was the most sensitive test plant. Potworm and earthworm tests showed close to 100 % reproduction inhibition for the samples containing both oil and heavy metals, whereas the sample containing only heavy metals only was 50 %. The earthworm was the most sensitive test animal. Samples for the investigation of the natural population pattern of soil invertebrates in the top soil layer at the site showed that both taxonomically and at trophic levels, the abundance of soil invertebrates were similar to that of broad-leaved forest in Southern Finland.

From metals, Pb is clearly the most critical element in soil since high risk scores were attained both in the case of ecological and human targets. As, Ba, Cd, Cr, Cu, and Sb are also relevant contaminants to follow. In the sediment Cu, As, Cd, Pb, and Zn were clearly the most critical elements related to ecological risks. From the petroleum derived hydrocarbons (PHCs) analysed in groundwater, naphthalene and xylenes are clearly the most critical. Dioxins have been analysed in previous site studies. The maximum concentration exceeded the Finnish soil limit value (500 pg/g). Therefore, dioxin contamination has to be considered in the risk assessment. The following recipients were identified to be relevant at the study site: 1) groundwater, 2) humans, 3) ecological (terrestrial and aquatic). The magnitude and probability of risks to these recipients vary depending on the contaminant.

Mass balance calculations have been carried out to assess conservatively the mass-fluxes of petroleum hydrocarbons, naphthalenes and total BTEX-components into the groundwater. The obtained estimates range from few fractions of kilograms to few tens of kilos per year depending on the component. The results are likely however, a conservative (overestimation) as the measured fluxes suggest that the plume is shrinking. Numerical modelling of moisture and ion fluxes into the oil polluted unsaturated zone have been carried out using HYDRUS 1D model. The results verify the estimated moisture retention and unsaturated hydraulic conductivity characteristics of soil will produce vertical moisture profiles similar to the monitoring observations.

Discussion

The results show that the boreal climate is no hinder for the progress of natural attenuation of petroleum hydrocarbons. The preliminary estimated cleanup time is several decades and the land-use scenario for the site is recreational, and this will be taken into account in the final risk assessment. The use of a an array of different methods for site investigation, monitoring and for the assessment of biological parameters proved to be necessary for a proper risk assessment, and gives a good basis for the evaluation of parameters to be included in future guidelines. The experience gained can be applied to other oil-contaminated sites in Northern Europe, Baltic countries as well as in other European countries.

References

- ASTM. 1998. Standard guide for remediation of groundwater by natural attenuation at petroleum release sites. ASTM E 1943-98.
- Bakermans, C., Hohnstock-Ashe, A.M., Padmanabhan, S., Padmanabhan, P. and Madsen E.L. 2002. Geochemical and physiological evidence for mixed aerobic and anaerobic filed biodegradation of coal tar waste by subsurface microbial communities. *Microb. Ecol* 44: 107-117.
- Bekins, B.A., Godsy, E.M., and Warren, E. 1999. Distribution of microbial physiological types in an aquifer contaminated by crude oil. *Microb. Ecol.* 37:263-275.
- Chapelle F.H. 1999. Bioremediation of petroleum hydrocarbon-contaminated ground water: The perspectives of history and hydrogeology. *Ground Water* 37: 122-132.
- ISO 10301. 1997. Water quality – Determination of highly volatile halogenated hydrocarbons. Gas-chromatographic methods.
- ISO 9377:2. 2000. Water quality – Determination of hydrocarbon oil index – Part 2 Method using solvent extraction and gas chromatography.
- ISO 16703. 2004. Soil quality – Determination of content of hydrocarbon in the range C₁₀ to C₄₀ by gas chromatography.
- ISO 11269-2. 1995. Soil quality – Determination of the effects of pollutants on soil flora – Part 1: Effects of chemicals on the emergence and growth of higher plants.
- ISO 16387. 2004. Soil quality -- Effects of pollutants on Enchytraeidae (*Enchytraeus* sp.) -- Determination of effects on reproduction and survival.
- ISO 17126. 2005. Soil quality – Determination of the effects of pollutants on soil flora – seedling emergence, screening test with lettuce (*Lactuca sativa* L.).
- Salminen, J.M., Tuomi, P.M. Suortti, A.-M., and Jørgensen, K.S. 2004. Potential for aerobic and anaerobic biodegradation of petroleum hydrocarbons in boreal subsurface. *Biodegradation* 15: 29-39.
- Stookey. L.L. 1970. Ferrozine – a new spectrophotometric reagent for iron. *Anal. Chem.* 43: 779-781.
- Tuomi, P.M., Salminen, J.M., and Jørgensen, K.S. 2004. The abundance of *nahAc* genes correlates with the ¹⁴C-naphthalene mineralization potential in petroleum hydrocarbon-contaminated oxic soil layers. *FEMS Microbiol. Ecol.* 51: 99-107.

RECENT DEVELOPMENTS FOR FUNGAL BIOREMEDIATION TECHNOLOGIES OF SOIL

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Abstract

Wood-degrading and litter-decomposing fungi can degrade and even mineralize a wide range of organopollutants with their ligninolytic extracellular enzymes. Fungal treatment of contaminated soil could be applied with a lignocellulosic substrate: fungi are first grown in sterile or steamed lignocellulose, and then inoculated with the substrate. The substrate enhances survival of the fungi in non-sterile soil. Fungi are also able to degrade organic woody materials in soil. Thus, fungal pretreatment would decrease the amount of organic carbon in contaminated soil and make combustion of the soil more feasible. In a preliminary screening eight fungal strains capable to grow on contaminated wood and able to compete with native microbes was found. In near future the degrading capacity of these fungi for large scale applications will be studied.

Introduction

There is a demand to develop methods for remediation of soil where the structure of soil is preserved and no more toxic chemicals are needed. Wood-degrading and litter-decomposing fungi possess lignin degrading capabilities and form a potential to remediate soil contaminated with recalcitrant organic compounds. These fungi produce non-specific extracellular enzymes and thus they are able to degrade also large and humic bound molecules. Fungal degradation of polycyclic aromatic hydrocarbons (PAH), pentachlorophenol (PCP) and even chlorinated dibenzodioxins and -furans (dioxins) has been proved to occur (Fig. 1, Mori and Kondo 2002, Takada et al. 1996).

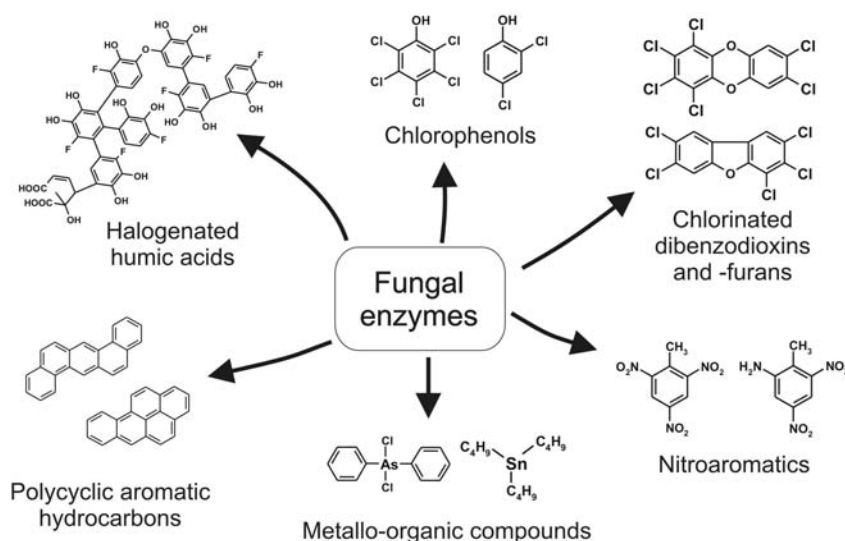


Figure 1
Organic pollutants degraded with enzymes (e.g. manganese peroxidase) produced by ligninolytic fungi.

Currently combustion is the recommended treatment method for dioxin-contaminated soil. However, the high content of organic matter in soil slows down the combustion, and the evolving carbon dioxide wastes may block up the combustion gas treatment process. Wood-rotting fungi, which in nature are the main degraders of wood and organic woody materials,

could remove a substantial amount of organic matter from contaminated soil, and thus, the combustion of soil treated with fungi would be more feasible. The prerequisites for fungi suitable for pretreatment are 1) efficient degradation of all components of lignocellulosic material, 2) tolerance of various pollutants, especially dioxin, 3) capability to survive and grow in non-sterile substrates as well as in the presence of native soil microorganisms. Fungi with dioxin degrading capability would be additionally interesting.

Materials and methods

Our aim has been to improve inoculation techniques to enhance survival and colonisation of fungi in contaminated soil. The selected fungi should quickly colonize substrates and penetrate soil in the presence of soil microbes. An additional carbon source is needed because fungi do not utilize pollutants as substrate. We have tested inexpensive forestry and agricultural wastes to be used as substrates by fungi e.g. bark, straw or flax, and screened their suitability with several fungi (Steffen et al. 2006). Degradation of various pollutants, such as PCP and benzo(a)pyrene (BaP), by fungi has been studied with ¹⁴C-labelled compounds and analysing with liquid chromatography.

To find suitable fungi for soil pretreatment we have screened 35 fungal strains including white-rot, brown-rot and litter-decomposing fungi. In preliminary screening fungi were grown together with dioxin-contaminated wood block on malt agar plates, in order to find out if they tolerate and grow on contaminated wood, and if fungi can compete with native wood microbes. In the first screening wood was treated with H₂O₂, and in the second screening wood was not treated. White-rot fungi selected for screening had caused considerable mass loss in a wood block test of an earlier study (Hakala et al. 2004). Brown-rot and litter-decomposing fungal species, which were known to be either efficient wood degraders in nature or efficient degraders of recalcitrant compounds, were also included into the screenings, especially seven strains of a powerful brown-rotting fungus *Piptoporus betulinus*.

Results

The survival of a fungus in soil is enhanced when it is introduced to soil together with a lignocellulosic substrate. The tested materials give shelter to fungi and help to keep optimal moisture in soil (Steffen et al. 2006). Pine bark was found to be more suitable than straw for most of the white-rot and litter-decomposing fungi. Many fungi can grow even on non-sterile bark, but the growth is faster on autoclaved bark. When fungi were inoculated to non-sterile soil together with bark, they were able to survive, degrade, and even mineralize PCP and five-ring PAHs (Steffen et al. 2006, Tuomela et al. 1999). Fungi have also been found to tolerate high concentrations of heavy metals (Tuomela et al. 2005).

When screening fungi for dioxin tolerance and competitiveness, the most promising fungi of the first preliminary screening of 35 fungal strains were the white-rot fungi *Dichomitus squalens* PO114, *Irpex lacteus* CCB-196, *Physisporinus rivulosus* T241i, and the brown-rot fungi *Poria (Postia) placenta* 280/VTT and four strains of *P. betulinus* (29, 30, A117, and T35). All of these fungi colonized the contaminated wood and no growth of native microbes from the wood was observed.

Discussion

We have found many fungi that fulfil the criteria of fast growth, competitiveness with native soil microbes, and efficient degradation of various pollutants (Steffen et al. 2006, 2003, 2002; Tuomela et al. 2002, 1999). We can conclude that enzyme production and colonization of contaminated soil are prerequisites of successful bioremediation strategies. Organic

pollutants are not only converted but also mineralized by fungi (Steffen et al. 2003, 2002; Tuomela et al. 2002, 1999). Fungi can tolerate non-sterile conditions, which makes it possible to inoculate them to non-sterile soil. Natural fungal strains can be left in the environment and the used substrate can improve the structure of soil.

In near future screening of fungi for maximal organic matter degradation is continuing. White-rot fungi are a group of fungi that has been studied widely but the interest has been focused to species that degrade lignin selectively. Our new approach is to find fungi that could degrade all components of wood. The less studied groups of brown-rot and litter-decomposing fungi will also be included into this study. Later we plan to perform pilot and large scale experiments in order to test fungal technologies in practice.

References

Mori T, Kondo R. 2002. Oxidation of chlorinated 2,7-dichlorodibenzo-*p*-dioxin and dibenzofuran by white rot fungus, *Phlebia lindtneri*. FEMS Microbiology Letters 216: 223-227.

Hakala T, Maijala P, Konn J, Hatakka A. 2004. Evaluation of novel wood-rotting polypores and corticioid fungi for the decay and biopulping of Norway spruce (*Picea abies*) wood. Enzyme and Microbial Technology 34: 255-263.

Steffen KT, Schubert S, Tuomela M, Hatakka A, Hofrichter M. 2006. Enhancement of high molecular mass PAH in contaminated non-sterile soil by litter-decomposing fungi. Biodegradation (in press).

Steffen KT, Hatakka A, Hofrichter M. 2003. Degradation of benzo[a]pyrene by the litter-decomposing basidiomycete *Stropharia coronilla*: role of manganese peroxidase. Applied and Environmental Microbiology 69: 3957-3964.

Steffen KT, Hatakka A, Hofrichter M. 2002. Removal and mineralization of polycyclic aromatic hydrocarbons by litter-decomposing basidiomycetous fungi. Applied Microbiology and Biotechnology 60: 212-217.

Takada S, Nakamura M, Matsueda T, Kondo R, Sakai K. 1996. Degradation of polychlorinated dibenzo-*p*-dioxins and polychlorinated dibenzofurans by the white rot fungus *Phanerochaete sordida* YK-624. Applied and Environmental Microbiology 62: 4323-4328.

Tuomela M, Steffen KT, Kerko E, Hofrichter M, Hartikainen H, Hatakka A. 2005 Influence of Pb contamination in boreal forest soil on the growth and ligninolytic activity of litter-decomposing fungi. FEMS Microbiology Ecology 53: 179-186.

Tuomela M, Oivanen P, Hatakka A. 2002. Degradation of synthetic ¹⁴C-lignin by various white-rot fungi in soil. Soil Biology and Biochemistry 34: 1613-1620.

Tuomela M, Lyytikäinen M, Oivanen P, Hatakka A. 1999. Mineralization and conversion of pentachlorophenol (PCP) in soil inoculated with the white-rot fungus *Trametes versicolor*. Soil Biology and Biochemistry 31: 65-74.

Field Demonstration of Biostimulation and Bioaugmentation for Remediation of Chlorinated Solvents in a Sand Aquifer

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Abstract

In North America, enhanced reductive dechlorination (ERD) has emerged as a cost-effective technology for in situ remediation of chlorinated contaminants. ERD involves the creation of reducing conditions via the injection of fermentable organic substrates that serve as electron donors (biostimulation), and often includes injection of specialized bacteria (bioaugmentation). For chlorinated ethenes such as tetrachloroethene (PCE), trichloroethene (TCE), and cis-1,2-dichloroethene (cDCE), the objective is to stimulate complete dechlorination to ethene. Research indicates that certain bacteria belonging to the genus *Dehalococcoides* (*Dhc*) are required to catalyze complete dechlorination to ethene. At sites where indigenous *Dhc* populations are low or absent, bioaugmentation with *Dhc* cultures has been demonstrated to significantly accelerate treatment. In 2006, the Danish EPA and Funen County sponsored an ERD pilot test at a facility in Odense. The test involved active biostimulation with lactate and bioaugmentation with *Dhc* (KB-1® culture) using a groundwater recirculation system. The test demonstrated that ERD is an effective technology for Scandinavian sites impacted by chlorinated solvents. Within a six month period, chlorinated ethenes were substantially reduced, with concurrent mass balance conversion to ethene. After 200 days, the average molar fraction of ethene in all the monitoring wells was > 50%, indicating substantial treatment throughout the test zone.

Methods and Materials

Site Description. The demonstration site (Rugårdsvej 234, Odense) was chosen from a group of 13 chlorinated solvent sites screened for suitability for ERD pilot testing. The site is a former manufacturing facility where cDCE and vinyl chloride (VC) persist as the primary groundwater contaminants. This contamination occurs in a sand aquifer, as well as an overlying moraine clay deposit. The sandy aquifer within the pilot test area (PTA) occurs at a depth of approximately 11 meters below ground surface, has a thickness of approximately 1-2 meters, and an estimated horizontal hydraulic conductivity of 2×10^{-4} meters per second (m/s). Prior to initiation of the pilot test, cDCE and VC were the primary contaminants, and dissolved phase concentrations of these constituents near the source area were 15 and 5 milligrams per liter (mg/L), respectively. The geochemistry in the PTA, prior to pilot test start-up, was mildly anaerobic, with evidence of iron- and sulfate-reduction, as well as methanogenesis. Ethene was also detected at several wells at concentrations in the range of 100 microgram per liter ($\mu\text{g/L}$), suggesting that an indigenous *Dhc* population was active at the site.

Pre-Design Analyses. Pre-design analyses included bench treatability tests, aquifer pump testing, and groundwater flow modelling. Treatability tests using aquifer material from the site

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determined that a combination of lactate and the *Dhc* culture KB-1® could stimulate rapid and complete transformation of TCE and cDCE to ethene in the site groundwater. Although the detection of ethene in the field data suggested the presence of an indigenous *Dhc* population in the aquifer sediments, biostimulation alone, without bioaugmentation, failed to stimulate dechlorination of cDCE in the laboratory bench test within an objective six month timeframe.

An aquifer step test was performed to estimate pumping yields and the horizontal hydraulic conductivity in the sand aquifer. Injection testing was performed to evaluate the relationship between injection pressures and injection rates in the aquifer. The results of the aquifer test were used to construct a groundwater flow model of the site consisting of four horizontal layers that represented the sand layer for the pilot test, as well as stratigraphic layers above and below the sand layer. The model was calibrated using water level measurements throughout the model domain. The calibrated flow model was used to evaluate alternative ERD designs, all of which involved forced-gradient, closed-loop groundwater recirculation (i.e., extraction of groundwater at the downgradient edge of the target treatment zone, and mass balance injection of groundwater at the upgradient edge of the target treatment zone). Closed-loop recirculation was chosen as the baseline simulation design, as this approach has been shown successful for ERD demonstrations in North America (e.g., Ellis et al. 2000; Major et al. 2002). With the assistance of particle-tracking, model simulations compared travel times within the treatment zone (i.e., between injection and extraction wells), capture efficiency, and aerial extent of treatment over a range of injection/extraction well layouts and pumping rates. Of the pumping layouts evaluated, a layout consisting of one extraction well and three injection wells, operated with a recirculation rate of 3 liters per minute (L/min), yielded the most favorable results in terms of predicted capture, residence time (~ 42 days), and area of influence of injected fluids.

Pilot Test Design. The pilot test system consisted of one extraction well, three injection wells, piping, a system control shed that housed pumps, meters, and reagents, and nine primary performance monitoring wells. The layout of the injection/extraction and monitoring wells is shown in Figure 1. The three injection wells were spaced only a couple meters apart and located immediately upgradient of the chlorinated solvent source area. The close spacing of injection wells was necessary to achieve effective distribution of injected lactate, given the low injection rate of approximately 1 L/min for each injection well.

The aquifer pore volume of the PTA was approximately 300,000 L. The extraction well was located 29.4 m from the injection well, and the hydraulic travel time between the injection wells and extraction well was approximately 47 days based on tracer testing with bromide. Groundwater was pumped continuously from the extraction well at a rate of 3 L/min, amended with electron donor (lactate), and injected into the three injection wells. Biostimulation commenced on 3 November 2005, and lactate was added on a variable pulsed schedule (daily to weekly) to achieve a time-weighted-average concentration of 400 mg/L. After two months operation, strongly reducing conditions were achieved uniformly throughout the PTA. Subsequently, each



Figure 1. Layout of wells in pilot test area: injection wells (I101, I102, and I103); extraction well (AV1); and performance monitoring wells (stars). Distance between I102 and AV1 is 29.4 m.

of the three injection wells was bioaugmented with 9 L of the *Dhc* culture KB-1® (SiREM Laboratories, Ontario, Canada) on 11 January 2006. Bioaugmentation with *Dhc* was included as a component of the ERD design based on the bench treatability test, which found that biostimulation alone did not dechlorinate cDCE in aquifer microcosms over a six month period. The pilot test system operation was conducted for approximately 200 days, terminating subsequent to the final performance monitoring event, on 23 May 2006.

Performance Monitoring. A robust network of monitoring points was installed to support a data-rich evaluation of ERD. The primary performance monitoring network included nine monitoring wells (shown in Figure 1), as well as sampling ports on the extraction well (AV1) and the central injection well (I102). All monitoring points were sampled prior to system start-up operation (baseline sampling event), and approximately once a month for each month of system operation. Samples were analyzed at the Danish Technical University for volatile organic compounds (VOCs), volatile fatty acids (i.e., lactate and its fermentation products propionate and acetate), dissolved hydrocarbon gases (methane, ethene, and ethane), and anions. Certain redox parameters were measured in the field at the time of sample collection, including dissolved hydrogen, oxygen, and ferrous iron. In addition, samples were periodically collected and analyzed for *Dhc* to monitor growth and proliferation of *Dhc* in the PTA. Analyses focused on *Dhc* that contain the vinyl chloride reductase gene (*Dhc-vcrA*), which are present in KB-1® and have been identified as the primary *Dhc* responsible for effective dechlorination of cDCE and VC (Muller et al. 2004). The *Dhc-vcrA* cell counts were performed by SiREM Laboratories.

Results

Within one to two months after the commencement of lactate addition, strongly reducing conditions were established throughout the PTA as indicated by negative oxidation-reduction potentials, consumption of sulfate, and increased production of dissolved ferrous iron and methane (data not shown). Results indicate that lactate was rapidly degraded within a few meters of the injection well, yielding dissolved hydrogen and acetate that provided reducing equivalents to support dechlorination throughout the PTA (data not shown).

Significant reductions in chlorinated VOC concentrations in PTA monitoring points were apparent by day 62, and those reductions continued throughout the course of the pilot test (see Figure 2). Decreases in chlorinated VOC concentrations appeared to be balanced by concurrent increases in ethene concentrations (with trace amounts of ethane). Increases in ethene concentration became evident by Day 62, and the molar fraction of ethene (relative to other chlorinated ethenes present in the sample) in the PTA wells continued to increase throughout the course of test (see Figure 3). At the time of the final sampling event on 23 May 2006, the average molar fraction of ethene in PTA wells exceeded 50%. Concentrations of *Dhc-vcrA* also increased throughout the PTA during the pilot test (data not shown).

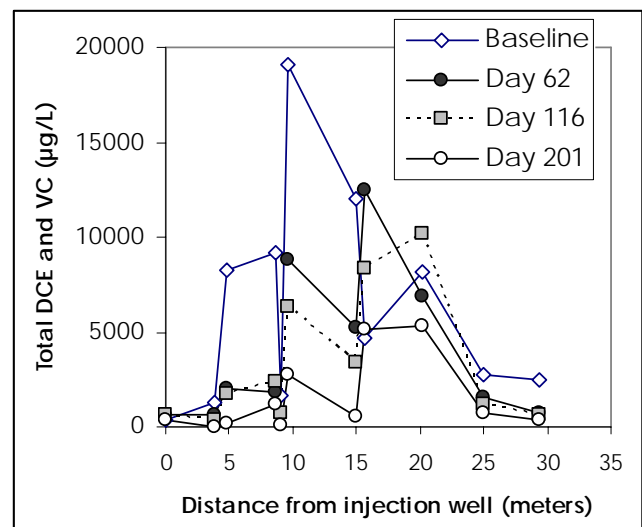


Figure 2. Trends in sum of DCE and VC concentrations in the PTA over the course of the pilot test.

Discussion

The results of this pilot test demonstration confirm that ERD can achieve relatively rapid and substantial reductions in chlorinated VOCs in groundwater, converting cDCE and VC to ethene. While complete destruction of all the cDCE and VC in the sand layer was not achieved during this 6-month test, the results suggest that majority of cDCE and VC in the sand layer was converted to ethene. It is important to recognize that the actual extent of chlorinated VOC treatment achieved by this test may have been significantly greater than can be readily discerned by the data, as potential diffusion of cDCE and VC from the overlying moraine clay (where residual pure phase solvents may still exist) into the sand layer likely provided a steady influx of VOC into the PTA during the course of the test.

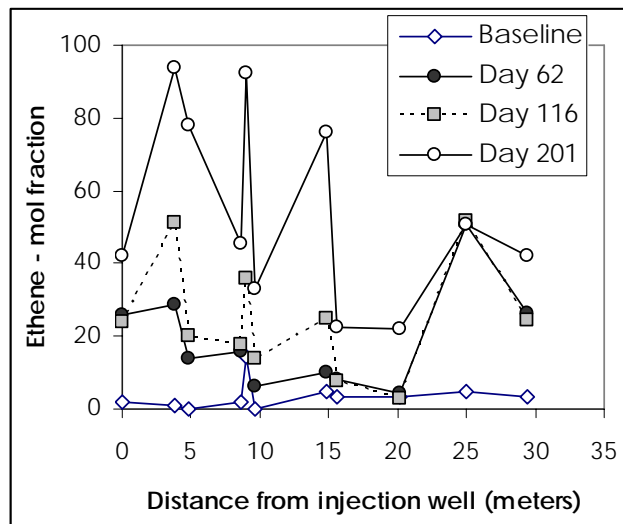


Figure 3. Trends in molar fraction of ethene in PTA wells over the course of the pilot test.

The growth trend in *Dhc-vcrA*, concurrent with increases in ethene concentrations suggests that *Dhc-vcrA* played an important role in the dechlorination of cDCE and VC in this test. It is not clear from the results of this test, however, that bioaugmentation was required to achieve the observed treatment results. Significant ethene generation and growth in *Dhc-vcrA* was observed by Day 62, prior to bioaugmentation with KB-1®. The indigenous population of *Dhc-vcrA* may have played a significant role in the outcome of the pilot test, in spite of results from the pre-design treatability test, which found that biostimulation alone failed to catalyze dechlorination of cDCE to ethene in site aquifer materials. In any case, bioaugmentation with *Dhc* remains an effective design option for ERD at many sites, and has been demonstrated in other cases to significantly accelerate treatment performance (e.g., Ellis et al., 2000; Major et al., 2002; and Environmental Security Technology Certification Program [ESTCP], 2005). At the Rugårdsvej 234 test site, bioaugmentation accounted for approximately 3% of the overall project cost.

References

- Ellis, D.E., E.J. Lutz, J.M. Odom, R.J. Buchanan, C.L. Bartlett, M.D. Lee, M.R. Harkness, and K.A. Deweerd. 2000. Bioaugmentation for accelerated in situ anaerobic bioremediation. *Environmental Science and Technology*, 34(11):2254-2260.
- ESTCP. 2005. Bioaugmentation for Remediation of Chlorinated Solvents: Technology Development, Status, and Research Needs. <http://www.estcp.org/Technology/upload/BioaugChlorinatedSol.pdf>.
- Major, D.W., M.L. McMaster, E.E. Cox, E.A. Edwards, S.M. Dworatzek, E.R. Hendrickson, M.G. Starr, J.A. Payne, and L.W. Buonamici. 2002. Field demonstration of successful bioaugmentation to achieve dechlorination of tetrachloroethene to ethene. *Environmental Science and Technology*, 36(23):5106-5116.
- Müller J.A., B.M. Rosner, G. Von Abendroth, G. Meshulam-Simon, P.L. McCarty, and A.M. Spormann. 2004. Molecular identification of the catabolic vinyl chloride reductase from *Dehalococcoides* sp. strain VS and its environmental distribution. *Applied and Environmental Microbiology*, 70:4880-4888.

REMEDIATION OF THE RESIDUAL SLUDGE FROM SOIL WASHING

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Abstract

Sludge of heavy metal contaminated soil fines is the main process residue from the soil washing process after treatment of heavy metal contaminated soils. Onerous dewatering and expensive deposition of this material severely limits commercial implementation of the soil washing technology. In this work we evaluated the potential of using electrodynamic soil remediation (EDR) for treatment of this material. EDR is an electrokinetic remediation method, where ion-exchange membranes are applied as barriers between soil and electrolytes. During treatment, a direct electric current is applied to the material and ionic species, including heavy metals, are mobilized. Vital advantages are obtained, when treating the sludge in suspension as opposed to conventional EDR and EKR in stationary matrix: the sharp concentration and pH profiles are avoided; the treatment may take place in a continuous process; and process control becomes more convenient. Due to its widespread occurrence, Pb was chosen as a sample heavy metal in this work. Our results show that Pb-contaminated sludge can be decontaminated completely within a reasonable timeframe, and without unacceptable destruction of the soil.

Materials and methods

The fine fraction of an industrially contaminated soil was separated from the coarse fraction by sieving through a 0.063 mm sieve. Pb and cations were analyzed by flame AAS, while samples with concentrations below the detection limit were measured by GF-AAS. Prior to analysis, the samples were digested according to DS259. The Pb concentration in the grain size fractions was determined upon wet sieving through a 0.063 mm sieve followed by dry sieving of the coarse fractions. Samples of each fraction were crushed, and the Pb concentration in each fraction was measured by AAS. All measurements were made in triplicate. Electrodialysis experiments were made in cylindrical Plexiglas-cells with three compartments (fig. 1). The anolyte and catholyte was separated from the soil specimen anion- and cation-exchange membranes respectively. pH in the electrolytes (0.01M NaNO₃) was kept between 1 and 2 by manual addition of HNO₃/NaOH. The liquid to solid ratio (L/S) of the slurry was 4.3 (87g soil and 375ml distilled water), and the current was set to 40 mA (~ 0.8mA/cm²). A detailed description of the setup is found in [1].

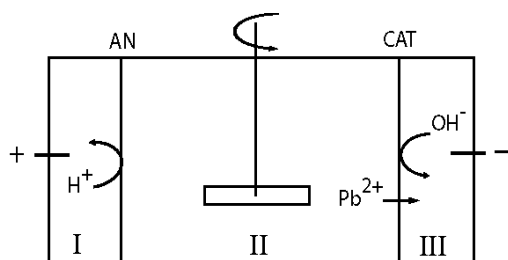


Fig. 1. Schematic view of a cell used for experimental EDR of soil-fines in suspension. AN = anion-exchange membrane, CAT = cation-exchange membrane. I = anolyte compartment, II = compartment containing slurry of soil fines, III = catholyte compartment.

Results

Figure 2 shows how Pb was distributed in the size fractions of ten arbitrary Danish industrially contaminated soils (initial concentrations between 300 and 9000mg Pb/kg dw). Pb clearly concentrated in the finest fraction of the soils.

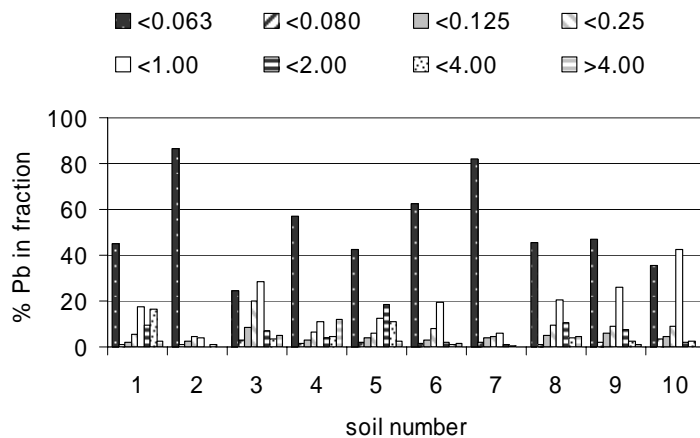
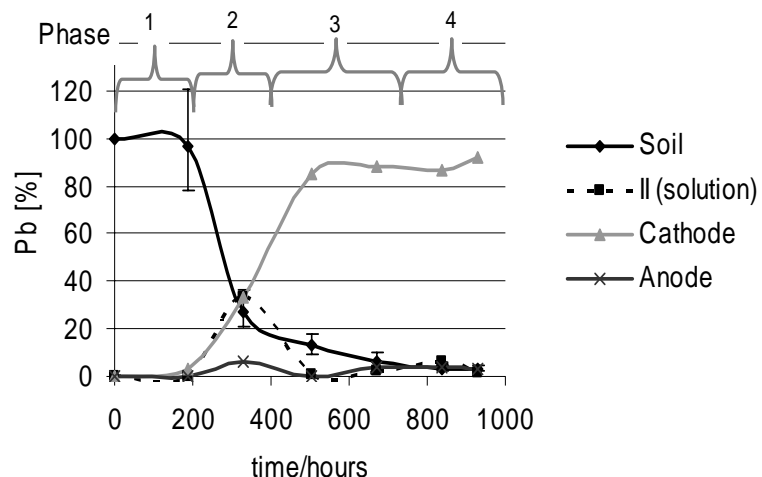


Fig. 2. Pb distribution in grain size fractions of ten industrially contaminated soils. Grain size fraction in mm.

In figure 2 the results of six identical remediation experiments of various duration with the fine fraction of soil 10 (initial concentration 673 ± 101 mg Pb/kg dw) are illustrated. Pb could be

removed from the soil fines with 23 mg/kg as the lowest concentration obtained. This is close to background level, and leaves the soil fines in class 1 (uncontaminated, < 40 mg Pb/kg). The remediation process could be divided into four phases: (1) an initial phase with almost no Pb removal; (2) a phase with a high removal rate; (3) a phase with a low removal rate, and (4) a phase in which removal had ceased. Practically all the Pb was transferred into the cathode section, where >90% precipitated at the cathode. During phase 2, however, a significant fraction of the Pb dissolved in the solution of the soil slurry, from which a smaller fraction diffused into the anolyte. This occurred simultaneously with a sharp pH-decrease from the initial neutral to 1-2.

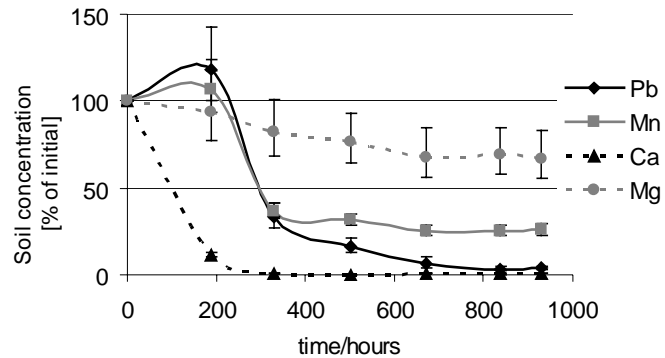
Fig. 2. Fraction of Pb found in each compartment of the electrolytic cell after treatment: in soil (comp. II); in solution (comp. II); in anode and cathode sections, i.e. sum of Pb in membrane, electrolyte, and at electrode (comps. I and III).



Approximately 20% of the soil dissolved during remediation, indicating that not only contaminants, but also soil constituents were affected by the treatment. We investigated the effect of the treatment on specific soil cations (Al, Ca, Fe, K, Mg, and Mn). The most affected element was Ca followed by Mn and Mg, for which results are illustrated in figure 3. Ca was removed completely from the soil prior to Pb extraction, while Mn was removed simultaneously with Pb, however to a smaller extent. Mg was removed at a relatively constant, low

rate throughout the treatment. K, Al and Fe were all less affected by the treatment (data not shown), although Fe and Al oxides acted as buffers at the low pH-level, the final concentration in the soil was >90% of the initial. 17% of the soil consisted of carbonates (dominated by calcium carbonate), which, based on the total extraction of Ca and the low pH, are likely to have been completely dissolved.

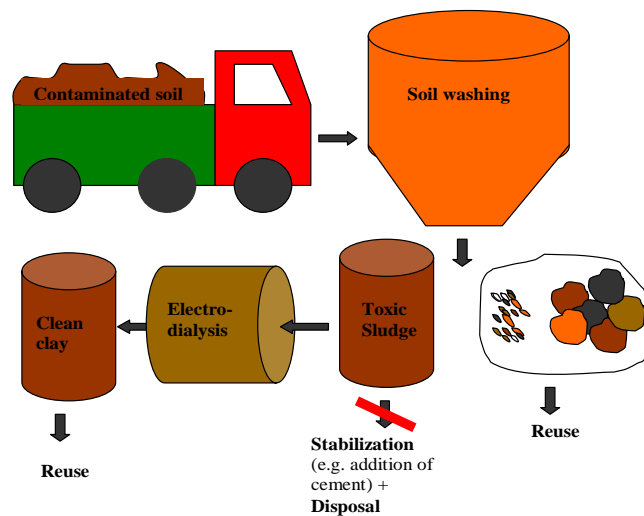
Fig. 3. Effect of EDR on natural Mn, Ca, and Mg in soil. Ca was completely extracted prior to Pb-extraction. Also Mn and to some extent Mg were depleted from the soil.



Discussion

Pb concentrates in the fine fraction of contaminated soils, thus separation of the fine and coarse fractions (i.e. soil washing) would result in a significantly smaller volume of highly contaminated soil fines and a large volume of less contaminated soil. The concentrations in the less contaminated soil fractions were, however, still above the background level, and more than half of the soil fractions remained in category 4 (> 400 mg Pb/kg), thus size-fractionation would not be sufficient for remediation of contaminated soils. A number of works showed how improvements may be obtained by washing with a reagent such as an acid or a complexing agent e.g. [4-6]. Another approach could be size-fractionation followed by density separation of the individual size fractions or vice versa, by which method the more dense grains could be sorted out. This technique would be efficient for soils where the contaminating heavy metals are found concentrated in single particles, as Pb commonly is [7, 8]. By either of the techniques, however, the fine fraction is left after treatment as highly contaminated slurry with a liquid-to-solid-ratio (L/S) of around 4 (fig. 4). In a continuous process, the remediation should proceed at the highest possible removal rate. At the low target concentrations required (during phase 3) the rate could be increased by increasing the current density, because in this phase, the conductivity of the slurry increased significantly.

Fig. 4. Soil washing succeed by electro-dialytic treatment of the toxic residual sludge. The soil washing process should consist of size fractionation and density separation in series, possibly complemented by washing with extracting reagents. The electro-dialytic process could take place as a continuous process. The clean soil fines could be reused e.g. in ceramics (bricks and tiles), lightweight-expanded aggregates etc.



In addition to the illustrated results, we showed that the process works equally well when the soil-fines are suspended in tap-water, while the remediation may be

enhanced by addition of nitric acid at an increased current density (data not shown). A detailed discussion of the processes governing the remediation may be found in [1, 9]. During phase 2 of the remediation, pH decreased sharply to 1-2, and the question arose if simple acid extraction would be as efficient as EDR. We tested this hypothesis on the ten soils illustrated in figure 2 (by extraction with HNO₃). The result obtained was that the 97% extraction obtained from soil 10 by electro dialysis could not be obtained by acid extraction even at pH as low as 0.4. In general pH had to be as low as 0-1 before efficient Pb extraction was obtained. In addition, acid extraction leaves a large volume of Pb contaminated liquid for treatment, whereas the majority of the Pb (>90%) was precipitated at the cathode for easy handling and disposal after electro dialytic treatment. The optimal remediation method would remove the contaminant, while leaving the soil itself untouched. Such a method has not yet been developed, and during EDR 20% of the soil was dissolved. Calcium carbonate was, however, the major affected component, while only 3% of the remainder soil constituents were dissolved after completed remediation. The soil may thus be almost completely reconstituted simply by addition of lime.

References

- [1] P.E. Jensen, L.M. Ottosen and C. Ferreira, 2006. Kinetics of Electro dialytic Extraction of Pb and Soil Cations from Contaminated Soil Fines in Suspension, *Journal of Hazardous materials* (in press, doi: 10. 1016/j. jhazmat. 2006. 05. 0732006).
- [2] G. Maini, A.K. Sharman, C.J. Knowles, G. Sunderland and S.A. Jackman, 2000. Electrokinetic remediation of metals and organics from historically contaminated soil, *Journal of Chemical Technology and Biotechnology* 75: 657-664.
- [3] L.M. Ottosen, H.K. Hansen, G. Bech-Nielsen and A. Villumsen, 2000. Electro dialytic remediation of an arsenic and copper polluted soil - Continuous addition of ammonia during the process, *Environmental Technology* 21: 1421-1428.
- [4] S.A. Wasay, S.F. Barrington and S. Tokunaga, 1998. Remediation of soils polluted by heavy metals using salts of organic acids and chelating agents, *Environmental Technology* 19: 369-379.
- [5] S. Tandy, K. Bossart, R. Mueller, J. Ritschel, L. Hauser, R. Schulin and B. Nowack, 2004. Extraction of heavy metals from soils using biodegradable chelating agents, *Environmental Science & Technology* 38: 937-944.
- [6] N.N. Sauer, D.S. Ehler and B.L. Duran, 2004. Lead extraction from contaminated soil using water-soluble polymers, *Journal of Environmental Engineering ASCE* 130: 585-588.
- [7] P.E. Jensen, L.M. Ottosen and A.J. Pedersen, 2006. Speciation of Pb in industrially polluted soils, *Water Air and Soil Pollution* 170: 359-382.
- [8] C. Lindskov and F. Oemig. 2001. Afprøvning af jordvask. Miljøprojekt nr. 616, Miljøstyrelsen, Miljøministeriet, Copenhagen.
- [9] P.E. Jensen, L.M. Ottosen and C. Ferreira, 2006. Electro dialytic Remediation of Pb-Polluted Soil Fines (< 63µm) in Suspension, *Electrochimica Acta*, in press.

**TITLE****SPEAKER**

A Methodology to Prioritize Remediation Technologies

Jaana Sorvari, Finnish Environment Institute (SYKE)

Experience from the Shell Sola refinery remediation project, Norway

Ola Bruskeland, Multiconsult AS

Spatial risk assessment at contaminated sites; Benefits and limitations

Niklas Törneman, SWECO VIAK AB

Quality management of site investigations

Lizzi Andersen, DHI Water & Environment

Updating of Norwegian soil quality guidelines and inclusion of new substances

Mona Weideborg, Aquateam

Risk-based prioritization of contaminated sites

Lars Rosén, SWECO VIAK AB

Human exposure from contaminated areas in relation to background exposure – a comparison between different sources

Annika Åberg, Umeå University

What do you mean by Bioavailability?

Rodney L. Stevens, Göteborg University

A METHODOLOGY TO PRIORITIZE REMEDIATION TECHNOLOGIES

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Abstract

In the selection of remediation methods for a contaminated site, the decision criteria include achievable risk reduction, costs, environmental effects and other factors such as psychological and socio-cultural effects, e.g. loss of cultural heritage, and imago aspects. These decision criteria also determine the eco-efficiency of a remediation option. In our PIRRE¹-project, we produced a tool to evaluate these criteria and tested it with fictitious model sites, e.g. a gasoline station and an outdoor shooting range. For these model sites we created risk management scenarios and created site-specific data based on equivalent factual remediation cases and by interviewing experts. The weights for the decision criteria were set by experts representing different stakeholders using the direct weights –technique. The primary results were processed using the Hipre software based on the multi-attribute value theory (MAVT). The mean values of the weights mean values of the weights were used to identify the preferred remediation options. The procedure we used proved to enable efficient communication between different stakeholders, e.g. risk managers and authorities. The calculation tool also makes it possible to reveal the data gaps and problems (e.g. in quantifying the risks or evaluating costs) involved in the comparison of risk management options.

Introduction

In Finland, the decisions on risk management of contaminated sites have typically been based on generic guideline values, which do not consider site-specific risks. At the same time, soil excavation and *ex situ* treatment or disposal is still the most common remediation method. Excavated soil, either treated or untreated, is considered as waste and it is mainly disposed or reused on landfills while recycling elsewhere is minimal. The eco-efficiency of the dig and dump method and remediation based on generic guideline values has been questioned. At the end of 2003 we launched a project called PIRRE¹ to study these issues. The main goal of PIRRE -project has been to promote the realization of eco-efficiency in contaminated land management (CLM). While eco-efficiency has been studied previously in different contexts and in various branches in Finland, our project is the first attempt to study it systematically in the context of CLM.

The eco-efficient management of contaminated sites can be interpreted as comprising of activities, with which the land use restrictions, ecological and health risks, and overall adverse environmental effects are minimized using minimal amount of resources, e.g. time, money, manpower, equipment, energy, and natural resources. In practice, the interpretations, however, vary and besides ecological and economic dimensions, eco-efficiency can also be understood as a wider concept including socio-cultural aspects. As such it approaches the definition of sustainable development. The means to increase eco-efficiency include reduction of material consumption, reduction of energy use, and increasing of reusability of materials, among others. In the CLM context this could mean, e.g. enhancement of the reuse of contaminated soil.

Within the PIRRE –project, we developed a decision support tool originally based on the Dutch REC² system. This calculation tool allows determination of the eco-efficiency of different remediation options. We included also decision analytical methods which allow the prioritization of remediation alternatives based on the preferences set by different stakeholders. We tested the final decision support tool with fictitious model sites and organized a seminar focused on giving weights to the different decision criteria. This paper presents a short overview of this study.

¹ PIRRE = Eco-efficient risk management of contaminated soil and groundwater. For further information on the project please visit: <http://www.environment.fi> > Finnish Environment Institute > Research > Research projects and results > Eco-efficient risk management...

² REC comes from the Risk reduction (R), Environmental merit (E), Costs (C). The tool is available in http://www.ivm.falw.vu.nl/Research_projects/index.cfm > Climate, water and spatial analysis > REC

Material and methods

The PIRRE decision support tool includes four separate modules (decision criteria) to assess the achievable risk reduction, costs, environmental effects and other factors (e.g., psychological and socio-cultural impacts, imago aspects). The fictitious model sites we created for testing the tool included a former gasoline station and an outdoor shooting range (Table 1). For these sites, we created several risk management scenarios including "traditional" *ex situ* and more novel on site and *in situ* remediation techniques (Table 2).

Table 1. Description of the studied fictitious model sites.

	Size, m ²	Contaminants	Location	Land use scenario	GW involved
gasoline station	15,000	PHCs	urban	housing	yes
shooting range	160,000	Pb (As, Sb)	rural	recreation (natural state), groundwater uptake	yes

GW = groundwater, PHCs = petroleum derived hydrocarbons

Table 2. Remediation alternatives for the studied model sites. GW = groundwater.

A. Shooting range

Remediation alternative	Method	Remedial targets	Volume of soil and GW treated (m ³)
Alternative 0	No soil remediation; closure of water intake, building of a new water work	-	-
Alternative I	Soil excavation + landfill disposal; closure of water intake, building of a new water work	Lower guideline values for soil (old)	Soil: 45,000
Alternative II	Soil excavation + landfill disposal; closure of water intake, building of a new water work	Upper guideline values for soil (new)	Soil: 16,500
Alternative III	Soil washing + reuse on site; closure of water intake, building of a new waterwork		Soil: 45,000
Alternative IV	Top soil including the shots excavated, shots recycled + land use restricted; GW treated <i>in situ</i> by a reactive wall	No target for soil, GW below the quality standards for domestic water	Soil: 1,300
Alternative V	No soil remediation, land use restricted; GW treated on site (at water work) by Metclean technique	No target for soil, GW below the quality standards for domestic water	-
Alternative VI	No soil remediation, land use restricted; GW treated on site (at water work) by membrane filtration	No target for soil, GW below the quality standards for domestic water	-

B. Gasoline station

Remediation alternative	Method	Remedial targets	Volume of soil and GW treated (m ³)
Alternative 0	No remediation; closure of water intake, building of a new water work	-	-
Alternative I la. lb. lc.	Soil excavation + a. soil composting and reuse on site b. landfill disposal c. combustion off site GW treated <i>in situ</i> by adsorption to activated carbon	Soil limit values (old); GW: BTEX < 10 µg/l, TVOC 1000 µg/l, heavier 1000 µg/l,	Soil: 805 GW: ca. 500
Alternative II (a, b, c)	See Alt. I (a, b, c)	Lower guideline values for soil (old), GW: see Alt. I	Soil: 1,978 GW: see Alt. I
Alternative III	Monitored Natural Attenuation (MNA)	Final concentrations in soil and GW defined on the basis of data from a Finnish research project	-
Alternative IV	Soil vapor suction (6 months) & catalytic combustion + MNA	See Alt. III	Soil: 0 GW: 360

To determine the values for the main decision criteria and subcriteria in different remediation options, we created site-specific data on the basis of equivalent factual remediation cases and by interviewing experts. The total time considered was 30 years in all cases. In the case of the subcriteria under the main criterion “other impacts” we used a qualitative scale divided in seven impact categories (significant/moderate/minor positive impact – no impact – significant/moderate/minor negative impact) representing values between +3 and -3.

In order to prioritize the remediation alternatives, we asked a group of experts to value the decision criteria. The invited experts involved representatives of different stakeholders, i.e. consultants, authorities, researchers, landowners, and experts in the Finnish Environment Institute. In the valuation we used the direct weights –technique. The primary results were processed using the Hipre software that aggregates the values and weights according to the multi-attribute value theory (MAVT). The values of the decision criteria were normalized to the extremes prior to the calculation of the weighted values. As a result, the preference of each remediation alternative is shown by a total preference value, i.e. the higher this value is the better (= higher preference) is the alternative.

Results

The results showed the preferred remedial options for the fictitious model sites (Fig. 1). To calculate the final total preference values we used the arithmetic mean values of the weights given for different decision criteria by all experts.

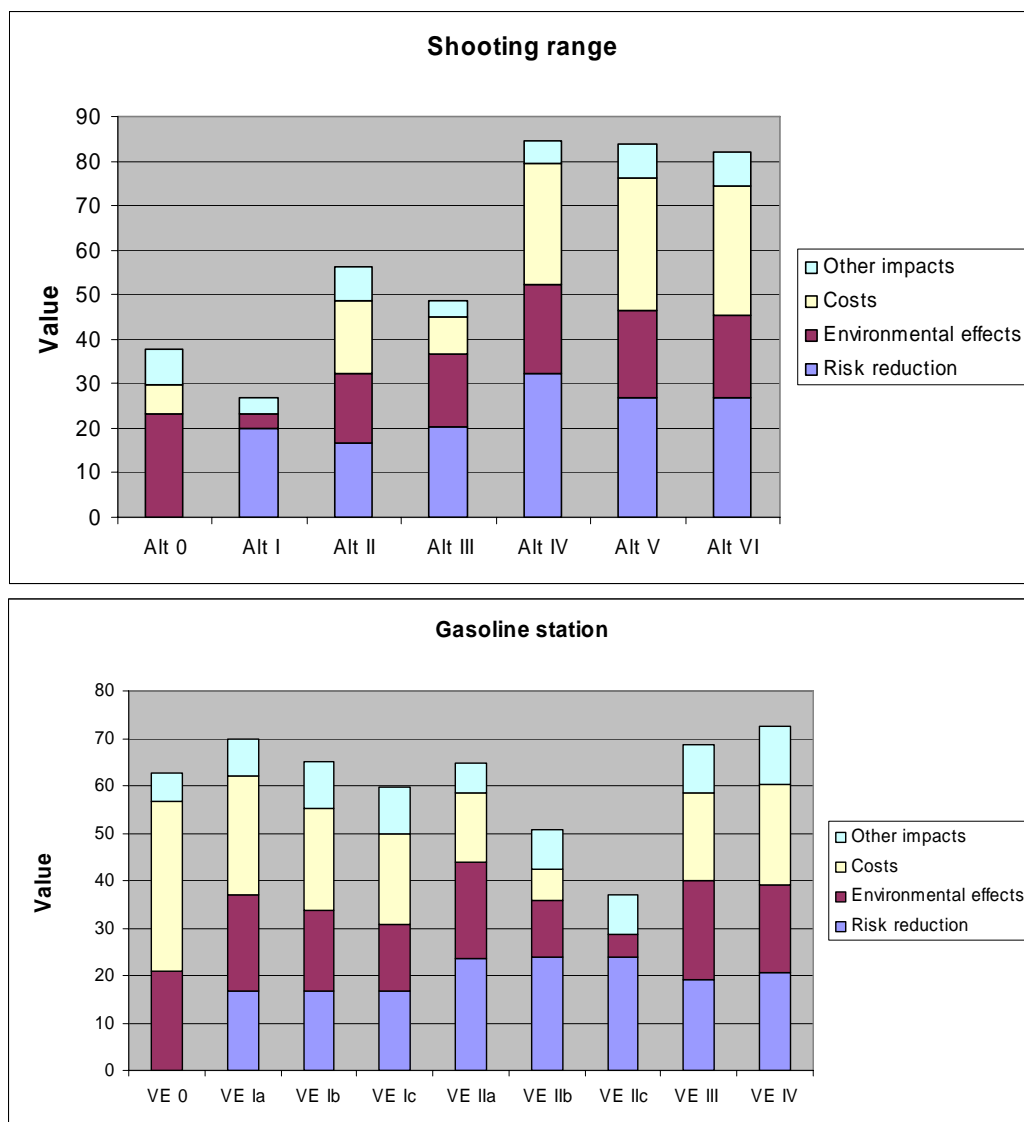


Figure 1. Preference of the remediation options of the fictitious model sites (shooting range, gasoline station) and the proportion of each decision criteria in the final total preference value.

In the case of the shooting range, the remediation alternatives referring to soil washing (Alt. IV) and land use restrictions with groundwater treatment at water works (Alt. V and VI) gained almost equal preference values. Costs and risk reduction were clearly the most important decision criteria. In the case of the fictitious gasoline station, the Monitored Natural Attenuation (MNA) combined with soil vapor suction combined with catalytic treatment proved to be the most preferred alternative. Soil excavation combined with composting on site gained a slightly lower preference value. In the case of the gasoline station, besides “costs” and “risk reduction”, the decision criteria “environmental effects” also proved to be a significant decision criteria contributing to the final preference value. Here, the main subcriteria were “space use” (area of the land not usable due to contamination or ongoing remediation activities), “volume of waste generated” and “soil loss” (soil that is not reused but disposed as waste).

It has to be noted that the results apply only to the sites defined in our study. The results are very sensitive to the variability of the cost estimates, in particular. Furthermore, in the case of novel remediation methods, e.g. MNA (gasoline station) and a reactive wall (shooting range), the data on costs and the magnitude and time scale of risk reduction include high uncertainty since they are based on single experimental projects.

Discussion

The decision support tool developed proved to be suitable for the determination of the most preferable remediation option. The results can also be used to identify the most eco-efficient alternative, i.e. the alternative with the best benefits (output) – resources (input) ratio.

The weights set by the experts varied considerably reflecting individual preferences and differing perspectives of different stakeholders. This variation may also result from the difficulties to comprehend the problem and to consider the characteristics of the specific sites. Therefore, attention should be paid on proper and detailed problem formulation. Weighting of some decision criteria also proved to be somewhat problematic due to the difficulties in comparing the criteria, e.g. the volume of waste generated against the emissions to air or space use. It is also important to stress that in the weighting of the subcriteria, the main criterion has to be kept in mind. For example, when weight is given to the criterion “volume of waste generated” which is under the main criterion “environmental effects” only the environmental aspects but not, e.g. the costs, must be considered.

It is also evident that weights should be set site-specifically taking into account the type, magnitude, and scope of contamination as well as land use and environmental conditions. Due to the site-specificity, the results are not applicable to other situations. However, the weights can be adapted in the case of equivalent sites. Furthermore, the tool itself is useful particularly in the case when none of the available remediation options can be clearly prioritized. It also enables the use of qualitative data and consideration of subjective and differing perspectives in decision making. Moreover, it facilitates communication and information exchange between different stakeholders, e.g. risk managers and authorities, and provides means for public participation. This way conflicts which can delay risk management actions could be avoided.

The results of the calculations also revealed the critical data gaps and problems (e.g. in quantifying risks and evaluating costs) and their influence on the final output used as the basis of a CLM decision. If quantitative data is not available it can be replaced with qualitative data which can be scaled.

In the future, we plan to test the system with factual sites in order to develop it more usable. Here, we will try to consider the limitations of the data available. Moreover, we will perform more detailed sensitivity analyses.

PRACTICAL USE OF RISK BASED REMEDIATION TARGETS FOR HYDROCARBON FRACTIONS ACCORDING TO THE HAZARD INDEX PRINCIPLE. EXAMPLE FROM THE SHELL SOLA REFINERY REMEDIATION PROJECT, NORWAY

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Abstract

The A/S Norske Shell Oil Refinery near Stavanger, Norway, was closed down after 32 years of operation. The refinery ground had become contaminated by crude oil, a range of refined petroleum hydrocarbons and blended products and some landfilled refinery waste. Soil and groundwater remediation works started in November 2001 and was completed in January 2005.

Due to the variations in the composition of the hydrocarbon contaminants across the site, one "total TPH" criterion could not be used to determine the need for remediation or to define the remediation target. Instead, a simplified fractionated approach based on the hazard index principles recommended by the Total Petroleum Hydrocarbon Criteria Working Group (TPHCWG), was used to evaluate the hazard due to the soil oil content. The remediation contractor carried out fractionated TPH analyses of all soils before and after treatment and calculated the corresponding hazard index, which had to be < 1 for the subject soil to be acceptable for reuse on the site. A separate evaluation of PAH content was also carried out. Approximately 8200 soil samples were analysed by the contractor during the remediation period.

The paper is presented by the consultant who has been assisting Shell with regulatory matters and coordination of the risk assessments.

Introduction

The Shell Oil Refinery at Sola was closed down in April 2000. The landholding was subsequently sold for redevelopment. Shell was obliged to renovate the site to enable both recreational and industrial future land use anywhere on the site. The remediation was completed in January 2005. Approximately 1,2 million tons of soil were excavated and classified, and 460 000 tons needed treatment, which was carried out on site. The site is now being developed as a national harbour.

The refinery site is a rocky peninsula facing the North Sea. The process area, most tankfarms, pipelines, and sludge / waste handling areas were situated on native soil (sand, silt and clay) and fill of rock and imported sand, and with groundwater level 1 - 2 m below grade. Oil contamination had impacted the soils in these areas to below the groundwater level. Typical soil TPH values varied from 2 000 to > 10 000 mg / kg. Groundwater was also contaminated, but there was no evidence of active contaminant migration of environmental concern to the marine environment.

The environmental risk and remediation target assessment for the oil contaminated soil at the site were initially developed according to the Norwegian Pollution Control Authority (SFT); report 99:06, "Guidelines on risk assessment of contaminated sites". For oil products, the guidelines are based on the BTEX components and six fractionated groups of aliphatic hydrocarbons. The fractions and their assigned toxic properties are as identified by the Total Petroleum Hydrocarbon Criteria Working Group (TPHCWG).

Due to the complex hydrocarbon contamination at the Shell Sola site, the SFT requested the development of additional toxicity acceptance criteria for other aromatic hydrocarbons than the BTEX-components. This was done for six fractions in accordance with the TPHCWG recommendations.

The risk assessment concluded that the human health risk would be controlling the soil remediation target values. However, in addition to human health risk, the remediation criteria for the site on land also addressed terrestrial ecology. The ecological criteria were developed through comparative, ecotoxicological testing of uncontaminated soils, contaminated soil and treated soils. The results of the tests were of importance for the evaluation of the reuse of the different treated soil fractions and the evaluation of untreated, contaminated soils.

Results

The final remediation target values combined the risk based values for the site zoning categories of industry and recreation, as shown by the last column in table 1. The risk (toxicity) of inhalation of soil gas in future buildings decided the remediation criteria for BTEX and most aromatic and aliphatic petroleum hydrocarbon fractions, whereas direct exposure to soil and dust decided the PAH criteria (cancerous properties).

Table 1. Risk based soil remediation target values (mg/kg dry matter)

Chemical	SFT Norm ¹ Most sensitive land use	Industry		Recreation	Combined Industry and Recreation
		Open surfaces	Paved / built surfaces		
PAH total	2	11	2868	18	11
Benzo(a)pyren	0.1	0.67	179	1.1	0.67
Benzene	0.005	312	0.05 ⁴	474	0.05 ⁴
Toluene	0.5	²	1.4	²	1.4
Ethylbenzene	0.5	²	1.8	²	1.8
Xylene	0.5	²	2.7	²	2.7
Aromatics >C07-C08	None	²	15	²	15
Aromatics >C08-C10	None	8400	20	2600	20
Aromatics >C10-C12	None	8400	110	2600	110
Aromatics >C12-C16	None	8400	570	2600	570
Aromatics >C16-C21	None	6300	4400	2000	2000
Aromatics >C21-C35	None	6300	²	2000	2000
Aliphatics > C5-C10	7	²	24	²	24
Aliphatics >C10-C12	30	²	117	²	117
Aliphatics >C12-C16	100 ³	²	561	²	561
Aliphatics >C16-C35		²	²	²	²

Notes:

1) SFT Norm: National quality norm for soil to most sensitive land use (residential areas and soil for unrestricted disposal).

2) values > 10.000 mg/kg dry matter

3) based on ecotoxicological values

4) Value for detection limit, in line with international screening values for contaminated soil.

- These target values are valid for soil down to groundwater level
- Soil remediation below the groundwater level may be limited to removal of any mobile oil contamination that may cause free phase product formation on the groundwater.
- Soil down to 0.5 m depth shall meet additional ecological requirements.

Practical application of the remediation criteria for hydrocarbons

For petroleum hydrocarbons, each remediation target value (SSTL) defines the concentration above which the content of that hydrocarbon fraction in the soil constitute an unacceptable threat to human health. The fractionated SSTL values are not additive, as each value has been worked out under the assumption that no other hydrocarbon fractions are present.

As the soil contamination consists of a mixture of several hydrocarbon fractions, the mass fractions of the contaminants present on the site also have to be considered to determine the remediation criterion. The "inverse weighted average" method as proposed by the TPHCWG was used for these calculations.

The total petroleum hydrocarbon criterion $SSTL_{TPH}$ defined by this method is: **$SSTL_{TPH} = C_{tot} / HI$**

where: C_{tot} = total concentration of all fractions

$$HI = \text{Hazard Index} = \sum_{i=1}^n HQ_i = \sum_{i=1}^n \text{Min} \left[\left(\frac{c_i}{SSTL_i} \right); \left(\frac{C_{sati}}{SSTL_i} \right) \right]$$

HQ = Hazard Quotient, for each TPH fraction

c_i = concentration of fraction "i"

$SSTL_i$ = the site specific target level for fraction "i"

C_{sati} = upper-bound value, the saturation concentration for the soil source area for the TPH fraction "i".

C_{sat} is the contaminant concentration in soil at which absorptive limits of soil particles, solubility limits of soil pore water, and saturation limits of soil pore air are reached. For most petroleum hydrocarbons, C_{sat} is much lower than the concentration of hydrocarbons at which mobile free product or separate phase would be present.

Because the formula for the $SSTL_{TPH}$ criterion include the mass-fraction properties of the contaminant, one single criterion for the Sola refinery site with a variety of products, would be very conservative and unpractical. Calculations showed that remediation would be required for total concentrations varying between 100 and > 2.500 mg/kg, depending on the actual hydrocarbon product mixture. But use of the full detailed criteria would require a comprehensive program of chemical analyses and calculations for the management of the remediation works, which would also be costly and impractical.

Based on evaluation of the results of the Phase II ESA investigations and some supplementary analyses of all the defined TPH fractions, it was found that a simplified fractionated approach based on the following parameters would be sufficiently accurate for the Shell Sola site:

Table 2. Recommended simplified screening levels for TPH (sum of aliphatic and aromatic components). Human health considerations

TPH fraction	Simplified SSTL Industrial closed (mg/kg) (from aliphatic fraction values)	Site-specific C_{sat} values mg/kg (from aromatic values)
>C6-C10	24	1237
>C10-C12	117	565
>C12-C16	561	262
>C16-C35	5000 ¹	92

Note: 1) Not a risk based value, but an upper limit imposed by the SFT

Any soil with a hazard index HI > 1 must be treated. The calculations do not have to be carried further than to the HI step to determine the need for remediation. In addition the soil has to satisfy the benzene and PAH criteria shown by table 1.

Results from the remediation works

During the practical remediation works, one sample pr every 100 m³ of soil < 50 mm was analysed chemically by method GC/FID to evaluate the need for treatment with respect to oil components. Each 100 m³ of treated soil was also tested. The analyses comprised the BTEX components, the four TPH fractions given by table 2 and PAH. A simple PC-program calculated the HI-value of each sample, and flagged any sample with HI >1 (in need of remediation).

The pre-remediation evaluation of the minimum number of hydrocarbon fractions necessary to define satisfactory soil quality indicated that:

- An evaluation based on 3 TPH values (>C6-C10, >C10-C12, >C12-C35) detected all samples with an HI > 1, but tended to underestimate the HI value and could thus possibly "approve" unacceptable soil
- The chosen evaluation based on 4 TPH fractions (see table 2) gave the best agreement with the evaluation based on both aliphatic and aromatic fractions, and seemed to err on the safe side

This was studied by some further testing (a total of 24 samples) during the remediation, mostly on treated soils and soils with a hydrocarbon content near the HI = 1 concentrations. The results seem to confirm the initial evaluation.

- In 63 % of the samples the HI based on the 4 TPH fractions (HI-4) was higher than the value based on 8 fractions (HI-8)
- In two cases the HI-4 approach caused disapproval of an acceptable soil by the HI-8 method
- In one case the HI-4 approved a soil found to be unacceptable (marginally) by the HI-8 method. In that case the content of heavy aromatics constituted approximately 30 % of the total hydrocarbon content, which is untypical for this site.

For 14 of these samples the HI-3 value (based on 3 TPH values) was also calculated, as summarised by table 3.

Table 3. Evaluation of the number of fractions required to calculate the Hazard Index (HI)
Average results

TPH mg/kg	Sum TPH from 8 aromatic / aliphatic fractions	% content of aromatic fractions	HI-3	HI-4	HI-8 ¹
1194	1424	15	1.84	2.19	1.25

Note: 1) HI-8 = Aliphatics: >C5-C10, >C10-C12, >C12-C16, >C16-C35. Aromatics: C8-C10, >C10-C12, >C12-C16, >C16-C35

The main reason why the HI-4 method seems to give less accurate HI values compared to the HI-3 method (see table 3) is the default use of C_{sat} values for aromatic fractions, which are 4 to 7 times higher than the C_{sat} values for aliphatic hydrocarbons of the same fraction interval. For the investigated samples with a content of aromatics mostly varying between 10 and 20 %, use of aliphatic C_{sat} values would have given better accuracy for the simplified calculations, but with a risk of not detecting the few borderline samples with untypically high aromatics content. Optimum choice of C_{sat} value for use with the HI-4 method could have been evaluated for some of the sub-areas with relatively uniform contamination properties, to improve the accuracy of the method.

Discussion

- The fractionated criteria for acceptable content of petroleum hydrocarbons, gave a more accurate evaluation of the remediation needs than the standard Norwegian risk based method. This was important for a site with considerable variations in the contaminant product composition and concentrations.
- Full use of all the 13 fractions according to the TPHCWG method, or the reduced packet of 8 and fractions tried in this project is impractical at the remediation stage, due to the time and cost required for the comprehensive fractionated analyses. Speedy results from the chemical testing laboratory was a key issue to avoid delays in the remediation works
- The simplified approach adopted by the Shell Sola project seems on the safe side and satisfactory with regards to limiting the amount of soils needing treatment.
- Use of the full 13 (or 8) fraction system for some selected samples could possibly have resulted in a more accurate and cost saving evaluation of some borderline soils and assisted the management of biological treatment processes.

References

Norwegian State Pollution Control Authority (SFT): Guidelines on risk assessment of contaminated sites. SFT report 99:06. TA-1691.

Total Petroleum Hydrocarbon Criteria Working Group (TPHCWG): Volume 5; Human Health Risk-Based Evaluation of Petroleum Release Sites: Implementing the Working Group Approach, Amherst Scientific Publishers, June 1999.

DNV Consulting. Shell Refinery. Final report for remediation of contaminated ground. June 2005 (in Norwegian)

Main participants in the Shell Sola contaminated ground remediation project:

- IWACO BV (Netherlands) with sub-contractor Norconsult AS (Norway); Main environmental site investigations (Phase II ESA), environmental risk assessment, evaluation of remediation needs.
- NIVA (Norwegian Institute for Water Research); Marine investigations
- Multiconsult AS (formerly NOTEBY AS); Environmental advisor to Shell; Phase I ESA, management/coordination of risk assessments, permitting and regulatory matters, preparation of tender documents, works supervision, supplementary ESA II investigations, marine environmental surveys (with NIVA).
- T. Stangeland Maskin AS, Contractor for the remediation works, with subcontractors SITA Remediation and DNV Consulting

Spatial risk assessment at contaminated sites; Benefits and limitations

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Abstract

The following short paper uses a number of site remediation projects to exemplify the difference between focusing on the spatial distribution of contaminants and the spatial distribution of risks at contaminated sites. It is shown that focusing on risks rather than concentrations can lead to other considerations regarding the extent of areas of concern, placement of new sampling points and the effect of risk reduction measures.

Background

A common property of contaminated sites is an inherently high spatial variability of soil properties including soil contaminants. Since exposure to soil contaminants is connected to contaminant concentrations in the soil it may be concluded that ecological and human risks and contaminant concentrations exhibits similar spatial variability. However, there are complications that may obscure this connection:

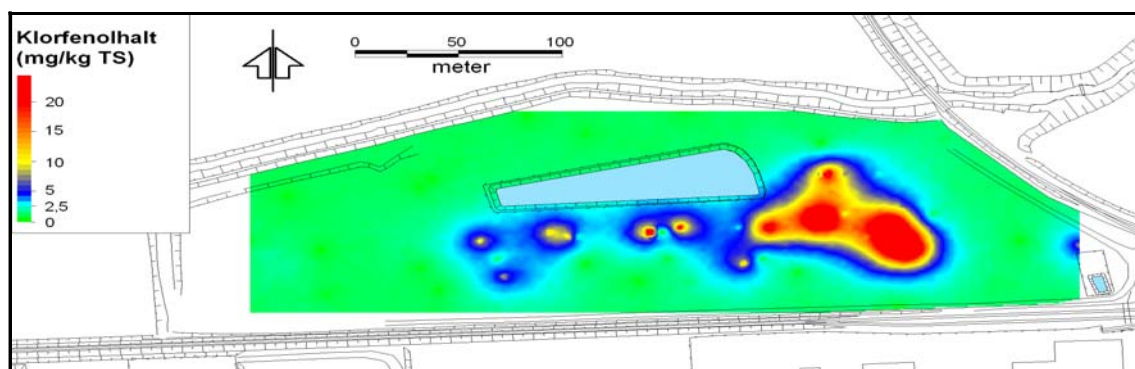
- The inherent spatial variability of risks is obscured because the risk assessment process leads to the calculation of one guideline value that is representative of the whole site
- Risk concentrations (i.e. guideline values) are well below the median concentrations at the site which uncouples the presence of contaminants hot spots from the presence of risk hot spots
- Apart from contaminant concentrations there may be other parameters that influence environmental risks

Here I will present some practical examples that illustrates these complications

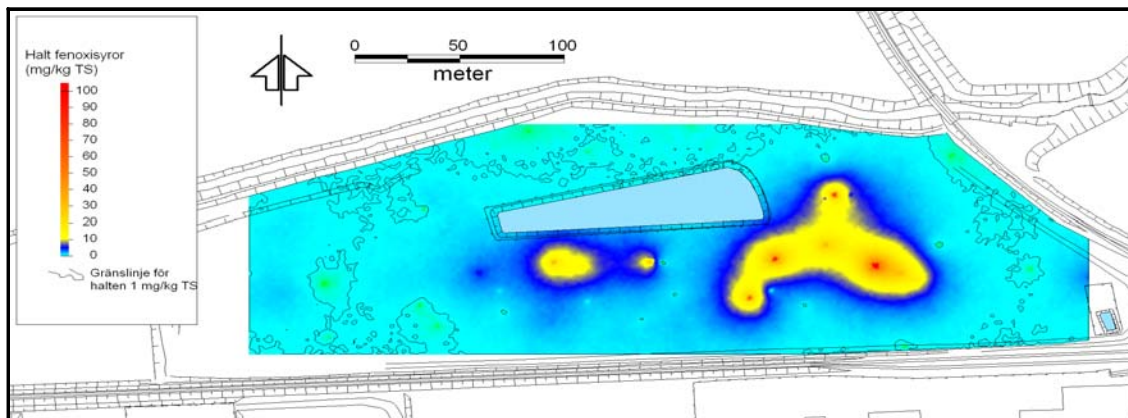
Example 1, BT Kemi

BT Kemi is a former pesticide manufacturing site in Southern Sweden where the soil is heavily contaminated with chlorophenols, chlorocresols, phenoxy acids and dioxins. Site specific guideline values for these compounds have been calculated based on site specific data regarding exposure pathways and exposure parameters.

Using mostly geostatistical interpolation techniques the measured concentrations of these compound groups were mapped as shown in figure 1 and 2 for chlorophenols and phenoxy acids at the site. Chlorophenols, dioxins, phenoxy acids and chlorocresols had very similar spatial distributions.



Figur 1. Interpolated chlorophenol concentrations at the BT kemi site

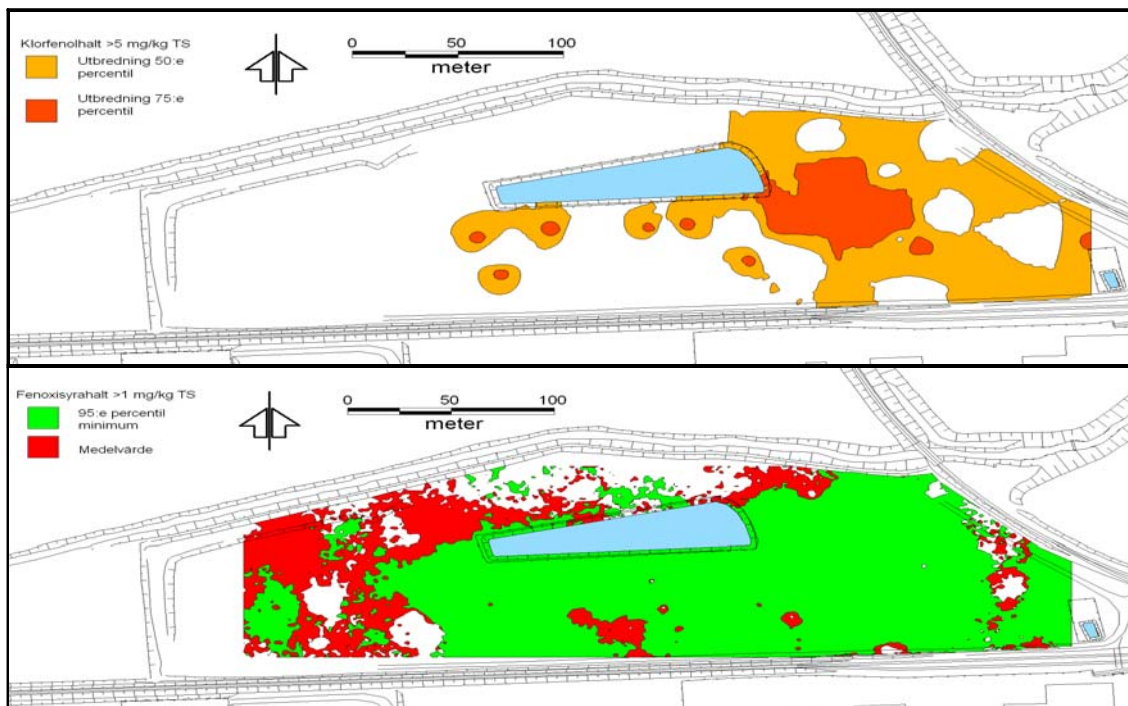


Figur 2 Interpolated phenoxy acid concentrations at the BT kemi site

Similar interpolation techniques were used to map the areas where the calculated guideline values were exceeded. This is shown for chlorophenols and phenoxy acids in figure 3. For most compound groups it was evident that the spatial distribution of concentrations and the spatial distribution of risks overlapped. However, for phenoxy acids this was not the case. Instead, the results indicated that the guideline values for phenoxy acids were exceeded over a large portion of the site. Hence, the map showing the spatial distribution of phenoxy acid concentrations was not a good indicator of the spatial distribution of risks at the BT kemi site (compare figure 2 and 3).

The reason for the larger risk area of phenoxy acids compared to other compounds was probably its higher water solubility and hence higher propriety for transport from the site to the adjoining surface water, which the risk assessment had shown to be the one of the most important exposure receptors.

These results prompted investigations into uncertainties regarding the spatial distribution of phenoxy acids, their degradation rate and the groundwater transport patterns at the site.



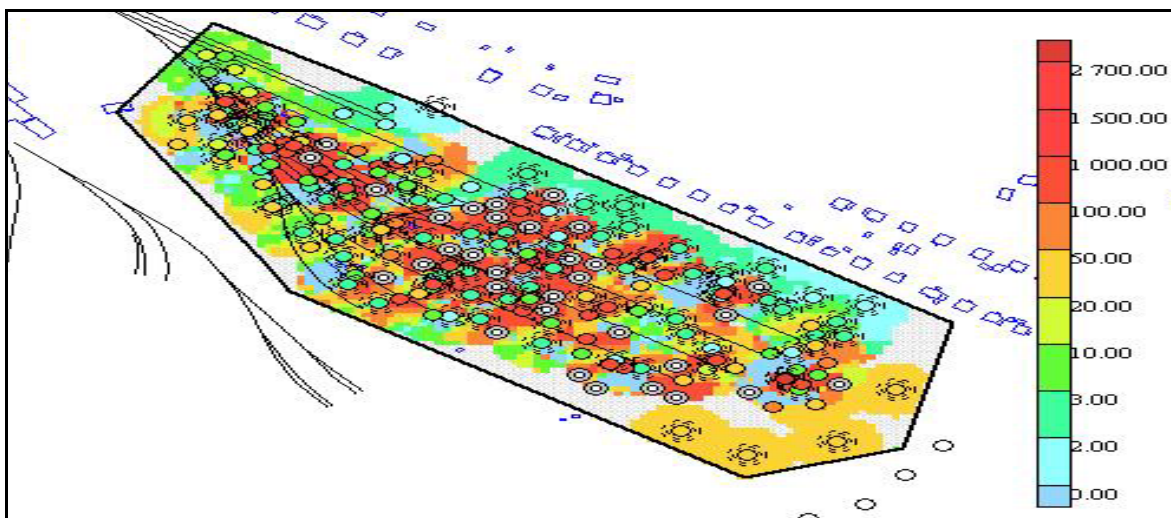
Figur 3. Maps showing the extents of areas where the site specific guideline values were exceeded for chlorophenols and phenoxy acids.

Example 2, Krylbo impregnation site

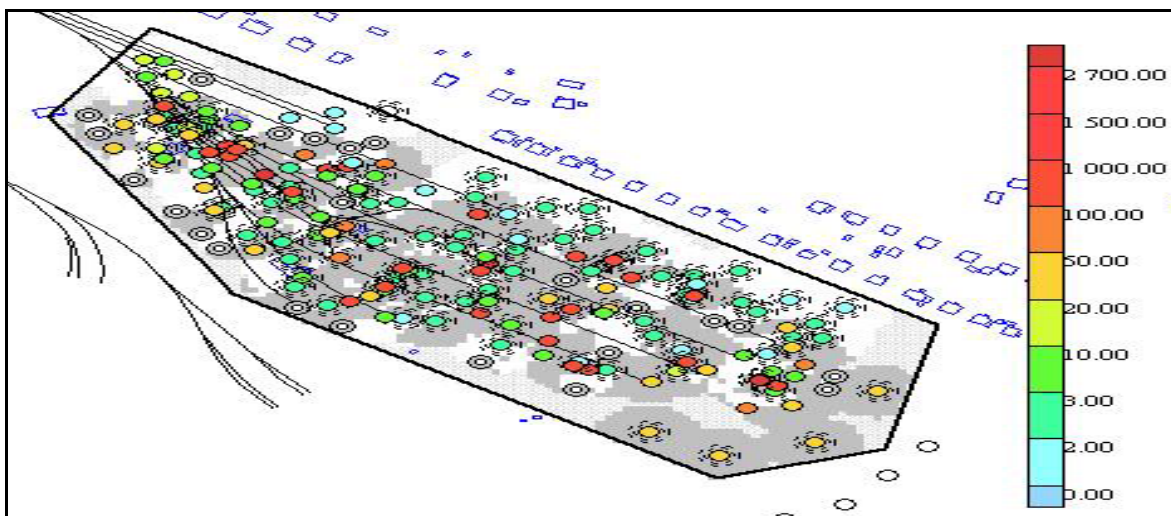
At the Krylbo impregnation site creosote and arsenic has been used to impregnate railroad sleepers for a very long time resulting in high soil concentrations of these compounds. During the on-going site investigation and remediation project, a sampling strategy has been adopted where the position of new sampling points is based on a statistical evaluation of older sampling point.

Figure 3 below demonstrates the position of new sampling points (grey points) based on a strategy that delineates Arsenic hot spots at the site. Figure 4 on the other hand, shows the position of new sampling point (grey points) based on a strategy that delineates the areas where a generic guideline value of 15 mg/kg is exceeded for Arsenic.

It is evident that a strategy based on delineating hot spots vs. delineating risks leads to different placement of new sampling points. The reason is that contaminants hot spots do not equal risk hot spots at the site because the generic guideline value lies well below the median arsenic concentrations at the site.



Figur 1. Measured (colored points) and interpolated (colored fields) arsenic concentrations at the Krylbo site. The grey points indicate positions of new sampling points if the goal is to delineate arsenic hot spots.



Figur 2. Measured (colored points) arsenic concentrations at the Krylbo site. The grey points indicate positions of new sampling points if the goal is to delineate areas where a generic guideline value is exceeded.

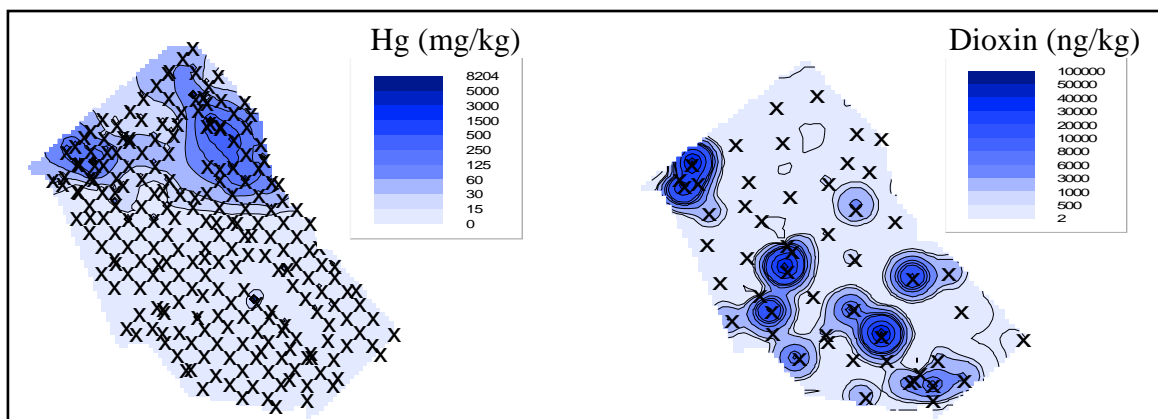
Example 3, Bengtsfors

At the Bengtsfors site, the production of gaseous chlorine for a number of years and the dumping of industrial waste from these industrial activities, has resulted in very high soil concentration of dioxin and mercury. The spatial distribution of mercury and dioxin at this site does not overlap to any large degree (figure 6) which complicates the risk assessment.

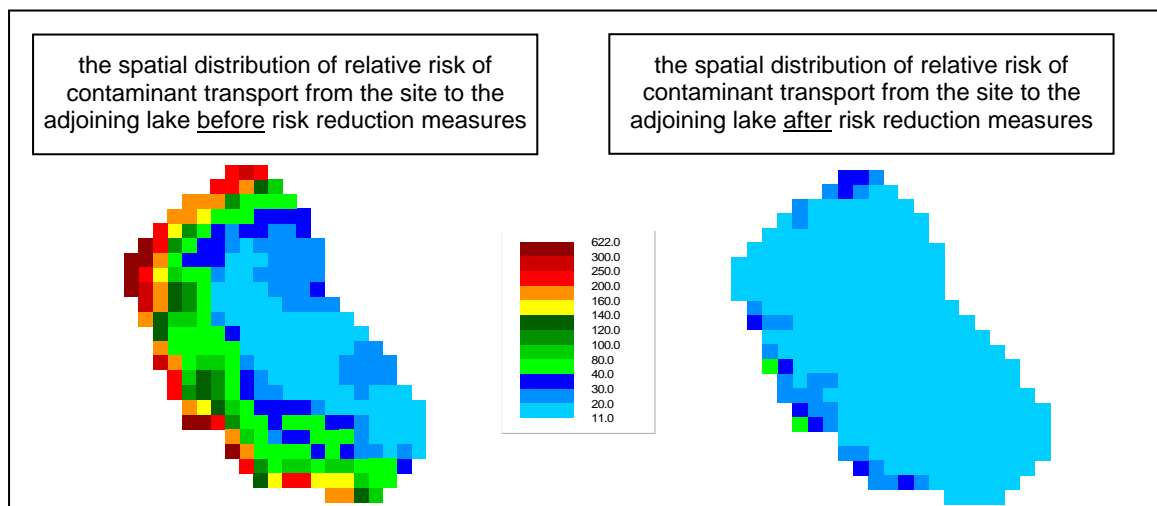
A major risk concern has been the transport of contaminants from the site to surface waters since a lake adjoins the site in two directions. A high hydraulic groundwater gradient further strengthens this concern. In this case it is clear that not only soil concentrations but other parameters such as proximity to the lake, hydraulic conductivity and erosion tendency control the transport risks.

A number of remedial actions have been planned for the site, including excavation, capping and particle filters between the site and the adjoining lake. From a risk assessment point of view the problem has been to; 1) Assess the spatial distribution of transport risks and 2) To connect the risk reduction measures to a reduced risk of transport from the site. A spatial risk reduction analysis was undertaken with the aim of describing the spatial distribution of transport risks and the spatial distribution of risk reduction.

The spatial distribution of transport risks can be viewed in figure 7. It is clear that the spatial distribution of contaminant concentrations and risks differs to a large degree. The reason is mostly that contaminant concentrations are not the most important parameter governing the risk of contaminant transport from the site. If the risk reduction analysis had focused only on soil concentrations the result would have been a totally different view regarding the spatial distribution of risks and the effect of risk reduction measures at the site.



Figur 6. Interpolated mercury and dioxin concentrations at the Bengtsfors site



Figur 7. The spatial distribution of contaminants transport risk from the site to the adjoining lake. The scale is a relative scale based on the model used to assess transport risks.

QUALITY MANAGEMENT OF SITE INVESTIGATIONS

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Abstract

Introduction

All site investigations are associated with uncertainties, and the acceptable uncertainties depend upon the objectives of the investigations and the risk associated with errors. Currently, a number of approaches are being developed to enable quality management aiming at a sufficient but not excessive quality of an investigation ("fitness for purpose"). All approaches include identification of acceptable uncertainty or error, as well as subsequent control of the uncertainty resulting from the investigation. The presentation will introduce a methodology for evaluation of groundwater monitoring uncertainty as well as the use of uncertainty estimates to identify points of improvement with presentation of data from a field study in Århus County, Denmark. The methodology was developed in order to allow planning and evaluation of groundwater monitoring and control aiming at achieving defined quality objectives. The methodology was developed as part of the formulation of an international guideline for estimating uncertainty in sampling, and the basic principles can also be found in the current version of the guidelines for groundwater monitoring prepared to support implementation of the groundwater directive in preparation in the European Commission. The methodology can be used as well in investigations of soil and examples hereof are also given in the abovementioned guideline.

Materials and methods

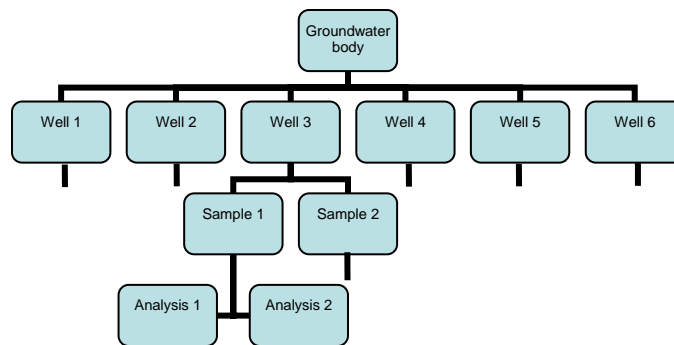
A group of groundwater bodies that are an important drinking water resource for the city of Århus, the second largest city of Denmark, has through surveillance monitoring been identified as being at risk of deteriorating quality due to intensive drinking water abstraction. A monitoring program was established in order to control the development in water quality. The groundwater body is in glacial outwash sands with Miocene sands and clays below and glacial till above. The natural quality of the groundwater is anaerobic without nitrate, with sulphate and reduced iron, but without hydrogen sulphide and methane. One of the threats to the groundwater bodies is oxygen intrusion into the aquifer as the result of the water abstraction and concomitant groundwater table draw down. One groundwater body representing the group, 2 km x 2 km x 10 m, starting 20-30 m below the surface, was selected as the object of monitoring.

During the planning of the monitoring, it was decided to use dissolved iron as a target parameter that would be a sensitive indicator of aquifer oxidation (decreasing of iron content in the water with increasing oxidation). It was further decided to aim at monitoring one well twice per year and the objective of the monitoring was set as a 95% probability of recognising a 20% quality deterioration. This requires a measurement uncertainty including both sampling and analysis of not more than 10% (comparison of two means each for two samples, 95% confidence interval, two sided test) corresponding to an expanded measurement uncertainty of 20%.

To ensure the compliance of the monitoring program with this objective, a sampling validation study was initially conducted including all wells available and based upon the results from this, a routine sampling quality control program was set up for implementation as part of the monitoring program for the selected monitoring well.

An empirical approach was selected in order to provide estimates of heterogeneity in the groundwater body (between-target variation well to well or over time) and measurement uncertainty, split into sampling uncertainty and analytical uncertainty. The basic principle of the empirical approach is to apply replicate measurements.

The objective of the validation study was to ensure that measurement uncertainty meeting the set quality objective could be obtained and to describe the components of uncertainty in order to identify points of improvement, if required. The validation study was set up with sampling of 6 wells, two independent samplings per well and 2 sub-samples per sample analysed, see overleaf figure. The validation study thus included one sampling round with a total of 12 samples taken and 24 sub-samples sent for analysis.

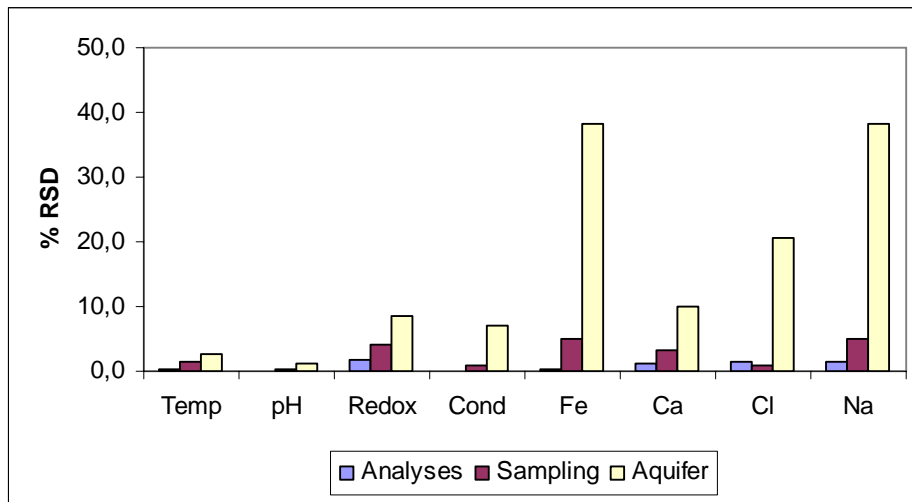


Sampling was done using the groundwater monitoring sampling protocol developed by the county. Analyses were performed at an independent, accredited (ISO 17025) laboratory using accredited methods subject to the required quality assurance and analytical quality control. Estimates of laboratory uncertainty and analytical detection limits were obtained from the laboratory quality control scheme and evaluated with the data from the monitoring validation and quality control.

Results

The replicate data were treated using the range method (ISO 3085), see below table for results. The applied calculation methods are demonstrated in the guide on uncertainty from sampling, calculations are easily done using standard spread sheets, and an example can be downloaded from <http://www.samplersguide.com>. The data treatment provided estimates of analytical, sampling and total measurement uncertainty, in addition to the uncertainty due to heterogeneity (in space or time). Only random errors were included, whereas the occurrence of systematic sampling errors was not assessed quantitatively, but the consistency of the obtained results for different chemical parameters was used as a qualitative control of systematic errors.

The results for on-line and laboratory chemical measurements from the six wells are shown in the figure below.



The objective of the quality control programme for the long-term monitoring was to ensure that measurement uncertainty did not increase over time. The quality control programme was set up after careful evaluation of the results from the validation study and ended up consisting of duplicate sampling on one of the two annual sampling occasions of the monitoring programme.

Dissolved iron in groundwater	Expanded uncertainty, coverage factor 2			Between-target Heterogeneity
	Analysis	Sampling	Measurement	
Validation	2.1%	10%	10%	35% ¹
Quality control			4.0%	9.9% ²

The data show that the requirement of less than 20% expanded measurement uncertainty could be fulfilled for dissolved iron (sampling validation), and that the required measurement uncertainty was in reality achieved during the routine monitoring (sampling quality control). Furthermore, the data show that if an improvement of the certainty of monitoring was required, the obvious point of improvement would be increased monitoring density (with the between-target heterogeneity dominating the overall uncertainty).

Discussion

A tool for quality management of site investigations has been developed that can be used in practical soil and groundwater monitoring and control to quantify the uncertainty (variability, sampling and analysis uncertainties) and thus enable assessment of the confidence or trust possible in the investigation results.

¹ In the validation study, between-target variability was between wells

² In the quality control, between-target variability was between sampling occasions, first 6 sampling occasions included

In planning soil and groundwater monitoring, fitness for purpose (monitoring uncertainty corresponding to set quality objectives) can be ascertained by a simple monitoring validation approach. If required, points of improvement of monitoring can be identified from the contributions to the monitoring uncertainty (analysis, sampling, heterogeneity). With a simple and cost efficient quality control, it can be ascertained that the routine monitoring uncertainty remains as required for the purpose.

Considering the total costs of soil and groundwater monitoring and the costs associated with decisions taken from monitoring data, the costs of including an initial monitoring validation during planning and a subsequent monitoring quality control during routine monitoring seem justified.

The principles applied are described in the Eurachem/EUROLAB/CITAC/Nordtest Guide, Estimation of measurement uncertainty arising from sampling.

References

Eurachem/EUROLAB/CITAC/Nordtest Guide. 2006. Estimation of measurement uncertainty arising from sampling. 6th Committee Draft. UfS_6_1 Apr06.

Grøn, C. 2006. <http://www.samplersguide.com>

UPDATING OF NORWEGIAN SOIL QUALITY GUIDELINES AND INCLUSION OF NEW SUBSTANCES

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Abstract

Aquateam has been engaged of the Norwegian Pollution control Authority (SFT) to update the background material for a new revision of "Guideline for the risk assessment of contaminated sites" (SFT: 99:01). A preliminary proposal of new soil quality guidelines for existing and new substances with sufficient data is performed. Aquatic data is also included in the descriptions, and a comparison of $PNEC_{soil}$ based on terrestrial and aquatic data were carried out.

Introduction

The Norwegian Pollution Control Authority (SFT) published in 1999: *Guidelines on risk assessment of contaminated sites* (SFT: 99:01; Vik *et al.*, 1999). The Guidelines should be updated, since more human and ecotoxicological data now are available, as well as migration and transport data. EU has performed new risk assessments for several substances after 1999. New information of environmental hazards are coming up (e.g. brominated flame retardants, PFOS, alkyl phenols and phthalates), the laboratories have achieved lower detection limits (LOD) for chemical analysis. In addition, aquatic data should be included in the Guideline since leaching from contaminated sites to nearby water recipients happens. Aquateam was in 2004 engaged by SFT to update the background material for a new revision of this Guideline. The proposed updated generic soil quality criteria (SQC) for most sensitive landuse for existing substances are presented in this paper. The Guidelines are not yet decided by the Norwegian authorities (SFT), but are used as a baseline for discussion.

Materials and methods

Ecotoxicological and human toxic data, as well as migration and transport data for the substances included in SFT 99:01 and "new" substances have been gathered, and soil quality Guidelines for substances with sufficient data have been proposed. It has been focused on new challenges in the environmental risk assessment work, e.g. requirement and evaluation of data quality, handling of substances with insufficient or missing data. Aquatic data is also included in the descriptions, and a comparison of $PNEC_{soil}$ derived from terrestrial and aquatic data were carried out.

The procedure for deriving Guideline values is the same as earlier. "Ideal" soil quality Guidelines for most sensitive land-use are derived from the health based (acceptable threshold dose, sum for all pathways) and ecotoxicological based (Predicted no effect concentration, PNEC) soil quality Guidelines. The "ideal" soil quality Guideline for each substance is the lower of the health and ecotoxicological based soil quality Guideline. The ideal soil quality Guideline value is thereafter adjusted for the detection limit (LOD) for the most probable analytical method. If the soil quality Guideline is lower than the reported

Norwegian background values, an assessment is completed. The soil quality Guidelines are then adjusted to what is referred to as the “proposed new soil quality Guidelines”.

Results

The Guideline values have been proposed changed for some parameters (Table 1). Sufficient data are available for some of the “new” substances to suggest new soil quality guidelines (Table 2).

Table 1. Proposed new Guideline values for substances included in SFT 99:01.

Parameter	SQC (mg/kg) SFT 99:01	Proposed new SQC (mg/kg)	Background for changes
Lead (Pb)	63	100	New human health data
Cadmium (Cd)	3	4	New human health data
Chromium (Cr)	25	32	New ecotox data
Nickel (Ni)	50	63	New ecotox data
Benzo(a)pyren	0.1	0.01	New ecotox data
Fluoren	0.6	0.8	New ecotox data
Fluoranthene	0.1	1	New ecotox data
Pyren	0.1	1	New ecotox dat.
Pentachlorophenol	0.005	0.007	New transport data (Kd and H)
Monochlorobenzene	0.5	0.03	New ecotox data
1,2-Dichlorobenzene	0.5	0.1	New LOD
1,4-Dichlorobenzene	0.5	0.02	New LOD
1,2,4-Trichlorobenzene	0.2	0.05	New LOD
1,2,4,5-Tetrachlorobenzene	0.3	0.05	New human health and Kd and H data
Hexachlorobenzene	0.03	0.01	New human health and Kd and H data
Trichloromethane	0.01	0.02	New human health data
Trichloroethylene	0.01	0.2	New Kd and H data
Tetrachloroethylene	0.03	0.01	New ecotox data
Benzen	0.005	0.006	New human health data
Toluen	0.5	0.3	New ecotox data
Ethylbenzen	0.5	0.01	New ecotox data
Xylen	0.5	0.03	New ecotox data
MTBE	2	0.7	New ecotox data
1,2-dichloroethane	0.003	0.1	New human health data

The changes in SQC for most of the parameters are relatively small, but the values are more reliable, since more data (especially ecotoxicological data) have been available since 1999.

Table 2. Proposed Guideline values for substances not included in SFT 99:01.

Parameter	Proposed SQC (mg/kg)	Background
PFOS/PFOA	-	Sufficient data to propose a Guideline value are not available
Brominated flame retardents:		
PentaBDE	0.08	Health data (consumption of vegetables) for PBDE-99
DecaBDE	0.002	Health data (consumption of vegetables) for PBDE-209
HBCDD	-	Not sufficient data available
Tetrabromo bisphenol A	0.001	Ecotox data
Bisfenol A	0.001	Ecotox data
Alkyl phenols:		
Nonyl phenol	0.005	Ecotox data (endocrine effects), adjusted for LOD
Nonyl phenol ethoxylate	0.1	2.5% degrades to nonyl phenol
Octyl phenol	0.005	Ecotox data (endocrine effects), adjusted for LOD
Octyl phenol ethoxylate	0.04	Ecotox data
Organo-tin:		
Tributyltin-oxide (TBTO)	0.001	Ecotox data (PNEC marine Imposex gastropoda 0.07 ng/l), adjusted for LOD
Triphenyltin chloride	0.001	Ecotox data (PNEC marine 0.6 µg/l), adjusted for LOD
Phthalates:		
Di(2-ethylhexyl)phthalate (DEHP)	-	No sufficient data. EU PNEC-soil 13 mg/kg (not based on endocrine effects)

Parameter	Proposed SQC (mg/kg)	Background
Chlorinated paraffins:		
MCCP (Medium chain)	-	No sufficient data. PNEC-soil 1 mg/kg
SCPP (Short chain)	-	No sufficient data. PNEC-soil 0.8 mg/kg
Dioxine	0.00025	Naturvårdsverket's ecotox-value
Polychlorinated naphthalenes (PCN)	0.00025	The same as for dioxine, since the substances have similar characteristics
<u>Tricresyl phosphate</u>	-	PNEC-soil 0.009 mg/kg. Only one laboratory was found to analyse this compound. LOD too high to be used as SQC
Volatile chlorinated hydrocarbons:		
Tetrachloromethane	0.02	Health data (inhalation of vapour)
1,1,2-trichloroethane	0.01	Ecotox data (0.001 mg/kg), adjusted for LOD
Phenols and chlorophenols:		
Sum mono,di,tri,tetra chlorophenols	0.06	Health data for the most toxic component trichlorophenol (inhalation of vapour)
Phenol	0.1	Ecotox data.
Chlorinated benzenes:		
1,2,3-trichlorobenzene	0.01	Ecotox data.
1,3,5- trichlorobenzene	0.01	Ecotox data.

The comparison of $PNEC_{soil}$ derived from terrestrial and aquatic data for substances with sufficient data are shown in Figure 1. $PNEC_{soil}$ from aquatic data: $K_d \bullet PNEC_{water}$. K_d : partition coefficient soil – water (l/kg).

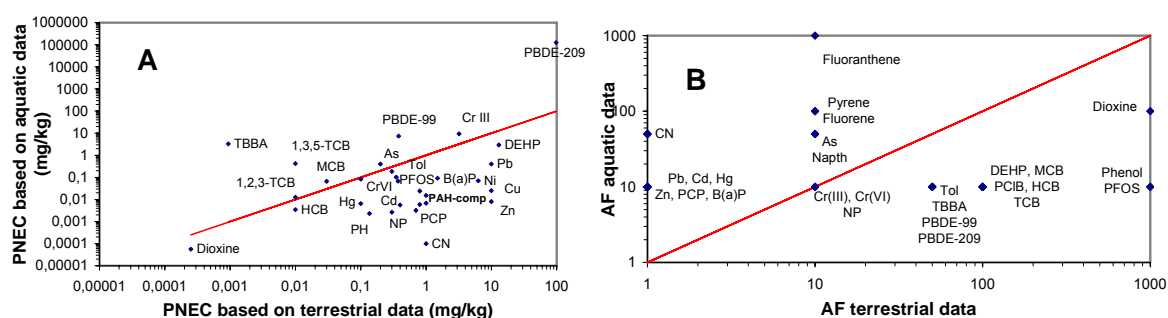


Figure 1: A. PNEC-soil based on aquatic and terrestrial data. The substances below the red line have lower PNEC based on aquatic data than based on terrestrial data. B. Assessment factors (AF) based on aquatic and terrestrial data.

Discussions

Effects taken into consideration

Both ecotoxicological data ($PNEC_{soil}$) and health data should be available as a basis for determining soil quality criteria. Data from EU Risk Assessments (RA) are preferred. In most of the RAs, all relevant effects are evaluated. But in some cases all important effects are not evaluated, e.g. nonyl phenol: The EU risk assessment for nonyl phenol does not include endocrine effects. EU has derived a $PNEC_{soil}$ of 0.3 mg/kg based on NOEC 3.44 mg/kg and an assessment factor of 10. $PNEC_{soil}$ derived from $PNEC_{water}$ would give a $PNEC_{soil}$ of 0.003 mg/kg based on endocrine effects. Results from chronic fish tests (endocrine effect) show NOEC values in the range 0.4-190 µg/l. $PNEC_{water}$ 0.04 µg/l is derived, based on the lowest NOEC value and an assessment factor of 10. A K_d value (partition coefficient soil – water) 66 l/kg (EU, 2002) gives a $PNEC_{soil}$ of 0.0026 mg/kg ($PNEC_{soil} = K_d \bullet PNEC_{water}$). The SQC was derived from this value, but adjusted for LOD to 0.005 mg/l.

PNEC_{soil} derived from PNEC_{water}

When PNEC data from toxicity testing with soil organisms are not available, a PNEC has to be derived from aquatic toxicity data: PNEC_{soil} based on aquatic data seem to be considerably lower than the PNEC_{soil} based on terrestrial data (see Figure 1A). The main reason for the very low PNECs based on aquatic data is probably that the substances are more toxic to aquatic organisms than to terrestrial. Another reason could be that more aquatic data than terrestrial data are available. The PNEC_{water} is often derived from the lowest NOEC value from a range of tests. For terrestrial species, very few tests are normally available, and some of the species are not very sensitive. The PNECs values were derived in accordance with EU's technical guidance documents (TGD) for risk assessment (EU, 1997), and the recommended assessment factors were used. Assessment factors based on aquatic and terrestrial data are shown in Figure 1B. The terrestrial data for the substances above the red line (metals, PAH, cyanide, and pentachlorophenol) are considered as more reliable than data for the substances below the red line (brominated flame retardants, BTEX, phenol, PFOS, chlorinated benzenes and dioxine).

Use of data from degradation products

Degradation products (metabolites) should be taken into consideration if they are more toxic and persistent than the parent compound. An example is nonyl phenol ethoxylate (NPEO). It is not much data available for this substance. The proposed SQC value is derived from the PNEC for nonyl phenol (0.003 mg/kg). According to the EU Risk assessment approximately 2.5% of NPEO biodegrades to Nonyl phenol: SQC for NPEO 0.003 mg/kg • 100%/2.5% ~0.1 mg/kg

Practical use of SQC when assessing site specific conditions

In the further work with the guidelines, it should be focused on:

- Generating alternative acceptance criteria (areal specific quality criteria). The calculation tool gives some flexibility for calculating health risk (some pathways can be left out, some exposure data can be changed), but it gives no flexibility regarding ecotoxicological risk.
- Leakage of contaminants from the site to nearby water recipient, or other spreading of contaminants: Methods for deriving soil quality criteria in relation to environmental conditions in nearby water recipients. The worst case effects have to be taken into consideration. Since some of the substances will leach from contaminated sites to the nearest water recipient, aquatic data should also be considered in risk assessment of contaminated sites.
- The background values in uncontaminated soil: Diffuse pollution, areas with natural high concentrations of metals (Concentrations of As and Cr in uncontaminated ground may be higher than SQC in some areas in Norway)
- Establishment of an interactive web based chemical database (data frequently updated).

References

EU (1996): "Technical Guidance Documents in Support of the Commission Directive 93/67/EEC on Risk Assessment for New Notified Substances and the Commission Regulation (EC) 1488/94 on Risk Assessment for Existing Substances", Part I-IV, Luxembourg.

Vik, E.A., Breedveld, G., Farestveit m.fl. (1999): Veiledning om risikovurdering av forurenset grunn. SFT Veiledning 99:01.

Weideborg, M. og Vik, E.A. (2005): Bakgrunnsmateriale for revisjon av SFT 99:1 (upubliserte data).

Risk-based prioritization of contaminated sites

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Abstract

The Swedish National Rail Administration (SNRA) is managing more than 20 000 properties, of which a large number are expected to be contaminated. To identify potentially hazardous sites SNRA is presently conducting a national inventory. Due to the large number of sites, prioritization of efforts must be made to sites with the highest risk to human health and ecosystems. The SNRA has developed a System for Risk-based Prioritization (SRP), facilitating a three-step prioritization procedure: (1) after the initial inventory of all potentially contaminated sites; (2) after investigations aiming at verification of contaminant sources; and (3) after investigations aimed at spatial delineation of contaminations. Prioritization is in each step based on risk assessments of human health and ecological systems. In each step the risk assessment is updated, taking into consideration the additional knowledge from the investigation. The purpose of the prioritization procedure is to identify sites in most need for detailed investigations, and possibly remedial actions. SPR includes a probabilistic risk assessment method which uses a Bayesian statistical approach for incorporating subjective estimations and managing uncertainties inherent in early stage assessments. As new information becomes available in each step, the risk assessments are updated to a higher degree of certainty. The SPR method includes assessments of the contaminant source conditions, the pathways between the source and the receptors, the contaminant transport conditions along the pathways, and the health and ecological consequences of the receptors. In the final step, an estimation of the expected cost of remediation is included to facilitate a prioritization considering cost-efficiency aspects. This presentation gives a description of the SPR method and shows examples from applications of the method. SPR has so far been applied in three out of five SNRA regions.

Acknowledgements: Lars Grahn and Hans Kronberg, SWECO VIAK, who assisted in the development of the SPR method, and the reference group of the project (Mark Elert, Kemakta Konsult AB; Pär-Erik Back, Geo Innova AB; Per Olsson, County authorities Västra Götaland; and Tommy Hammar, County authorities, Kalmar) that provided constructive advise.

Introduction

The Swedish National Railway Administration (SNRA) expects to have more than 20 000 potentially contaminated sites. An inventory of these sites is currently being performed to identify and these sites. To prioritize further actions among the large number of sites, a well-structured prioritization method is needed. The purpose of the prioritization is to initiate investigations at the sites that exhibit the highest risks to humans and ecological systems. In order to achieve an efficient use of available resources, subsequent remedial actions will be prioritized on risk levels and expected costs for remediation. The philosophical rationale is thus that sites with high risks that can be remediated to low costs should be prioritized in order to achieve the most efficient total risk reduction at the SNRA sites, given available resources.

In order to facilitate a standardized procedure for all SNRA regions and thus making possible a national prioritization of efforts, the System for Risk-based Prioritization (SRP) has been developed. The system comprises of a three-step prioritization procedure: (1) after the initial inventory of all potentially contaminated sites; (2) after investigations aiming at verification of contaminant sources; and (3) after investigations aimed at spatial delineation of contaminations. The prioritization procedure is schematically described in figure 1.

Prioritization is in each step based on risk assessments of human health and ecological systems. In each step the risk assessment is updated, taking into consideration the additional knowledge from the investigation. SPR uses a Bayesian statistical approach for incorporating subjective estimations and managing uncertainties inherent in early stage assessments. As new information becomes available in each step the risk assessments are updated to a higher degree of certainty.

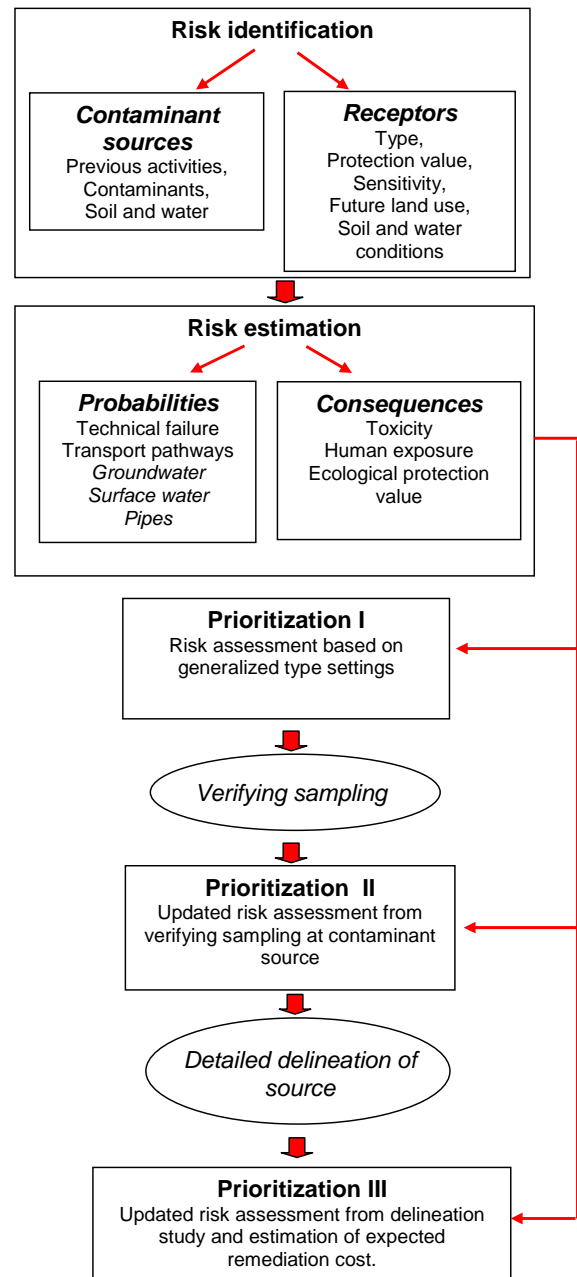


Figure 1. Schematic description of System for Risk-based Prioritization (SRP).

The purpose of this presentation is to (1) give a general description of the conceptual model for risk assessment in SPR, (2) to describe the approach for risk estimations and uncertainty management, and (3) to exemplify the use of SPR.

Conceptual model for risk estimations

The risk estimation in SPR is performed according to the conceptual model described in figure 2. The model was set up in line with common practice for risk assessment of human health and ecological systems.

The potential for release of contaminants is governed by the technical conditions of the contaminant source, e.g. type of construction of the source. The source may or may not have a barrier for reducing the possibilities for transport away from the source area in case of a contaminant release, e.g. collection systems at storage tanks. Contaminants from the contamination source may be transported to a receptor by surface water runoff, by groundwater flow, through piping systems, or in combinations of these transport pathways. There may be different types of pathway barriers, e.g. confining layers in aquifers or hydraulic barriers in piping systems. Finally humans and/or ecological systems may be exposed to the contaminants through several different exposure pathways. Receptor barriers, e.g. monitoring programs at water supplies, may or may not be installed.

The receptors may be the area at the contaminant source as well as external receptor, e.g. aquifers, wetlands, residential areas or schools.

The risk is defined as:

$$R = P_f C_f \quad (1)$$

where P_f is the probability of failure and C_f denotes the consequences of failure. *Failure* is defined as the unacceptable effects resulting from the activity at the *contaminant source*. The consequences of failure are the degree of negative effects with respect to (1) the toxicity for humans, (2) possible pathways for human exposure, (3) residence time of humans at the receptor, (4) toxicity for ecological systems, and (5) the protection value of ecological systems.

Risk estimations

Quantitative probability estimations are made using a Bayesian approach for integrating soft and hard data. The probabilities of the model are: (1) the probability of contaminant release at the source, (2) the probability of non-functioning source barriers, (3) the proba-

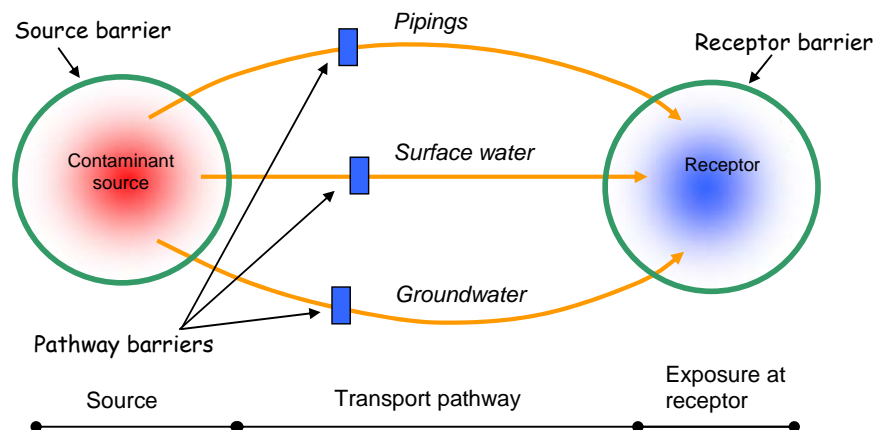


Figure 2. Conceptual description of risk assessment

bility of contaminants remaining in the source area, (4) the probability of transport pathways between the source and the receptor (including considerations of combinations of pathways and pathway barriers), and (5) the probability of non-functioning protection barriers at the receptor. In the initial stage (Prioritization I) the probabilities need to be based primarily on subjective interpretations of information gathered during the inventory. In later steps (Prioritization II and III) sampling is performed and the new information is used for updating the earlier probability estimations. The SPR manual includes guidelines for how to elicit subjective probabilities in the early stages and how to update the probability estimations as new information becomes available. Updating is made using Bayesian updating routines.

Each type of consequences (see above) is estimated using a classification methodology for categorizing the consequences into five different classes. The classification ranges from very low to very high consequences. The SPR manual provides guidelines for how to make the classifications using a matrix system where properties of the classes for the different consequence types are described. Consequences for humans and ecological systems are given equal importance in the risk calculations.

The risk is calculated for all combinations of identified transport pathways and receptors. The user can choose between the maximum risk of all identified combinations or to look at the sum of all risk values from the particular risk source. The risk estimations are displayed graphically in a risk matrix, see figure 3. A graphical display of risks provides important information and insight that cannot be provided by single risk values.

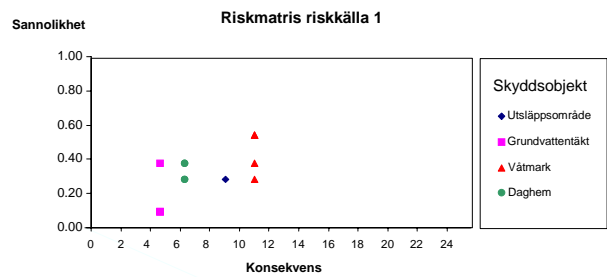


Figure 3. Example of display of risks in a risk matrix ('Sannolikhet' = probability; 'Konsekvens' = consequence).

An important feature of the SPR model is the ability to account for the inherent uncertainties in the estimations of both probabilities and consequences. The principal approach for managing the uncertainties is shown in figure 4.

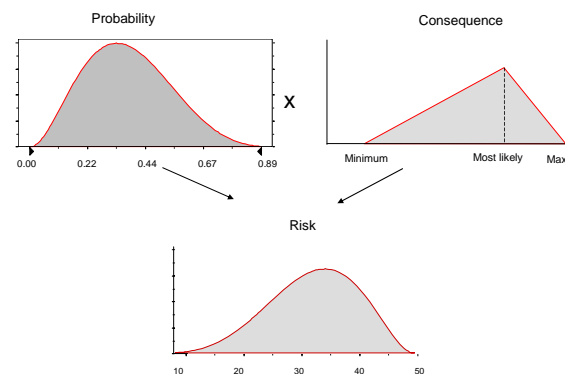


Figure 4. Schematic description of the approach for uncertainty management in SPR.

The uncertainties of the probability estimations are represented by a beta-distribution and the uncertainties of the consequences are estimated by triangular or binomial distributions. The resulting uncertainty of the risk is calculated by statistical simulation (Monte Carlo). The method allows for sensitivity analysis of the risk assessment to identify the most uncertain variables in the risk calculation. This information is then used in selecting variables most relevant for further studies in the subsequent steps of the SPR model.

A SPR calculation tool was developed to assist in the risk assessments. The SPR tool is an Excel spreadsheet model that includes all the guiding matrices, calculation tools, Monte Carlo simulation and manual for using the method.

Prioritization

The prioritization of investigations and further actions can be made in SPR using two different approaches. The first approach is to prioritize based on the maximum risk value of all combinations of transport pathways and receptors that is subject to risk from the specific contaminant source. This approach does not make any difference whether single or several transport pathways or receptors are present; only the maximum risk value is considered. The other approach is to look at the sum of all risk values for the contaminant source. This approach makes sites with many pathways and receptors receive higher risk values than sites with few pathways and receptors. Thus, it may be the case that a contaminant source with many receptors where each receptor is associated with a low to moderate risk may be prioritized over a contaminant source with only one receptor being exposed to a very high risk. Both approaches have favorable and unfavorable properties; it is the decision-maker that has the responsibility to decide what approach to choose.

In the final step (Prioritization III), the risk value is integrated with the expected cost of remediation in order to provide a prioritization basis that considers the cost-efficiency of remedial actions. For the approach of using the sum of all risks as a prioritization basis, a prioritization index is calculated as:

$$\Phi_{tot} = \frac{R_{tot}}{E_{tot}} \quad (2)$$

where R_{tot} is the sum of all risks from the contamination source and E_{tot} is the expected costs for remediation (expressed in MSEK; 9,30 SEK \approx 1€).

For the approach of using the maximum risk as a prioritization basis, the index is calculated as:

$$\Phi_{max} = \frac{R_{max}}{E_{tot}} \quad (3)$$

where R_{max} is the maximum risk among all combinations of transport pathways and receptors being at risk from the contamination source. The higher the index value, the higher the expected risk reduction per cost unit.

The expected costs for remediation are estimated using a costing model included in the SPR tool. The model assumes that remediation will be made by excavation and disposal of contaminated soil. The model considers explicitly the expected volume of contaminated soil, the contaminant concentrations, the transport distances to the disposal site, and the costs for disposal. In addition, there are a number of implicit factors that cannot be adjusted by the user, regarding e.g. weather conditions, types of excavation machines, and monitoring.

Applications

The SPR method is currently being applied in three out of five SNRA regions: The method has the status of 'Standard', which means that application is mandatory. So far, only the first step (Prioritization I) has been applied. Approximately 500 sites have so far been evaluated using the methodology. To make sure that estimations are made in similar ways between different users and in different regions, instructions as well as seminars on the applications are regularly performed.

ASSESSMENT OF HUMAN EXPOSURE TO PCDD/F AT CONTAMINATED WOOD IMPREGNATION SITES COMPARED TO BACKGROUND EXPOSURE

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Abstract

The multimedia fate- and exposure model CalTOX was used to model background concentrations in environmental media for PCDD/F in a generic landscape. The modelled concentrations were in good agreement with measured data, both for air, soil and water. A contaminated soil scenario generated the same or slightly elevated air concentrations compared to background scenario. The exposure module of CalTOX overestimated the exposure from food consumption by a factor of 100. Assuming that food consumption may contribute to at least 95 % of the total dioxin exposure, the modelled exposure from direct pathways were underestimated by a factor of 10, but still in quite good agreement with the Swedish food intake data. Further calculations are carried out to be able to compare exposure from contaminated sites to background exposure.

Introduction

It is generally agreed that most of the human exposure to PCDD/F is due to food intake. However, local point sources, such as contaminated wood impregnation sites, have been proved to contribute to exposure at a local scale (Dahlgren et al., 2003). Increased levels of PCDD/Fs in environmental media close to point sources may affect human exposure through several pathways, such as production and intake of locally produced food, inhalation of vapours and particles, dermal contact and ingestion of soil.

Usually, site specific investigations of contaminated sites in Sweden focus on environmental media concentrations in soil, sediment and water. The costs for PCDD/F analyses usually limits the possibilities to thoroughly investigate the fate and exposure at such sites. To improve our understanding of the extent to which PCDD/F contaminated soil can contribute to increments in human background exposure, we therefore used a multimedia fate- and exposure-modelling approach. Population defined scenarios at different environmental concentrations allows a quantitative comparison of exposure by several pathways.

Materials and Methods

The multimedia fate and exposure model CalTOX (McKone, 1993) was parameterized to represent the southern Swedish climate. Degradation rates appropriate to the range of temperatures in southern Sweden were taken from the study by Sinkkonen and Paasivirta (2000). Physico-chemical properties of selected PCDD/F congeners were mainly taken from Mackay et al. (1992). The congeners examined in this study were 2,3,7,8-TCDD, 1,2,3,4,7,8-HxCDD, 1,2,3,4,6,7,8-HpCDD, OCDD, 2,3,4,7,8-PeCDF and 1,2,3,4,6,7,8-HpCDF. The congeners TCDD and OCDD were chosen since they represent a broad interval of physico-chemical properties and thus the expected behaviour of PCDD/Fs in the environment. The congeners HxCDD, HpCDD and HpCDF represent congeners which usually contribute significantly to the total TEQ-value at chlorophenol contaminated sites. PeCDF is commonly found in food products.

A set of four exposure scenarios (Figure 1) were defined which allow a comparison of exposure at different source strengths as well as for different population behaviour.

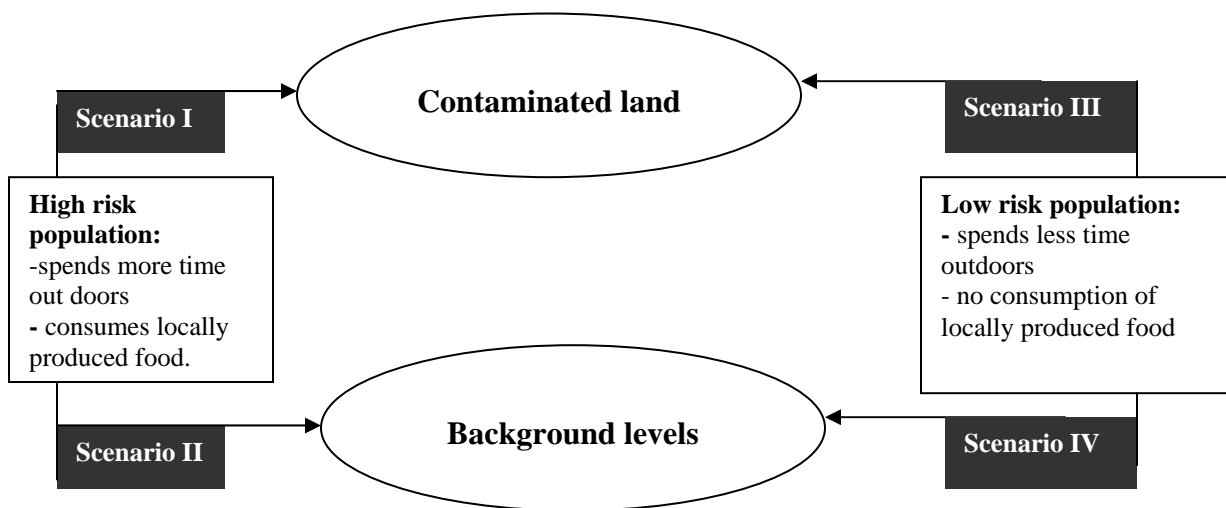


Figure 1. A schematic representation of the two populations at different source strengths combined to four modelling scenarios.

Food consumption characteristics of both populations were as far as possible estimated from a Swedish study of intake of PCDD/F from food products (Lind et al., 2002). A survey of contaminated wood impregnation site investigations was done to estimate the soil concentrations of PCDD/F in the contaminated land scenarios (I and III). The degree of contamination varied widely between different sites and different congeners. The contamination levels in the exposure scenarios were therefore set near the high end of this realistic range of soil concentrations. A continuous emission to air was used to model the background concentrations.

Results

The environmental media concentrations used in the modelling, together with measured background concentrations from Tysklind et al. (1993) and Matscheko et al. (2002), are shown in Table 1. Compared to measured background concentrations, the model estimates environmental background concentrations relatively well. Soil concentrations were underestimated by a factor of 10, while the air concentrations were nearly the same as measured concentrations. Surface water concentrations (not shown) were the same or slightly underestimated compared to measured values. The contaminated soil scenario generated the same or slightly elevated air concentrations compared to background scenario. Surface and groundwater concentrations were elevated with a factor of 100 or more, compared to measured values.

Table 1. Modelled and measured environmental concentrations (*nd* = not detected, *dw*= dry weight).

Environmental concentrations	2,3,7,8-TCDD	1,2,3,6,7,8-HxCDD	1,2,3,4,6,7,8-HpCDD	OCDD	2,3,4,7,8-PeCDF	1,2,3,4,6,7,8-HpCDF
Contaminated soil, modelled, (<i>pg/g d w</i>)	20	20 000	200 000	200 000	200	200 000
Air in contaminated soil scenarios, modelled, (<i>pg/m³</i>)	0.001	0.07	0.6	0.6	0.0007	0.8
Background soil, modelled/measured, (<i>pg/g d w</i>)	0.02/ <i>nd</i>	0.03/ 0.23	0.3/2.8	3.8/15	0.03/ 0.45	0.3/3.1
Background air, modelled/measured, (<i>pg/m³</i>)	0.004/ 0.002	0.004/ 0.002	0.05/0.03	0.5/0.24	0.004/ 0.002	0.05/0.01

Exposure calculations for high and low risk adults at background concentrations are summarised in Table 2, together with food intake exposure from Swedish data.

Table 2. Modelled total exposure at background concentrations for adults in the high and low risk population. Both scenarios include 24 h of inhalation. Intake from a Swedish food consumption survey, including 17 congeners, is also reported (Lind et al., 2002) .

	^a Direct exposure	^b Indirect exposure	Total
	pg WHO-TEQ/kg bw/day	pg WHO-TEQ/kg bw/day	pg WHO-TEQ/kg bw/day
High risk adults	0.006	630	630
Low risk adults	0.001	-	0.001
Intake from food consumption (mean)	-	-	0.05-12.3 (0.8)

^a Inhalation, consumption of water, soil ingestion and dermal contact

^b Consumption of fish, meat, milk, egg and vegetables.

Discussion

The fate and transport module of CalTOX seems to be an efficient tool to model environmental concentrations due to the relatively good agreement between modelled and measured environmental concentrations for the background scenario. By modelling a generic contaminated site we have tried to approach a "worst-case" scenario. Since highly variable and site specific landscape parameters, such as inflow of surface water, were generalised in the generic landscape, the influence of such parameters have to be investigated with a sensitivity analysis.

For the validation of the exposure calculation, the food consumption for the high risk adults at background concentrations were the same as in the Swedish investigation. It is therefore obvious that the model overestimates the exposure from food consumption by a factor of 100. This could be due to too high biotransfer factors, which were estimated from Chiao et al., (1994). When modelling the contaminated site scenario, only a fraction of the total food consumption is assumed to be locally produced. Assuming that food consumption may contribute to at least 95 % of the total dioxin exposure, the modelled exposure from direct pathways are in quite good agreement with the Swedish food intake data. The mean food intake in the Swedish investigation were 0.8 pg TEQ/kg bw/day, which then corresponds to 0.04 pg TEQ/kg bw/day from direct exposure pathways. In that case, the modelled direct exposure was then underestimated with a factor of 10. Further calculations are carried out to be able to compare exposure from contaminated sites to background exposure.

References

- Dahlgren J, Warshaw R, Horsak R D, Parker FM, Takhar H. 2003. Exposure assessment of residents living near a wood treatment plant. *Environ Res.* 92:99-109.
- McKone TE. 1993. CalTOX: A Multimedia Total Exposure Model for Hazardous Waste Sites, Part I-III. Department of toxic substances control, Lawrence Livermore National laboratory, Livermore.
- Sinkkonen S, Paasivirta J. 2000. Degradation half-life times of PCDDs, PCDFs and PCBs for environmental fate modeling. *Chemosphere.* 40:943-949.
- Mackay D, Shiu W-Y, Ma KC. 1992. *Illustrated handbook of physical-chemical properties and environmental fate for organic chemicals. Volume II. Polynuclear aromatic hydrocarbons, polychlorinated dioxins and dibenzofurans*, LEWIS Publishers, Michigan, USA.
- Lind Y, Darnerud PO, Aune M, Becker W. 2002. *Exponering för organiska miljökontaminanter via livsmedel*. SLV Report 26, National Food Administration, Uppsala, Sweden.
- Tysklind M, Fångmark I, Marklund S, Lindskog A, Thaning L, and Rappe C. 1993. *Environ Sci Tech.* 27:2190-2197.
- Matscheko N, Tysklind M, de Wit C, Bergek S, Andersson R, Sellstrom U. 2002. *Environ Toxicol Chem.* 21:2515-2525.
- Chiao FF, Currie RC, McKone TE. *Intermedia transfer factors for contaminants found at hazardous waste sites. 2,3,7,8-Tetrachlorodibenzo-p-dioxin (TCDD). Final draft report*. The Office of Scientific Affairs, The Department of Toxic Substances Control, The California Environmental Protection Agency.

WHAT DO YOU MEAN BY BIOAVAILABILITY?

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Abstract

The numerous methods for evaluating sediment contamination arise, in part, from the difficulty in separating anthropogenic and natural influences, which is further complicated by interdependencies between sediment parameters and highly variable site conditions. A comparison of alternative methods suggests that a system perspective is necessary, but that the simple combination of multi-parameter results is insufficient. Methodological improvements are possible by considering the sedimentologic conditions, which in turn strongly influence the geochemical and ecological character. The Functional Facies concept utilizes this physical perspective to make sediment sampling and classification of pollution levels more sensitive to bio-geo processes, sediment-parameter relationships and site-specific conditions.

Introduction

Sediment quality is ideally based on the sum of ecological risks that result from contamination or other disturbances. In practice these risks can seldom be measured directly, and various criteria, chemical and biological, are used as proxies in this evaluation. An increasing variety of sediment-quality measures complicates their comparison and overall value. The central hypothesis below is that this diversity can, nevertheless, be beneficial if sediment quality criteria are related to the in situ sediment conditions and the functional relationships of sediment components, similar to the facies concept used in geological descriptions. In this context, "bioavailability" involves physical, geochemical and biological factors, which are strongly integrated and not easily reduced to a single-parameter measurement.

Comparison of evaluation methods

Evaluation methods for sediment contamination can be compared regarding their ability to account for natural geochemical variability, thereby allowing comparisons between sites and sampling occasions (Fig. 1). Total bulk sediment chemistry (TC) is the most frequently used criterion, but also early recognized as misleading since it does not account for parameter interdependencies, such as between metal contents and grain size, organic matter content and oxide precipitants. Compensation for parameter dependency can be attempted by normalization if relationships can be empirically defined (Loring and Rantala 1992). Multi-parameter (MPN) normalization is less common, but offers greater sensitivity to the presumably complex associations in natural sediments than does single-parameter (SPN) normalization (Spencer et al. 1968; Engström et al. 1999). Since normalization is related to associations defined by population statistics, the site specificity is low (Fig. 1).

A more direct geochemical approach to identifying the sediment components associated with element concentrations is to leach specific fractions (LFC) using increasing aggressive reactants (Tessier et al. 1979). This "speciation" of the carrier sites (e.g. exchangeable sites, carbonate phases, oxides, sulfides and organic matter) is, nevertheless, only operationally

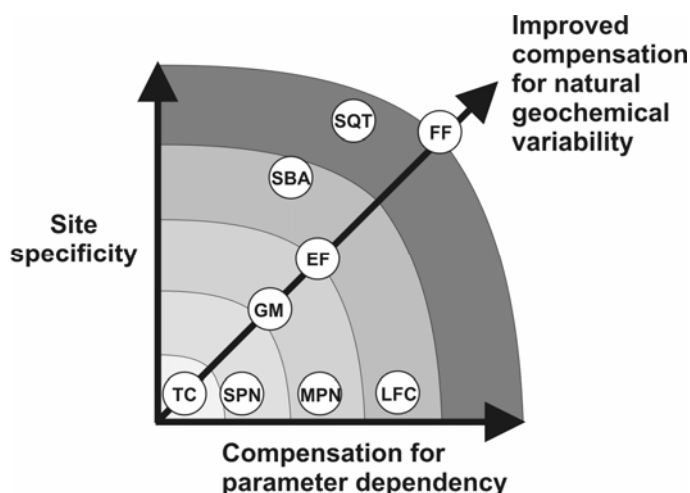


Fig. 1. Common evaluation methods for sediment contamination. The relative ability to account for natural geochemical variability (the shaded bands) is largely dependent upon how well these approaches compensate for parameter interdependencies and provide site specificity. The method abbreviations are given in the text.

defined and requires interpretation regarding sediment geochemistry for each site, as discussed later. An improvement in site specificity is obtained using the Gradient Method (GM; Cato 1989), where contaminant concentrations are plotted against a carrier phase (usually organic matter or clay content). Individual sites can be compared with the trend of the local or regional dataset. Also, if the contaminant variations are not fully explained, the intercept of the regression line on the axes can suggest the degree of parameter dependency, and the need for considering a multi-parameter approach. Enrichment factors (EF) have good site specificity, especially if the background values are taken from pre-industrial levels at the same locality and with similar sedimentologic conditions. National or global values should be used with prudence since grain size, mineralogical composition and other parameter differences are likely to be more complex than to allow direct comparisons. Bioassays (SBA) test toxicity for selected organisms using natural sediments or pore water. In similarity to the use of leached fractions, this method does not require the identification of sediment parameter relationships. However, the site specificity is somewhat lowered by the unavoidable differences in laboratory conditions compared to the investigated site, and parameter dependencies may be important if separate sites are to be compared.

The Sediment Quality Triad (SQT) introduces the site-specific documentation of the benthic ecosystem and combines analyses of the geochemistry, toxicity and benthos to assess relative sediment quality (Chapman 1990). Using synoptic data collected in the same area at the same time and evaluated together, this method recognizes the value of multiple perspectives and the integral relationships between the sediment components. However, the combination of data from the different test to produce a single numerical index is a simplification that is not recommended other than for screening. Rather, multiple indices for each component of the SQT are favourably summarized and the collective data set is used in the final analysis, where also the interdependencies between sediment parameters can be better accounted for. SQT combines measures of contamination and its effects, offering a more complete characterisation and a high degree of site-specificity, especially when using *in situ* evaluation of ecologic structure. However, parameter variability and site specificity are related to dynamic processes, including physical sedimentation and reworking. Further improvements are therefore possible through a “system” perspective and classification of the bottom environments from a sedimentologic basis, as discussed below.

Methodological improvement: Functional facies

Functional Facies (FF) are defined here as bottom-environment classes that integrate physical, geochemical and biological relationships. The facies concept is a traditional tool within ecological and geological sciences, and in sedimentology the specific characteristics

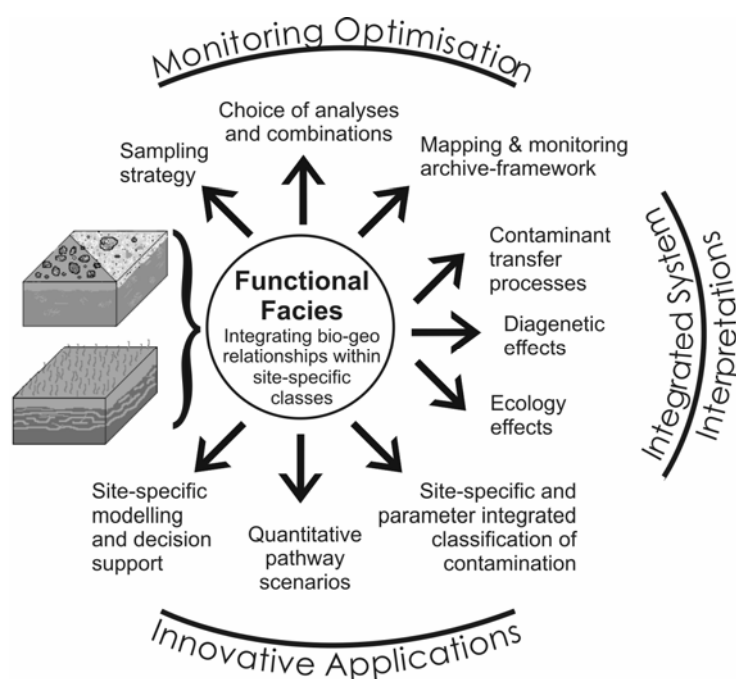


Fig. 2. The Functional Facies (FF) concept and its use in monitoring, system interpretations and possible applications.

of each bottom-environment class reflect the sedimentation, diagenesis and ecological conditions and processes in three dimensions, and also the integrated effects over time. FF can be used to optimise monitoring strategies and to assist process interpretations of the dynamic environment (Fig. 2). This system perspective of FF can also be used to improve the validity of the empirical parameter relationships within each FF class, and thereby maximize the applicability of assessment methods such as SPN, MPN, GM and SQT.

The FF classes should include the parameter observations made during sampling, but should also be able to incorporate subsequent detail from laboratory analyses. Parameter observations that are often accessible during sampling include the presence of living macrofauna or fauna remains, bioturbation, macroscopic plant remains and anthropogenic material, as well as internal sediment structure, surface texture and relief (e.g. smooth surface or lumps of clay), grain-size estimates and depth of oxidation or other geochemical layering. Digital core and sample images are now feasible as routine monitoring documentation, and should facilitate future interpretations with improvements in theory or staff expertise. The choice of sediment analyses can also be guided by the site-specific sediment character of FF units (Fig. 2). The presence of fauna in the sediment suggests, for instance, that geochemical analyses should be combined with SBA. The *in situ* biological effects, such as exposure to bioirrigating and bioturbating organisms, can be difficult to simulate in the laboratory. This is an important limitation on the evaluation of SBA results, but can be partially accounted for within a FF scheme where analogous settings can support these process interpretations.

Classification of FF in the Göteborg harbor resulted in seven divisions based upon the characteristic bottom surfaces and vertical variability of sediment appearance (Stevens and others 1999). The main two classes are smooth sediment surfaces with abundant worm tubes (67 of 122 sites) and smooth surfaces without macrofauna. Both of these FF include coarse-grained bottom environments in the river area and fine-grained environments in the harbor. The extensive sediment surfaces covered by worm-tubes or organic mats indicate limited turbulent re-suspension and sediment reworking (Johannesson et al. 2003), which are otherwise common processes in harbors. A detailed description of the geochemical zonation in the sediment together with leached fraction content (LFC) gave a good basis for interpreting diagenetic effects (cf. Johannesson et al. 2003). The macrofauna reflect the site specific ecology effects, but also the effect of the fauna itself upon the sediment character

can be interpreted using FF. Although they have different grain size properties, the relationships between sediment parameters are believed to be importantly influenced by the processes induced by the bottom fauna: bioturbation, bioirrigation, filtration and baffling. The contents of clay, Fe and Zn are particularly well correlated within each of these FF classes. The system perspective provides further support for interpreting the parameter trends in each of the FF subsets.

Discussion

An increasing interest for bioassay testing emphasizes the need for site-specific relevance when evaluating sediment pollution. Although “bioavailability” is an attractive term, focusing only upon the net effects of contamination begs the question of how biological, geochemical and sedimentological processes interrelate. “Bioavailability” needs to be defined within the dynamic framework of the bottom environment, which is largely dependent upon background sedimentological conditions and sediments. It is concluded that the environmental relationships relevant for characterizing bioavailability can be integrated using the facies concept. The FF concept is suggested to optimise site specificity and limit the masking effect of parameter interdependencies, allowing for more realistic comparisons between sites and better predictions of the environmental responses to change. The connection of this classification to an open database structure should also be favorable for techniques using stochastic or deterministic modelling. Simple geochemical threshold levels or enrichment factors should be considered suspect until comprehensive and site-specific evaluations have established the relationships necessary to justify these widely applied classifications.

References

- Cato, I. 1989. Normalization of heavy metal data by the gradient method. ICES CM. 1989/E2: 88-108.
- Chapman, P.M. 1990. The Sediment Quality Triad approach to determine pollution induced degradation. *Science of the Total Environment*. 97/98: 815-825.
- Engström, L.T., and Stevens, R.L. 1999. Multiparameter normalisation of geochemical data from Göteborg Harbour. 4th International Conference on Characterisation and Treatment of Sediments, Antwerpen. 159-168.
- Johannesson, L.T., Stevens, R.L., and Eriksson, K. 2003. The influence of an urban stream on sediment geochemistry in Göteborg Harbour, Sweden. *Environ. Geology*. 43: 434-444.
- Loring, D.H., and Rantala, R.T.T. 1992. Manual for the geochemical analyses of marine sediments and suspended particulate matter. *Earth-Science Reviews*. 32: 235-283.
- Spencer, D.W., Degens, E.T., and Kulbicki, G. 1968. Factors affecting element distribution in sediments. In Ahrens, L.H. *Symposium on the Origin and Distribution of the Elements*. Pergamon Press, New York. 981-999.
- Stevens, R.L., Engström, L.T., Brack, K., and Book, S. 1999. Harbour sediment documentation for undefined, future purposes. 4th International Conference on Characterisation and Treatment of Sediments, Antwerpen. 551-559.
- Tessier, A., Campbell, P.G.C., and Bisson, M. 1979. Sequential extraction procedure for the speciation of particulate trace metals. *Analytical Chemistry*. 51: 844-851.