Environmental assessment of secondary construction materials

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LICENTIATE THESIS

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PREFACE

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ABSTRACT

Process industry, construction and other comparable activities produce large quantities of waste with potential use in geotechnical applications. Prior to utilisation, an acceptably low risk of contamination to humans and the environment must be demonstrated. This work focuses on the identification and evaluation of critical factors for environmental assessments of secondary construction materials.

The market potential and the main barriers for usage of industrial wastes were analysed and showed a good potential especially in urban areas. The main obstacle is the long and complicated permit process involved. Further, the lack of a general procedure to investigate the suitability of intended usage leads to inconsistent assessments.

An evaluation of leachate emissions from a large-scale test road demonstrated the importance of construction design and site-specific field conditions on the potential environmental impacts. It was also shown that pollutant concentrations in leachate from secondary construction materials tend to become comparable, or for some pollutants, even lower than from rock materials.

Different assessment methods and criteria to judge the acceptability of an intended use were reviewed and various methods were identified. However, a generic method to evaluate materials under various environmental conditions is lacking.
SAMMANFATTNING

Processindustrier, byggbranschen och andra liknande verksamheter producerar stora mängder avfall som skulle kunna användas i geotekniska konstruktioner. Innan användning kan ske måste man dock visa att detta inte medför en oacceptabel risk för människor och miljö. Den här avhandlingen fokuserar på identifiering och utvärdering av kritiska faktorer för miljöbedömningar av sekundära konstruktionsmaterial.

Marknadspotentialen och de huvudsakliga hindren för användning av industriella avfall analyserades och visade på en god potential i urbana områden. Det mest betydelsefulla hindret är den långa och komplicerade tillståndsprocess som en tänkt användning av sådana material medför. Vidare leder avsaknaden av ett generellt förfarande för att undersöka lämpligheten av en tänkt användning till inkonsekventa bedömningar.

En utvärdering av lakvattenemissioner från en fullskalig provväg visade på betydelsen av konstruktionens design och de placesspecifika miljöförhållandena för möjlig miljöpåverkan. Det visades också att föroreningshalterna i lakvatten från sekundära konstruktionsmaterial tenderar att bli jämförbara eller, för några föroreningar, till och med lägre än från bergmaterial.

Olika bedömningsmetoder och kriterier för att bedöma godtagbarheten av en tänkt användning undersöktes översiktligt och olika metoder har identifierats. Dock saknas en generell metod för att utvärdera materialen under olika betingelser i miljön.
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1 INTRODUCTION

Process industry, construction and other comparable activities produce large quantities of wastes. In Sweden, an estimated 60 million tonnes of mineral industrial residues is generated annually (SGI 2003). In addition, hundreds of million tonnes of such materials are available in landfills.

Several driving forces to increase the reuse and recycling of wastes have been introduced recently. For example, alternative costs, e.g. for landfilling, have increased due to taxes raised on waste deposited on landfills, while the expansion and maintenance of road and railroad constructions consumes large amounts of aggregates. In densely populated areas the depletion of conventional materials, i.e. sand, natural gravel and crushed rock, and the need for resource conservation has lengthened transport distances. In addition, the implementation of the EU landfill directive (Council Directive 1999/31/EC) has increased the demand for landfill cover materials, with many sites lacking natural materials for the fitted purpose. There is thus a need to introduce alternative construction materials on the market.

Several wastes are technically feasible for use in construction, e.g. granular slags from ore processing, and incineration residues (Rogbeck and Hartlén 1996; Rogbeck and Knutz 1996; Hill et al. 2001; Arm 2003). However, it is necessary to demonstrate that an acceptably low risk of contamination to humans and the environment would result from the utilisation. Uncertainty about potential risks may limit their use and also lead to inappropriate usage causing adverse environmental effects.

The following questions are discussed:
- What are the market potential and the main barriers for use of secondary construction materials?
- Which factors should be considered in environmental assessments of such materials?
- Which methods can be used to assess the suitability of an intended use?
2 DISCUSSION

2.1 Market potential and barriers for use

The application of industrial waste in earth construction is not a recent development. The Romans were already using brick rubble and slag from forges in road-building. In Sweden, waste from works and mines have traditionally been utilised locally.

About 30 million tonnes of secondary materials with potential use in geotechnical constructions are generated annually in Sweden (Lidelöw and Lagerkvist 2004). The generation of such materials is compared with the production of conventional aggregates in Figure 1. In 2002, the total annual deliveries of aggregates amounted to 71.3 million tonnes of which about 57% were used within the road sector (SGU 2003).

Figure 1 Annual generation of aggregates in Sweden (Lidelöw and Lagerkvist 2004).

Mining generates the majority of potential alternative materials. Hundreds of million tonnes of mine waste rock suitable for construction purposes are also available at landfills (Johansson 1997; Vägverket 1999). Mine waste generally arises in remote areas where the demand for construction materials is low. Due to large transport costs, only a small fraction of the available amounts are recycled. Transporting the materials by boat or train from northern Sweden to the huge European market is however one alternative that is already somewhat working (Boverket 1998).

Excavated materials as well as construction and demolition waste arise at point sources scattered all over the country. The amounts produced vary and are not well known (Arell 1997). Large-scale recycling of such materials would require facilities for intermediate storage and pre-processing. Today, the materials are primarily deposited or
partly reused close to the production sources. Conversely, capping layer materials such as asphalt are often reused in e.g. new roads close to the sources.

Larger metal industries and incineration plants are often located close to larger urban centres. The amounts of waste produced from such facilities are reasonably constant and relatively well documented. Metal industry waste, including steel slag, blast furnace slag, ferrochrome slag, iron sand, and foundry sand, is reused to various extents. Blast furnace slag, iron sand and ferrochrome slag are regularly used in road or railroad constructions in areas around the production sites. Blast furnace slag has, for example, been used for more than 30 years and is regarded as a conventional material on the local market (Lidelöw et al. 2003). However, steel slags are used to a lesser extent. These materials are technically beneficial as construction materials, but environmentally suitable only for certain applications (Vägverket 1999; SGI 2003). Although having a great potential for recycling, foundry sand is mainly landfilled (Nayström 2002). The amount of incineration ashes produced annually is about 1 million tonnes, and includes ashes from municipal solid waste incineration (MSWI), as well as from the pulp and paper, energy, and wood based industries. According to Bjurström et al. (2003), about 25% of ashes are recycled. Increasing amounts of MSWI bottom ash are for instance used as filling material or sub-base material in roads. The amount of ashes generated is expected to increase as a result of current management strategy favouring incineration of municipal solid wastes over landfilling.

Waste can be classified as hazardous or non-hazardous. Hazardous waste is regarded as unacceptable for use in construction. According to Adler et al. (2004), the classification should be based on the total content analyses of selected reference substances, e.g. lead, in the waste. MSWI bottom ash, for example, may thus be classified as hazardous waste, although it is shown to generate leachates with low lead concentrations in the field (Lidelöw and Lagerkvist, in manuscript).

According to legislation, an environmental permit is required for each considered use of a waste material unless the potential environmental impact may be regarded as minor. The permitting authority is the county administrative board. If the environmental impact is judged to become minor, a notification is sufficient (i.e. permit is not required) (SFS 1998:808, 9 chap. §1). However, supervisory authorities have different interpretations of the concept of minor and national guidelines are lacking (Vägverket 1999; Klingberg 2002; SGF 2003). Consequently, approval practices and permit procedures vary in different regions. There are two main types of procedures (Figure 2): (a) no approval is required and the material is regarded as acceptable based on earlier experience; and (b) an environmental impact assessment (EIA) is required to demonstrate that its use will not pose any adverse effect on human health or the environment. The first procedure applies to the use of e.g. blast furnace slag and iron sand, which is accepted in some regions as long as certain conditions set up by the authorities are followed (Borell and Peterson 2001; Lidelöw et al. 2003). The use of blast furnace slag is for instance prohibited within water catchment areas. Also, the initial release of sulphate should be considered, though no limit value has been defined. The second procedure applies to the use of e.g. steel slags and incineration ashes, which are often subject to extensive investigations (RVF 2002; SGF 2003). The latter implies lengthy and complicated permit processes, considerably decreasing interest among prospective users.
It were shown that the earlier the secondary materials are considered in the planning process of construction, the greater the chances for acceptance (SGF 2003; Visser 2003; Kärrman et al. 2004). Normally, the material selection occurs in the end of the design phase when the EIA of the project is finalised, i.e. when the site and the construction designs are settled. It is then too late to consider the use of alternative materials requiring supplemental investigations regarding both technical and environmental properties.

![Diagram](image)

**Figure 2** An overview of the legal process of utilisation of wastes.

Environmental assessments are inconsistent and of varying accuracy (Hartlén et al. 1999; Östman 2002; Visser 2003). No general procedure to investigate the suitability of using alternative materials in different applications exists, as well as any criteria established to assess what should be regarded as an acceptable load. Different county authorities can also put forward different demands for the use of waste and request different information to be submitted with the application (Lidelöw et al. 2003), causing confusion for authorities as well as for prospective users.

Evaluation of the expected environmental impact of an intended use is usually based on results from laboratory leaching tests. The applied test methods vary. The lack of comparability between investigations creates problems in interpreting the data which also may cause direct economic consequences (e.g. rejection or acceptance of materials) (Lidelöw et al. 2003). Conventional materials were shown to have a high and variable leachability of certain constituents e.g. zinc, nickel and copper (Tossavainen and Forssberg 1999). It is thus pertinent to compare the leaching characteristics of the secondary material with those of conventional materials, though such materials are rarely investigated regarding leaching properties. Relevant reference data are thus sparse.

Due to the relatively limited monitoring of secondary materials usage, little information about their long-term environmental performance exists. Moreover, existing technical specifications of aggregates were developed for conventional materials and do not necessarily give proper guidance when applied to alternative materials (Arm 2000; Vägverket 2001), and the dissemination and exchange of information about alternative materials and their potential usage applications is inadequate (Arm 2003; Kärrman et al. 2004). No routines for beneficial experienceable usage of different applications have been established. In daily practice, huge amounts of data are generated with the sole
purpose of satisfying the authority. Such data is generally discarded as soon as its application is fulfilled. Thus, there is a lack of relevant documentation about secondary materials (Lidelöw et al. 2003), as was also pointed out by Vägverket (1999) as one of the main reasons to refrain from using such materials.

**Concluding notes**

The amounts of secondary materials used in construction are lower than the potential, both with regard to the available amounts and their technical and environmental properties. For transport costs, such materials are primarily a viable option close to the production sources. Waste with high utilisation rates generally arise close to bigger cities, where the demand for construction materials is high, and has gained acceptance based on locally developed experience and empiric verifications. As a result, they have been released from the permit obligation and are considered for use on the same premises as traditional materials.

Lengthy permit processes are one of the main obstacles in the recycling of relatively “new” and untried secondary materials, as well as their usage being hampered by obscurities regarding the classification of waste. Amendments of the existing regulations are needed to reduce the administrative burden of approval practices. The lack of a general procedure for environmental assessments leads to uncertainty about the potential environmental risks and inconsistent judgments. Criteria for acceptance are lacking. Judging the relevance of an impact assessment is difficult due to the lack of environmental data on the conventional materials used. There is a need to characterise different materials, primary as well as secondary, and their potential applications to evaluate the critical factors for environmental assessments. Documenting and distributing the resulting data is crucial to avoid costly duplication of work in the future.
2.2 Critical factors for environmental assessments

The characteristics of industrial waste vary. Some of them, either inherently or as a consequence of the process from which they originate, may contain potentially harmful constituents such as heavy metals. Blast furnace slag, for example, has a high content of vanadium (Kanschat 1996), while ferrochrome slag is rich in chromium (Lind et al. 2001).

The potential risks for the release and spreading of contaminants are mainly related to diffuse spreading or leaching of constituents to soil, surface, and ground waters. Diffuse spreading, e.g. dusting, is not expected to cause substantial impact as long as the material is incorporated in, for example, a road construction. However, it may be of concern during placement or later erosion.

Leachate, i.e. water seepage through the material, is the predominant emission flow to be expected from materials enclosed in a construction. The degree of contaminant mobilisation from the solid material is dependant on chemical and physical factors such as
- pH
- redox potential
- presence of complexing agents
- temperature
- liquid-to-solid ratio (L/S ratio)
- particle properties, e.g. particle size
- permeability of the solid matrix
- material durability (regarding e.g. freezing/thawing, abrasion etc.)
- contact time between phases

The solubility of several species is influenced by pH. The mobility of most heavy metal cations e.g. lead, zinc and cadmium is strongly dependent on pH. The leachability of lead and zinc, for example, is at their minimum at pH 9-10 and increases at both lower and higher pH, while the solubility of cadmium, at its lowest around pH 11, increases with a decrease in pH (van der Sloot 1996; van der Sloot et al. 2001). The redox potential of a system may affect the solubility of metals, such as copper, chromium, and iron, by changing the oxidation states of the elements, or the degree of precipitation with e.g. heavy-metal sulphides (Fällman and Hartlé 1994). Complexation with inorganic or organic ligands may increase the solubility of certain constituents. For example, the leachability of cadmium is influenced by complexation with chloride (van der Sloot et al. 1996), while copper mobility is strongly affected by its affinity for organic material (Meima et al. 1999). Temperature affects chemical equilibrium as well as biological activity, which in its turn affects the generation of dissolved organic substances as well as redox potential and pH. The L/S ratio is important because it relates to time through the rate of infiltration. An increased L/S enhances leaching of highly soluble species such as sodium, chloride and potassium. A reduced particle size leads to larger surface accessibility, promoting the leaching of organic and inorganic species.

Rain and snowmelt either percolate through the material or flow around it depending on the permeability of the material and its surroundings (Figure 3). The leaching behaviour of constituents can often be described in terms of either (i) availability or solubility
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controlled for percolation-dominated scenarios with loose granular materials, or (ii) mass-transfer controlled for “flow around” scenarios with monolithic materials, compacted granular materials or granular materials overlain by low permeability covers (Kosson et al. 1996).

Availability corresponds to the maximum leachable amount of a constituent and may control the release of, e.g. sodium and chloride. Solubility control occurs when a solution in contact with a material is saturated by a certain constituent. Such a condition is prevalent at low L/S ratios that typically occur in the field. For example, L/S ratios greater than five may be obtained over an interval of fifteen years or longer for a road base application with relatively high rates of infiltration (Lidelöw and Lagerkvist, in manuscript). In mass transfer controlled scenarios, diffusion is the rate limiting step. As indicated in Figure 3, the management scenario also affects the potential gas exchange with the atmosphere and thus also the redox state of the system. The redox potential might be further influenced by biological activity through the decomposition of organic material.

![Figure 3](image_url)

**Figure 3** Illustration of percolation and diffusion controlled release conditions in a field scenario. Adapted from Kosson et al. (1996).

In Lidelöw and Lagerkvist (in manuscript) two years of leachate emissions from a large-scale test road built of MSWI bottom ash and crushed rock were evaluated. Leachate from the respective materials was collected under the slopes outside the cover as well as in the middle of the road under the cover. It was shown that the majority of leachate was generated from the uncovered parts along the borders of the road base. The flow was generally higher through the crushed rock than through the bottom ash, probably due to differences in the permeability.

In the bottom ash layer, the release rates from the middle of the road were significantly lower than from the uncovered slopes (Figure 4). For example, leachate from the slopes contained 0.05-0.2 g/l of chloride after two years of leaching, while concentrations in leachate generated below the cover ranged from 1 to 2 g/l. A similar trend was observed for sodium. Thus, the cover of the road prevented a rapid wash out of chloride and sodium from the ash. No similar observations could be made for leachates from the crushed rock.
Estimated cumulative releases from the slopes indicated that highly soluble constituents may be released up to their availability within a matter of years. Such estimations are of course specific to the range of field conditions considered and the species of interest. Critical field conditions are for instance the amount of infiltration and the mode of water contact (i.e. diffusion or percolation controlled leaching). In the field, wet/dry cycles as well as variations in the degree of water saturation may also occur, reducing the estimated releases. For the use of bottom ash as a compacted granular road base (0.45m) and 35% exposure to wetted conditions, Kosson et al. (1996) estimated the reduction of chlorides release to a factor of 0.59. Several changes in the leaching chemistry may occur over time, e.g. changes in the pH and the redox conditions. The L/S ratio does not consider such changes limiting the possibilities for predictions, especially for metals and nitrogen (Kylefors et al. 2003).

Figure 4  PCA score plot for leachates from a road sections built of MSWI bottom ash (Lidelöw and Lagerkvist, in manuscript).

The quality of the leachates generated under the slopes and under the cover of the road differed, mainly due to the characteristics of ash leachate. Conductivity decreased when the L/S ratio increased, i.e. in the slopes. A similar trend was observed for trace metals such as cadmium, lead, and zinc. As previously stated, changes in leachability of these species may occur as a result of changes in pH. The pH was slightly higher in slope leachates (pH ~11.0) than in leachates generated below the cover (pH ~10.6), possibly influencing the decrease in cadmium concentrations. The reduced chloride concentration could also partially account for the reduced solubility of cadmium in the slopes. Reduced lead and zinc concentrations would be expected if the pH had been lower in the slope leachates than in leachates from the middle of the road. Since this was not the case, dilution with infiltrating water is the likely explanation of the reduction.

Aluminium and chromium showed increased mobility with an increasing degree of material wetting. Such effects were also noticed by Johnson et al. (1999), though they are not well understood. Since the pH and aluminium concentration of the leachates were shown to be strongly correlated, the increased Al mobility might also be due to the relatively higher pH of the slope leachates. Correspondingly, the mobility of chromium could have been influenced by fluctuating redox conditions. The initial drop in
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alkalinity indicated carbonation of the bottom ash. In contrast, the concentrations of ammonia nitrogen gradually decreased, while the concentrations of nitrate nitrogen increased, indicating oxidation of the ash upon prolonged contact with air.

The quality of the leachates was changing markedly over time. A significant difference ($\alpha=0.05$) between leachates from the first sampling season and the two following years of sampling was found. The difference between leachates from bottom ash and crushed rock was significant but decreased over time, mainly due to changes in the ash leachate composition (Figure 5).

![Figure 5](image)

**Figure 5** PCA score plot for observations from the test road illustrating the largest variation in the data explained by the use of two different materials: MSWI bottom ash and crushed rock (along PC1), and time dependency of data (along PC2). Modified after Lidelöw and Lagerkvist (in manuscript).

Most constituents occurred at high concentrations during the first few months of leaching and then rapidly decreased. Such trends were observed for e.g. copper, chromium, chloride, sodium, potassium, and calcium in the ash leachate. However, some elements, e.g. aluminium in ash leachate and zinc in rock leachate, showed high and largely fluctuating concentrations throughout the two years of sampling.

Seasonal variations of the leachate composition were observed in ash leachates from the slopes. An increased release of for instance copper, cadmium, easily leachable constituents (e.g. sodium, chloride, potassium, and sulphate), total nitrogen, and organic material occurred during the spring, while higher concentrations of nitrate nitrogen, nitrite nitrogen, nickel, and barium were found in autumn leachates. High concentrations of suspended material are commonly found in road discharges during snow melting. The season is also an important factor in assessing biological effects on recipients. The water quality during the summer is for example critical for the young of aquatic animals.

Considering the leachate concentrations, copper, chromium, aluminium, sodium and chloride were the most important pollutants from the bottom ash, while zinc was the main pollutant from the rock material. These figures however only apply to emissions from the boundary of the construction. Attenuation processes may occur close to the source, thereby reducing the concentrations downstream to acceptable levels. One
exception might be the chloride ion, which is not adsorbed to soil particles and therefore highly mobile in soil and groundwater.

The environmental impact of the released contaminants is dependant on their potential to be transported to the recipient. The properties and the variability of the substrata, e.g. with regard to the permeability, groundwater level and distance to surface water aquifers, thus need to be considered. Knowledge about the geology and hydrogeology of an area can be used to assess suitable and unsuitable areas for use of different materials and to design the construction as to reduce the pollution potential.

The impact of the leachate emissions will depend on the quality and characteristics of the leachate and the recipient. For example, an adequate recipient for leachates with high loads of salts would be seawater rather than fresh water. The land use must also be taken into account since this will affect the degree of exposure via different pathways. In addition, the regulations may put restrictions on the area considered such as water protection area, biologically protection worthy area etc.

Concluding notes
It is not possible to assess the risk of using a material in general terms. The effect of a specific location and methods of use must be considered. The widely different nature of various construction materials and the diverse application scenarios call for an identification of the relevant exposure conditions, since they will lead to different release rates and extents for the same material.

Among the factors that need to be covered are:

- The characteristics of the source of contaminants in terms of release controlling factors, i.e. pH, redox, complexing agents etc.
- The hydrology of the application, i.e. how much infiltration will occur and what are the preferential flow paths.
- The changes in release conditions with time due to processes such as depletion of soluble species, biological activity, carbonation, oxygen exclusion and decreasing permeability as a consequence of pore clogging or traffic load.
- The behaviour of the released constituents in the substrata of the construction.
- The recipients to be evaluated in the consideration of acceptance or rejection as well as recommended mitigation measures.

The understanding of the chemical and physical factors that influence leaching is a prerequisite for predicting leaching from materials under varying conditions. However, release and migration of pollutants in specific application scenarios may be estimated only when data on environmental conditions e.g. groundwater flow, precipitation frequency and amounts, and the construction design e.g. geometry, isolation measures and degree of compaction are included.

Environmental changes may affect the release of contaminants in the future, and the potential impacts of such processes need to be reviewed. For example, the long-term leaching behaviour needs to be addressed in relation to the durability of the construction which may be affected by both physical and chemical factors such as freezing/thawing, abrasion and chemical dissolution. It is also of interest to discuss what could happen if the construction is subject to extreme external loads such as flooding or extensive
reconstructions. The potential for sudden, uncontrolled releases of pollutants in such cases need to be investigated.
2.3 Methods for environmental assessments

Given knowledge about the material and its placement, different methods could be applied to estimate the potential environmental impacts and to demonstrate the acceptability of an intended use.

There are at least three different approaches to be noted (Lidelöw and Lagerkvist 2004):
- Laboratory/field verification of leaching behaviour
- Risk assessment
- System analysis

Most often, the potential use of secondary materials is environmentally evaluated by means of laboratory leaching tests. In Sweden, it is recommended to perform a characterisation procedure based on determination of the total elemental composition, the maximum potential release both under reducing and oxidising conditions (long term perspective), and the leaching behaviour over time (short term perspective) (Vägverket 1999; RVF 2002; Visser 2003).

In relation to leaching tests, case-specific risk assessment is a more comprehensive method for assessing the environmental effect of a material. Specific procedures based on this method have been presented in some countries, e.g. Denmark, Sweden, Finland, the Netherlands and France (Dahlström and Rasmussen 1999; Hartlén et al. 1999; Wahlström et al. 1999; Sorvari 2000; Eikelboom et al. 2001; Domas et al. 2003). The Swedish system is only a proposal based on general principles. Simplifications and development of a suitable tool-kit would be needed before implementation.

The need for using a broader perspective to address pollution and resource aspects of the use of different construction materials is addressed by several authors, e.g. Vägverket (2002), Birgisdóttir et al. (2003) and Roth and Eklund (2003). Life-cycle assessment (LCA) has therefore been suggested as a suitable tool but is so far used to a limited extent within the earth construction industry.

Laboratory/field verification of leaching behaviour

Constituent release can be estimated through measurements of fundamental leaching properties such as availability, solubility and diffusion coefficient in different laboratory leaching tests (Bergman 1996; Kosson et al. 1996).

The maximum potential release of a constituent can be assessed using availability tests. The availability of a specific element is generally less than the total content of that element. For example, expressed as a percentage of the total content, Zn (50%), Ca (30%) and Cu (20%) were the most leachable metals in the ash used in the field study (Lidelöw and Lagerkvist, in manuscript).

Determining the availability gives no indication of whether or not this maximum quantity of a particular constituent will be released, or over what time period the release will occur, for the exposure scenario of interest. The bottom ash used in the test road showed a high availability of lead, zinc, cadmium, calcium and nickel as compared to rock materials, while the leachability of arsenic and mercury was in the same range as that of rock materials. Also the availability of sulphate was markedly high in the ash. A comparison of leachates generated from bottom ash and crushed rock in the field study
is displayed in Figure 6. Variables positioned closer to “bottom ash” in Figure 6 had higher average values in ash leachate and vice versa, while variables positioned close to origin did not differ between crushed rock and bottom ash leachates. As shown in Figure 6, higher concentrations of zinc and calcium were leached from the rock material than from the ash. No significant difference in the concentration of lead, nickel and sulphate between the leachates was found. Figure 6 indicates a higher release of arsenic and mercury from the ash. However, the concentrations were very low and fell below the analytical detection limits after the first few months of leaching.

**Figure 6**  Weight plot for PLS model showing the effect of MSWI bottom ash and crushed rock on the quality of leachates generated from the test road over two years (Lidelöw and Lagerkvist, in manuscript).

Availability tests do not reflect actual field conditions and should not be used as the sole tool to evaluate and compare the potential load of contaminants from different materials. However, they may provide a first step assessment useful for screening purposes. In order to provide leaching data that is more accurate for the material in question and possible field conditions, the release over a range of environmentally relevant pH and L/S ratios (e.g. pH 4-13 and L/S 0.5-10) may be determined using pH dependence leaching tests and percolation tests. In addition, mass transfer rates of constituents may be estimated to account for the waste form using diffusion leaching tests.

No standard method will be generally applicable. Different materials, applications and environmental conditions require different information needs. As an alternative to attempt a full simulation of a scenario in one test, a combination of simple assays may yield sufficient information for many scenarios. Such an approach for evaluating leaching of constituents from different waste under specific conditions was suggested by e.g. Bergman (1996), CEN (1996), Kosson et al. (2002) and van der Sloot (2003).

The studies are however mainly restricted to inorganic compounds, since relevant and practical leaching tests for organic contaminants are lacking. Moreover, as most laboratory tests are carried out under saturated conditions, further work is needed to account for wet/dry cycles as well as variations in the degree of water saturation. Test
methods to assess the impact of material ageing under anoxic or reducing conditions are also a subject of future development. The impact of such conditions could be studied using physical simulator tests, by which also the influence of biological activity in the material could be considered (Bergman 1996; Kylefors et al. 2003). However, such simulations are time consuming and expensive to perform.

Lysimeter tests, where naturally produced leachate is collected and analysed, are useful to verify the results of laboratory tests. Differences between laboratory and field observations will indicate the impact of different environmental conditions. In order to address the time perspective of the emissions, long term field data for validation are needed.

Risk assessments
A relation between the results of leaching tests and the desired level of protection of human health and the environment can be estimated using risk assessments. The risk associated with the use will thus be assessed by means of different criteria aiming to protect e.g. soil and drinking water quality. All of the above mentioned procedures include leaching-based criteria. Criteria based on solid-phase analysis are only used for evaluation of health risks associated with direct exposure via e.g. oral intake or inhalation, and for assessing organic contaminants.

Within environmental impact assessments, predictions of potential impacts are typically limited to judgements that particular consequences are “likely” or “unlikely”. Risk assessment, in contrast, stresses formal quantification of probability and uncertainty (Andrews 1990). The probability that damage will occur is related to factors such as volume of materials used, potential for release and transport of pollutants transferred to the actual application and environment, and the state of the recipient. The scenarios are diverse and depend on many factors that are characterised by uncertainties. Proper risk assessments thus always require case-specific data. However, such data may not be available (or readily available) or may not be necessary. For these situations, default scenarios may be applied.

In the Danish and the Dutch procedures, for example, the effect of site-specific conditions and of the construction on the leaching of contaminants is considered in simplified models. The factors included are infiltration of rainwater, isolation measures, as well as the height and bulk density of the material. In the Danish system, seven scenarios with different levels and combinations of these factors are defined. The transport to and dilution within the recipient considered, in this case the groundwater aquifer, is represented by a dilution factor. In the Dutch system, different infiltration rates are assumed for scenarios where infiltration measures are or are not used. The transport is not modelled explicitly, since only immission to soil is considered. The time development of leachate emissions is not considered in any of the systems. Hence, the maximum acceptable pollutant emissions may occasionally be exceeded as long as they are not exceeded by the averaged values for a fixed period (e.g. 100 years).

Criteria for acceptance can be either generic or case-specific. The former generally implies quantitatively expressed restrictions, e.g. limit values, on concentrations or cumulative releases of substances. Examples include the limits for immission of substances to soil used in the Netherlands and the criteria based on drinking water standards derived in Denmark. The limits are defined for the source term (i.e. the
release of contaminants from the material) and not for the target points. The latter
implies an evaluation based on an assumed discrepancy between the application and the
state of the recipient. Examples include permits or notifications (i.e. use on conditions).
The state of the recipient may be assessed, for instance, through comparisons with
background values of surface and ground waters.

Applying limit values facilitates the evaluation and introduces clear restrictions. On the
other hand, this renders consideration of land uses, sensitivity and protection value of
the environment impossible. A case-specific assessment requires larger resources but
will provide a more accurate evaluation of the risk (i.e. it is less likely to over- or
underestimate the risk).

**Life-cycle assessments**

LCA is a method to assess the potential environmental effect of a material at any stage
of its life cycle. LCA can be used to make comparative studies for different construction
materials (ISO 1997; Vägverket 2002). It can also be used to identify hot-spots for
specific materials or activities, where measures made to decrease the environmental
impact would be especially efficient (Vägverket 2002; Roth and Eklund 2003;
Flemström et al. 2004). Another aim could be to compare different waste management
alternatives, e.g. landfilling versus recycling, of a material. Thus, LCA has a good
potential for use in planning at the strategic level. One example is a LCA based study of
the environmental impacts of the building sector prepared by Byggsöktorns
kretsloppsråd (2001). The results showed that the large amounts of mineral materials
used and the strategy of the waste management applied are the major environmental
impact of earth construction. The study led to the formulation of policies and plans of
action focussing on the identified hot spots.

The final interpretation of results from LCA studies is based on simplified assumptions
and subjective judgements of what environmental impact that is most important (Owens
1997; Vägverket 2002; Roth and Eklund 2003). Different impact assessment methods or
models for characterising and weighting may lead to very different results. Also, since
system boundaries and case specific data affect the outcome, the results cannot be
generally applied. LCA studies are thus often irrelevant for a specific utilisation.
Moreover, the LCA demands large resources and performing simplified LCAs may give
misleading results.

**Concluding notes**

The environmental evaluation of secondary materials should be based on interactions of
materials and their environment rather than their total content of potentially harmful
constituents.

Which material that is environmentally preferable is difficult to assess based on
standard leaching tests since (a) the tests used do not necessarily reflect the leaching
behaviour in the field, and (b) the results only apply to the source term of contaminants.
Field verifications are needed to investigate discrepancies between field and laboratory
leaching data. Long term data for validation are however sparse, indicating a need for
follow-up studies of old constructions. A database of reference objects could be
established to make better use of experience. Risk assessment could provide a relation
between the results of leaching tests and the maximum acceptable load at specific
targets. However, such assessments require large amounts of case-specific data and

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need to be simplified, *e.g.* through the use of predefined scenarios, in order to find practical use. Simplifications minimise the amount of factors to be considered, but also introduce generalisations that have to be taken into account when evaluating the risk. Life-cycle assessments may be used to compare the life-cycle impacts of different materials from a system wide perspective, though results from such studies often are irrelevant for a specific utilisation scenario.

The development of a general procedure based on different scenarios, and using combinations of different investigations is a promising path that should be further explored. Special emphasis should be put on the analysis of which critical factors that can make a method work for one material and application but fail for another. Possibly, a decision tree outlining the relevant factors for various scenarios could be developed. Also, an analysis of how different evaluation methods for characterisation and weighing of factors will influence the outcome is needed.
3 CONCLUSIONS

The amounts of secondary materials used in construction are lower than the potential. For transport costs, such materials are primarily a viable option in urban areas. The main obstacle for recycling is the long and complicated permit process involved. Further, the lack of a general procedure to investigate the suitability of an intended use leads to uncertainty about the potential environmental risks and inconsistent assessments.

Conventional materials are usually not investigated with regard to environmental properties. Such materials were however shown to generate leachates with high concentrations of certain pollutants. In all reason, the criteria for utilisation should be equal for primary as well as secondary materials.

It is not possible to assess the risk of using a material in general terms. The effect of a specific location and methods of use must be considered. The widely different nature of various construction materials and the diverse application scenarios call for an identification of the relevant exposure conditions since they will lead to different release rates and extents for the same material.

The understanding of the chemical and physical factors that influence leaching is a prerequisite for predicting leaching from materials under varying conditions. However, release and migration of pollutants in specific application scenarios may be estimated only when data on both the environmental conditions and the construction design are included. Environmental changes may affect the release of contaminants in the future, and the potential impacts of such processes need to be reviewed. For validation, long term field data for various management scenarios are needed. A database of reference objects could be established to make use of experience from existing constructions. Documenting and distributing environmental evaluation data is important to avoid costly duplication of work in the future.

The development of a general procedure based on different scenarios, and using combinations of different investigations is a promising path that should be further explored. Special emphasis should be put on analysing which critical factors that can make a method work for one material and application but fail for another. Possibly, a decision tree outlining the relevant factors for various scenarios could be developed. Also, an analysis of how different evaluation methods for characterisation and weighing of factors will influence the outcome is needed.
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