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N O R D I C W O R K I N G P A P E R S

ENCORT-CDW

Evaluation of the European recovery target for construction and demolition waste

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Concrete, bricks & ceramics, wood and gypsum-based construction materials

Bituminous mixtures (asphalt) and track ballast

ENCORT-CDW

Evaluation of the European recovery target
for construction and demolition waste

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Content

<i>Summary with main conclusions and recommendations</i>	5
<i>Extended summary</i>	9
Introduction	9
Project methodology and waste fractions studied	9
Recovery of bituminous mixtures	10
Recovery of concrete waste.....	13
Recovery of bricks, tiles & ceramics and mixtures of these and concrete	14
Recovery of track ballast.....	15
Recovery of gypsum-based construction materials	16
Recovery of wood waste	18
General discussion and recommendations.....	19
<i>1 Introduction</i>	21
1.1 Background.....	21
1.2 Purpose and goal	21
1.3 Limits and prerequisites	21
1.4 Methodology	22
1.5 Definitions	23
1.6 Identification of material flows and waste types	25
<i>2 General Nordic legislation and guidance on construction and demolition waste</i>	29
2.1 Denmark.....	29
2.2 Finland.....	30
2.3 Norway	30
2.4 Sweden.....	31
<i>3 Bituminous mixtures not containing coal tar – “asphalt”</i>	35
3.1 Description and uses of asphalt.....	35
3.2 Demolition of paving asphalt.....	36
3.3 Legislation and guidance regarding asphalt recycling.....	37
3.4 Recovery operations	39
3.5 Disposal/landfilling of recovered asphalt.....	41
3.6 Resources saved and impact on transports.....	41
3.7 Pollution content and release from reclaimed asphalt.....	47
3.8 Emission scenarios and calculated release/spread of pollutants	53
3.9 Comparison of scenarios	65
<i>4 Concrete waste</i>	69
4.1 Description and uses of concrete	69
4.2 Demolition of concrete	70
4.3 Legislation and guidance regarding recycling of concrete	71
4.4 Recovery operations	75
4.5 Disposal/landfill of concrete waste.....	76
4.6 Resources saved and impact on transports.....	77
4.7 Pollution content and release from recovered concrete	80
4.8 Emission scenarios and calculated release/spread of pollutants	81
4.9 Comparison of scenarios	88

5	<i>Bricks, tiles & ceramics – and mixtures of these and concrete</i>	91
5.1	Description and uses of bricks, tiles & ceramics	91
5.2	Demolition of bricks, tiles & ceramics	91
5.3	Legislation and guidance regarding recycling of bricks, tiles & ceramics	92
5.4	Recovery operations	92
5.5	Disposal/landfilling of bricks, tiles & ceramics waste	94
5.6	Resources saved and impact on transports	94
5.7	Pollution content and release from recovered bricks, tiles & ceramics	94
5.8	Emission scenarios and calculated release/spread of pollutants	96
5.9	Comparison of scenarios	98
6	<i>Track ballast</i>	101
6.1	Description and uses of track ballast	101
6.2	Generation of track ballast as a waste fraction	101
6.3	Legislation and guidance regarding recycling of track ballast	103
6.4	Recovery operations	104
6.5	Disposal/landfilling of track ballast waste	105
6.6	Resources saved and impact on transports	106
6.7	Pollution content and release from recovered track ballast	107
6.8	Emission scenarios and calculated release/spread of pollutants	111
6.9	Comparison of scenarios	115
7	<i>Gypsum-based construction materials</i>	117
7.1	Description and uses of gypsum-based construction materials	117
7.2	Demolition of gypsum-based construction materials	117
7.3	Legislation and guidance regarding recycling of gypsum based construction materials	117
7.4	Recovery operations	118
7.5	Disposal/landfilling of gypsum-based waste	121
7.6	Resources saved and impact on transports	122
7.7	Pollution content and release from recovered gypsum-based materials	123
7.8	Emission scenarios and calculated release/spread of pollutants	127
7.9	Comparison of scenarios	130
8	<i>Impact of the 70 % target on the handling of wood waste</i>	133
8.1	National data	133
8.2	Screening LCA	136
9	<i>Discussion and conclusions</i>	147
9.1	Which are the environmental consequences of the various scenarios studied?	147
9.2	Suggestions for the next generation of the 70 % target	151
9.3	How can the current data be improved?	152
9.4	How to promote the use of recycled products in the future?	153
	<i>References</i>	155
	<i>Annexes</i>	165
A	Extracts from the Waste Framework Directive 2008/98/EC	
B	Extracts from Decision 2011/753/EU	
C	European harmonized standards related to construction and demolition waste	
D	Examples on additives in bituminous binders	
E	Using leaching test results to assess the release of substances from granular C&D waste	

Preface

This document is the final report of the Nordic cooperation project entitled: “ENCORT-CDW – Environmental consequences of the EU recovery target for construction & demolition waste”.

The project was funded by the Nordic Council of Ministers, the Swedish Geotechnical Institute, Ångpanneföreningen's Foundation for Research and Development, the Foundation for the Swedish Environmental Research Institute, the Swedish Environmental Protection Agency, Norwegian Environment Agency and the Ministry of the Environment Finland.

The background of the project was the revised framework for waste management in the EU which was adopted in 2008. The [Waste Framework Directive](#) (WFD) includes a target for recovery of construction and demolition waste (C&D waste) which reads

“by 2020, the preparing for re-use, recycling and other material recovery, including backfilling operations using waste to substitute other materials, of non-hazardous construction and demolition waste excluding naturally occurring material defined in category 17 05 04 in the list of waste shall be increased to a minimum of 70 % by weight.” (WFD 2008/98/EC Article 11(2)(b))

The target was added during the final negotiations of the Directive text and thus did not include any consequence analysis. In 2014 (at the latest) the WFD target is to be evaluated and maybe strengthened. In order to provide a basis for assessing the consequences of the target, the Swedish Environmental Protection Agency launched this project.

The main objective of the project was to provide the Nordic EPAs a basis for assessing the consequences of the WFD target mentioned above, in terms of resource management and diffuse pollution dispersion. This basis will be used

- when focusing the efforts to achieve a flow of materials from the construction and demolition sector with minimal impact on the environment,
- to develop means of control for increased re-use and recycling of C&D waste,
- in further discussions with the EC, to give scientific facts to what effects the 70 % target in the Directive might have in the Nordic countries.

The report contains a description of:

- Possible future recovery scenarios and recovery scenarios currently available for a selected set of C&D waste.
- Quantification of the resources saved and the impact on transports involved for each waste depending on how it is recovered.
- Quantification of the potential spread of pollutants for each recovery option.
- A screening LCA comparing re-use, recycling and energy recovery of wood waste.

The project was carried out during 2012 and 2013 by a Nordic project group consisting of the following persons:

Maria Arm (project manager) and Ola Wik SGI, Sweden
Christian J. Engelsen SINTEF, Norway
Martin Erlandsson and Jan-Olov Sundqvist IVL, Sweden
Anke Oberender and Ole Hjelmar DHI, Denmark
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The project work has been followed by a steering group consisting of:

Henrik Sandström and Erika Nygren, Swedish Environmental Protection Agency
Jon Fonnliid Larsen, Norwegian Environmental Protection Agency
Else Peuranen, Ministry of the Environment, Finland
Metta Wiese, Denmark, has been project manager from the Nordic Council of Ministers.

January 2014

ENCORT-CDW

Evaluation of the European recovery target for construction and demolition waste

Summary with main conclusions and recommendations

The revised framework for waste management in the EU (WFD), adopted in 2008, includes a target for recovery of construction and demolition waste (CDW) which reads

“by 2020, the preparing for re-use, recycling and other material recovery, including backfilling operations using waste to substitute other materials, of non-hazardous construction and demolition waste excluding naturally occurring material defined in category 17 05 04 in the list of waste shall be increased to a minimum of 70 % by weight.” (WFD 2008/98/EC Article 11(2)(b))

The target was added during the final negotiations of the Directive text and thus did not include any consequence analysis. In 2014 the WFD target is to be evaluated.

In order to provide a basis for assessing the consequences of the target, the Swedish Environmental Protection Agency launched a Nordic project, ENCORT-CDW, the results of which is presented in this report. The study was aimed at resource management and diffuse pollution dispersion related to waste types, which were deemed to have the biggest impact on fulfilling the WFD target on CDW recovery.

The following fractions of C&D waste were studied: Bituminous mixtures not containing coal tar (“asphalt”); Concrete; Bricks, tiles & ceramics and mixtures of these and concrete; Track ballast; Gypsum-based construction materials and Wood.

Recovery scenarios were identified and estimations were made regarding expected savings in primary materials resources, impact on transports as well as pollution and emissions. Information was retrieved from databases, literature and personal contacts. In those cases where relevant data was lacking, a qualitative evaluation was done.

For wood waste, the main differences between re-use, material recycling and energy recovery were evaluated by means of collecting national data from the Nordic countries and performing a screening LCA.

Main conclusions

The EU recovery target does not ensure a sustainable waste recovery in its present form.

- **It favours recycling of high density waste types.** The result is that mineral wastes will have the largest impact while the largest environmental benefits might be on other waste types.

- **It does not favour the most sustainable recovery operations.** Above all, it does not distinguish between backfilling and other more resource efficient recovery operations. Since backfilling is a recovery option that generally results in both low benefits and future environmental risks, this increases the risk for “downcycling”, which means that the waste is not recovered in the most optimal way.
- **It is very sensitive to interpretations of what is considered as waste and waste recovery.** This fact is significant, since the WFD definitions of waste recovery actions such as re-use or recycling, are mainly aimed at the building construction field and does not fit well with materials recovered within other construction fields. As an example, asphalt and track ballast, which represent large material flows with a high re-use or recycling rate, are generally not included in the waste statistics and this will highly influence the interpretation of attaining the target.

Recommendations

Based on the conclusions the following recommendations are given regarding the EU recovery target:

- Transform the general weight based target into waste specific targets. This would favour recycling of CDW in general and not only for high weight materials.
- Rank the recovery operations in the calculation method for reporting progress. Backfilling should not be regarded as equal to other operations for recovery.

Furthermore, we recommend the Nordic countries to:

- **Improve the knowledge of waste flows and waste handling** in order to monitor progress regarding a sustainable CDW recovery. This is crucial since the current Eurostat waste data quality is low. It does not allow an assessment of resources saved, environmental benefits gained or potential environmental risks. One example is that statistics on mineral materials with high grade re-use or recycling options are often reported merged with low grade mixed mineral wastes and contaminated wastes. Another example is that statistical data does not distinguish between contaminated (painted) wood and pure wood suitable for material recovery.
- **Set up national recovery targets and management measures** so that the most effective recycling pathways are promoted and those posing the biggest environmental risk are avoided. National targets should be waste specific and operation specific, promotion should be done through facilitation of favourable options in technical specifications, guidelines and regulations and finally, prevention of undesired operations should be done through environmental quality criteria. High quality recycling is promoted if the waste fractions are successfully separated and there is a system for quality assessment and declaration of the waste fractions produced.
- **Improve the knowledge of pollution contents of CDW and emissions associated with the recycling.** A substantial share of CDW exceeds (the more or less applicable) environmental criteria. Available data, however, contains significant gaps due to scarce data, biased data, outdated information and inadequate or poorly documented sampling and measurement methods. To enable better assessment of environmental risk than currently possible, harmonized monitoring methods and collection of data are needed. This is a suitable area for Nordic cooperation.
- **Promote recycling of wood waste in new construction materials.** At present, wood waste is mainly incinerated with energy recovery, but there is a large opportunity for reducing the Global Warming Potential if material recovery is increased. However, further knowledge about technical and environmental obstacles and possibilities is required.

Some differences between the Nordic countries justify recovery targets and management measures tailored to national circumstances. However, there are also many similarities which make an increased exchange of knowledge on waste properties, risk assessment and management measures advantageous. A common approach also creates improved opportunities for the Nordic countries to influence European legislation.

Extended summary

This section is a short version of the full report.

Introduction

The revised framework for waste management in the EU (WFD) which was adopted in 2008 includes a target for recovery of construction and demolition waste (C&D waste) which reads

“by 2020, the preparing for re-use, recycling and other material recovery, including backfilling operations using waste to substitute other materials, of non-hazardous construction and demolition waste excluding naturally occurring material defined in category 17 05 04 in the list of waste shall be increased to a minimum of 70 % by weight.” (WFD 2008/98/EC Article 11(2)(b))

The target was added during the final negotiations of the Directive text and thus did not include any consequence analysis. In 2014 (at the latest) the WFD target is to be evaluated and maybe strengthened. In order to provide a basis for assessing the consequences of the target, the Swedish Environmental Protection Agency launched this project.

The main objective of the project was to provide the Nordic EPAs with a basis for assessing the consequences of the WFD target mentioned above, in terms of resource management and diffuse pollution dispersion. This basis will be used

- when focusing the efforts to achieve a flow of materials from the construction and demolition sector with minimal impact on the environment
- to develop means of control for increased re-use and recycling of C&D waste
- in further discussions with the EC, to give scientific facts to what effects the 70 % target in the directive might have in the Nordic countries.

Project methodology and waste fractions studied

A scenario-based approach was chosen for the study. This was due to ambiguities in how to apply the definition of waste, which in parts has led to shortcomings in the current statistics and as a consequence considerable uncertainty as to what material types and what quantities are/should be covered to meet the WFD recovery target.

The study has identified significant material flows and has described the effects of various recovery scenarios. Information and documentation required have been retrieved from databases, literature and personal contacts. The WFD definitions of C&D waste, re-use, recovery, material recovery, recycling and backfilling have been used and are given in the Definition section of the report.

Waste fractions studied

The following fractions of C&D waste have been studied.

- Bituminous mixtures not containing coal tar (“asphalt”)
- Concrete
- Bricks, tiles & ceramics and mixtures of these and concrete
- Track ballast
- Gypsum-based construction materials
- Wood

The selection represents waste streams of importance to fulfil the WFD target.

Possible recovery scenarios, recourses saved and impact on transports

Recovery scenarios currently available and recovery options that are possible in the future have been compiled. In addition, the savings in primary materials resources by replacing primary materials with waste materials have been identified and quantified using an appropriate functional unit. The impact on transports that each scenario entails has also been estimated. This evaluation can be done in several ways and some alternatives are shown in the report.

Pollution and emissions

Information about the pollution content of the various wastes and potential quality changes in the future have been compiled. Emission scenarios showing the potential for spreading of pollutants to air, land and water in connection with various recovery scenarios have been described for each selected waste type. A scenario for landfilling was also described.

For each scenario the release of pollutants was calculated for a selected number of substances and normalized to an appropriate functional unit (see appendix E for a short description on how results from a leaching test can be used to estimate the release to soil and groundwater). A service life of 100 years was chosen according to the European standard for environmental product declaration of construction products (EN 15804). In those cases where relevant data was lacking, a qualitative evaluation was done.

Finally, a comparison of different recovery scenarios with each other and with the landfilling scenario was carried out for each waste type.

Special study for wood waste

The wood waste study followed another procedure than described above. Energy recovery through incineration of wood waste is not included in the 70 % target, but frequently used in the Nordic countries. Therefore, the main differences between re-use, material recycling and energy recovery were evaluated by means of collecting national data from the Nordic countries and performing a screening LCA.

Note

It is important to note that no actual risk assessment has been made and no LCA was performed except for wood waste. The collected data in this report is relatively limited and only given to show examples of the expected pollution content in, and release from, C&D waste.

Results from leaching tests can be used to evaluate the environmental impact on soil, ground- or surface water of a given material applied in a given manner. However, the test results alone do not provide the answer. They must always be interpreted within the framework of a specific application or construction scenario.

Recovery of bituminous mixtures (EWC 170302)

The three major types of bituminous products are asphalt-based paints, roofing asphalts and paving asphalts. Asphalt-based paints represent a very small fraction of asphalt products and are not subject to recovery activities. They may however contaminate other construction wastes such as crushed concrete if not removed prior to demolition. Asphalt based roofing materials can be recycled into paving asphalt by existing techniques. Even though the amount of asphalt roofing material is small compared to paving asphalt, the bitumen content is appreciable.

Paving asphalt is a mixture of mineral aggregate, bituminous binder (usually 3-7 % by weight), filler and additives. The recovery of Reclaimed Asphalt pavement (RA) is an important and well established measure within the Nordic road pavement production, with a variety of demolition and recovery routines. RA is regularly recycled into new asphalt mixtures or used as aggregate in unbound layers. Available recovery scenarios are *e.g.*

- re-use in stationary or mobile asphalt plants to produce new asphalt mixes
- re-use through hot in-place recycling – to produce new surface or binder course
- recycling as aggregate in “unbound” road layers (bases and sub-bases)
- recycling as surface layer of gravel roads and parking areas
- recovery in mixed filling material (= backfilling).

The statistics on asphalt pavement recovery in the Nordic countries are uncertain. Available statistics from the European Asphalt Pavement Association indicate that large amounts of RA are generated in the Nordic countries (about 3.5 Mtonnes per year), corresponding to about 10-20 % of the total production of new asphalt mixes, and the recovery rate is high, well above the 70 % target set in the

WFD. The two most common recovery scenarios of RA in Sweden and Denmark are as input to new asphalt pavements through either hot or warm mix in stationary or mobile asphalt plants. In Norway, where the statistics are of better quality, it is most common to recycle the asphalt as aggregate in new “unbound” road layers. Prime reason for uncertain statistics is that RA recovery is not reported as waste handling and statistics are not kept regarding technical quality, contamination or recovery routes.

Conditions for asphalt recycling vary in the Nordic countries. In Sweden where rock quarries are ample, asphalt plants are generally located at or close to an aggregate production site and the transport distance for aggregates to the plants are negligible. Denmark however, imports high quality crushed rock aggregates from Sweden or Norway. Imported aggregates are mostly used in surface and binder courses while domestic low quality aggregates are used in bound bases. In Sweden, use of tar was widespread in the past and there is a substantial fraction of tar contaminated asphalt in old pavements. In Norway, the use of tar was less common and only a small fraction of the pavements is contaminated by tar. The widespread use of studded tires in Sweden, Norway and Finland generates a substantial wear of the pavement surface and thus a route for emission of potential contaminants.

Resources saved and impact on transports

Recycling of RA into new asphalt mixtures and bound courses could in some cases be regarded as re-use since the RA is used again for the same purpose for which it was conceived. Recycling into unbound layers is on the contrary a material recovery technique where the binder properties in the RA raw materials are not fully utilised.

The use of RA in production of asphalt pavements is an area which is covered by harmonized European standards and national guidelines. In Denmark, Finland and Åland (autonomous region in Finland) there are specific legislations regulating RA recycling in bound or unbound layers.

Plant recycling of RA in new pavements compared to landfilling gives small savings of transport work for a Swedish scenario due to the need for transferring RA forth and back from quality control and storage, but gives a significant reduction for a Danish scenario as a result of savings of imported high quality virgin aggregates. Recycling in unbound applications increases the transport work due to the need for transferring RA forth and back from a processing site prior to recycling, and the need for replacing the bitumen binder (Swedish unbound scenario) and imported high quality crushed rock (Danish unbound scenario). In-place recycling can drastically (by about 90 %) reduce the transport work both in unbound and bound applications. In the Danish unbound scenario though, high quality imported aggregates are lost in backfilling and need to be replaced, which limits the reduction to about 25 %. To keep down transports, it is important that several recycling sites are available at short distance.

Pollution and emissions

The former use of coal tar in asphalt products is a prime environmental and health concern, and bituminous mixtures with high tar contamination are classified as hazardous waste, *EWC 170303**. The phase-out of using tar in paving asphalts differs from country to country, but was ended in the mid-seventies. New asphalt layers have since then been put on top of the old layers resulting in older tar contaminated layers being superimposed by new uncontaminated asphalt layers. In this study, it is estimated that there are 14-22 Mtonnes of tar contaminated asphalt pavements in the Swedish state and municipal road networks alone.

A number of old and new additives and raw material used in the asphalt pavement production, or possible external contamination sources, could contribute to contamination. Potential emission of these has not been studied in this project due to: very limited use and very small fraction of recycled RA, expected low release due to immobilisation or lack of data on content and release. A more extensive use of additives and recycling of roofing felts could trigger the need for assessment, depending on the likely recycling routes.

Limit or guideline values for distinguishing between non-hazardous bituminous mixtures (*EWC 170302*) and hazardous mixtures containing coal tar (*EWC 170303**) vary between countries and

stakeholders. Regulated limit values on handling of contaminated non-hazardous RA are only provided by the Finnish region Åland where:

- PAH-16 content < 70 mg/kg is defined as uncontaminated without restrictions on recovery operations
- PAH-16 content 70-1000 mg/kg is defined as contaminated RA and restrictions on recycling are given.

A PAH-16 content below 70-100 mg/kg is regarded as uncontaminated RA feasible for “free use” in stakeholder guidelines of the Nordic countries.

The emission assessments in this study have been performed for a service life of 100 years, while the large scale use of asphalt on roads has a history of about 60 years and large scale RA recovery has taken place for about 15 years. Development of new recovery techniques and recovery routes are likely and the estimated long term emissions are highly uncertain.

Emission pathways for recycled tar contaminated RA are related to the

- emission of *particles* during demolition, storage and surface wear of the pavement by studded tires
- emission of PAH in *fumes* during hot recycling
- emission of PAH in *surface run-off* and in *percolating water* from the construction.

The emission of PAH-16 from recycled tar contaminated RA is expected to be basically proportional to the PAH content in the asphalt. However, for leaching processes and fumes, the distribution of compounds in the emitted PAH is different than in the pavement. This is due to the fact that the more volatile PAHs are subject to leaching and fume emission while the heavy and carcinogenic PAH compounds are retained in the asphalt matrix.

Recycling scenarios where RA is recycled in the **surface course** of a road result in high emissions related to surface wear, leaching in surface run-off and emission in fumes (paved roads) or in percolating precipitation (gravel roads). In the hot in plant recycling scenario a large fraction (40 %) of the PAH-16 in RA is emitted while essentially all PAHs are emitted in hot in-place recycling or gravel road surface recycling. Recovery scenarios where RA is recycled or backfilled into **subsurface constructions** result in low emissions, about 1-2 % of the PAH-16 content. The most important emission processes for this use are particle loss during demolition (0.5-2 %) but also leaching (0.2-0.4 %). Essentially all PAHs will remain in the subsurface construction after a service life of 100 years and any environmental or health risks will depend on the land use scenario for the construction in its future use. Leaching during **temporary storage** prior to recovery is the emission process that has the lowest impact on the total emissions.

The uncertainties associated with estimated emissions are large, due to **limited data or model descriptions** of the emission process. It is noteworthy that the most important emission processes (fumes, particle release and surface run-off) are the processes where the uncertainty is the largest.

There is no updated information on the **actual occurrence of tar contamination** and the PAH-16 content in recycled RA for the Nordic countries. In general, RA recycling techniques result in mixing of contaminated layers of asphalt with unpolluted and virgin asphalt or aggregates, resulting in a more diffuse, dispersed and diluted pollution. This makes the future detection of contamination as well as safe and sound management more difficult.

If an average PAH-16 content of 500 mg/kg is assumed for tar contaminated pavements and 100 mg/kg is assumed for RA, these figures can be used to estimate the total PAH emissions from Swedish tar contaminated pavements. The yearly average emission of PAH-16 from existing tar contaminated pavements is estimated to 140-200 kg, released by percolation. If tar contaminated RA is recycled into road base layers, these emissions will increase by merely 3 kg per year due to emissions in the demolition and recycling process. However, if contaminated RA is recycled into surface layers, the **emissions will increase and accumulate every year** by 60-160 kg due to increased emission related to surface wear, fumes and surface run-off.

Current limits on the PAH-16 content for a free use of RA, 70-100 mg/kg, are high compared to the content in asphalt based on virgin raw material, < 0.6 mg/kg. Thus, they may need to be lowered to

prevent increasing emissions to water as a result of tar contaminated RA entering recycling routes where the emission factors are higher.

It should be emphasised though that the results of this study do not represent a risk assessment but merely describe the potential release/emissions and that data are associated with large uncertainties. To improve the assessment and give recommendations on adjusted and lowered limit values for free use of tar contaminated asphalt the following improvements on data sources and emission models are recommended:

- Improved data on the generation routes and recycling routes of RA.
- Improved data on the content of PAH in RA from different generation routes.
- Improved data, models and estimations on the emissions of PAH in fumes.
- Improved data, models and estimations on the emissions of PAH in surface run-off.

It should be noted that LCA based studies on RA re-use into new asphalt mixes and bound layers have shown that re-use is advantageous due to the savings of bituminous binder and the emissions associated with virgin binder production. Backfilling or recycling in unbound road layers produce significantly less environmental benefits. Cold recycling of contaminated RA into bound base layers would ensure both a safe handling and low release of PAH while obtaining the environmental benefits of RA re-use.

Administrative barriers for processing sites could counteract high grade recycling and a simplified regulatory framework (permit system) for RA recycling into bound layers is held to be beneficial from an environmental point of view.

Recovery of concrete waste (EWC 170101)

Concrete waste is recovered to a large extent in the Nordic countries, about 3–4 Mtonnes per year, but statistics are very uncertain. Backfilling is the most common scenario while re-use of concrete structures or elements is rare at present but may become more relevant in the future due to a larger number of buildings being designed for dismantling.

Recovery of concrete often consist of homogenisation and crushing in order to obtain a specific particle size distribution. Selected recovery scenarios evaluated and compared in this study are recovery of crushed concrete

- as aggregate in unbound layers of roads and parking areas (*e.g.* sub-base)
- as aggregate in concrete production
- as final cover material on landfills, *e.g.* as drainage layer (= backfilling)
- as filling material at the demolition site (= backfilling).

Although the concrete application is considered to be high-grade recycling, recycled aggregates from concrete rubble are insignificantly used in the production of ready-mix and precast-element concrete in the Nordic countries.

Resources saved and impact on transports

When concrete rubble is recycled as road sub-base, it replaces primary aggregates in the proportion 1:1 by volume. Up to 1.2 kg of primary aggregates are saved per kg of recycled concrete aggregate used in the sub-base (due to differences in particle density). Concrete rubble used as backfilling at demolition sites or as final cover material at landfills, also replaces virgin materials in the proportion 1:1 by volume. It is emphasised that various primary materials are used for backfilling (*e.g.* soil, natural aggregates, etc.) making it difficult to specify the saving of primary materials by weight. When recycled concrete aggregate is used to produce new concrete, 20-30 % by weight of the primary coarse aggregate is saved per cubic meter concrete produced.

To evaluate the impact on transports the amount of total transports was estimated by assessing and summing the various distances between the demolition site, the processing plant, the natural aggregate source and the landfill. The assessments were made to cover the majority of the locations in urban areas. It seems clear that the total amount of transports is decreased in most cases where concrete is recovered.

Pollution and emissions

The exposure pathways in the following steps were evaluated for the recovery scenarios; storage of concrete waste, processing and production of final RCA and a user phase of 100 years. The chemical release to soil and groundwater during the user phase was found to be the most relevant exposure pathway regarding risk of pollution for processed concrete waste. Furthermore, a comparison showed a lower release of Cr, Zn and SO_4^{2-} in the road sub-base scenario than in the final cover at landfill scenario (used as drainage layer). It is also reasonable to conclude that the leaching from the road sub-base scenario will be lower than the leaching from backfilling at the demolition site. When RCA is used in new concrete (bound use), the chemical release is expected to be comparable to that of regular concrete, provided that the engineering properties are roughly the same.

Although one scenario is in favour of another, the maximum tolerable release was not assessed. Limit values at the potential pollution source are needed in order to decide if the release is within the acceptable range.

Recovery of bricks, tiles & ceramics and mixtures of these and concrete (EWC 170102, 170103 and 170107)

Recovery of bricks, tiles and ceramics in the Nordic countries is probably substantially smaller than the recovery of concrete. Limited data from Denmark and Finland indicates a flow that is $\frac{1}{4}$ of the flow of concrete but statistics are highly uncertain.

Undamaged bricks and tiles can be re-used, typically in renovation of old constructions, but the volumes are small in the Nordic countries. At present, the most common recovery scenario is recovery of mixtures of crushed bricks, tiles, ceramics and concrete in unbound low-grade applications or as filling material. Possible recovery scenarios are crushed mixtures recovered

- as aggregate in unbound road layers (sub-base)
- as drainage material
- as tennis court sand
- as plant substrates
- as pipe trench fill material
- as filling material in noise barriers etc. (= backfilling)
- as aggregate in concrete and mortar production.

In the future, re-use may become more common. However, the bricks and tiles need to be properly cleaned before re-use because excess mortar dust can inhibit the adhesion between mortar and bricks and lead to weaker masonry, depending on the mortar composition.

Resources saved and impact on transports

For technical reasons, the possible amount of crushed bricks in aggregate mixes for demanding applications is limited and this also limits the amount of primary aggregates saved to the same extent. The impact on transports is similar as presented above for concrete waste.

Pollution and emissions

For the road sub-base scenario, a relevant emission pathway is the release to groundwater from the final sub-base. The cumulative release of arsenic, vanadium and sulphate has been calculated for this case.

For the concrete and mortar production scenario there was too limited information in the literature to study the environmental properties further.

For the backfilling scenario, the cumulative release of arsenic, vanadium and sulphate has been calculated in a similar way as for the road sub-base case.

Bricks are often mixed and contaminated by other wastes and hence content and leaching is influenced by contaminants. There is a **lack of data** concerning the waste properties and better knowledge is needed to assess this waste stream.

Recovery of track ballast (EWC 170508)

Recovered track ballast is probably a large material flow. Most of it is re-used in an in-situ ballast cleaning process through rail-mounted machines. A residual fraction not fulfilling the technical requirements for track ballast is removed. Waste resulting from demolition or excavation of rail tracks is often re-used after appropriate upgrading. It can also be crushed to desired particle size and recycled in different applications. There is very **limited data** on how much of the track ballast in fact is re-used or recycled.

Selected recovery operations that were evaluated and compared with respect to emissions and impact on transport include:

- re-use as track ballast
- recycling as aggregate in asphalt production
- recycling as aggregate in unbound applications
- backfilling

Resources saved and impact on transports

For the first three recovery operations applies that 1 kg of primary aggregates is saved per kg of re-used or recycled track ballast, respectively. In backfilling, the amount of substituted material will depend on the substituted materials' properties as compared to track ballast.

Re-use of track ballast in-situ or on site will result in the shortest transportation distances. All other recovery alternatives for track ballast will result in local/national transport of the material and can be assumed to result in comparable transport distances. There is also a remaining aggregate fraction that cannot be re-used or recycled due to unsuitability in size, physical/mechanical properties or exceeding specific pollution levels and thus may need to be landfilled.

Pollution and emissions

Railway operations are associated with the emission of inorganic (mainly metals) and organic substances (*e.g.* oils, PAH, creosote, pesticides) into the environment. The release and spread of pollutants from track ballast to the subgrade and to the underlying subsoil will depend on the nature of the pollutants – their properties with respect to solubility, degradation, sorption and mobility etc. This will strongly influence the degree to which pollutants can be anticipated to be found in the track ballast layer or leach to groundwater, soil and surface water.

The level of pollution may vary for different parts of the tracks, *e.g.* close to stations and areas for maintenance of trains as compared to free stretches. There may also be a difference in pollution content of track ballast depending of the age of trains used. Newer trains that are used nowadays are improved on vital points and this reduces emissions/pollution from train traffic. Furthermore, the period of time since the last ballast cleaning has been conducted may have an influence on the level of pollution to be found. Data on total content from Swedish and Norwegian surveys indicate that a number of metals exceed risk based limit values, and carcinogenic PAH, especially benzo(a)pyrene, seems to represent the largest pollution problem for the samples analysed.

Determination of meaningful measurements of the content and leaching of potentially contaminating substances from coarse granular materials (aggregates) is a particularly difficult task, and for used track ballast it may be further complicated by the fact that it most often consists of relatively inert coarse particles fully or partially covered by a surface layer of contamination. Ideally, an analytical method that extracts the organic and inorganic contaminants quantitatively from the surface of the particles should be applied, and the results reported as mg/m². It would then be necessary to estimate the specific surface area of track ballast aggregates based on the particle size distribution to assess the total "content" of contaminants in a given mass of track ballast. Leaching will also be surface related, and probably mainly related to the layer of contaminants on the surface. The best description of the

release would most likely be obtained by using the so-called compacted granular leaching test which is part of the technical specification TS-2 developed by CEN/TC 351. This method will account for release of contaminants from both the surface layer and from diffusion from the core body of the particles. For certain types of residual fractions of track ballast with a high content of fine materials it may be possible to assess the leaching properties using a percolation leaching test under local equilibrium conditions and perform model calculations of emissions from various scenarios as described elsewhere in the report.

For the above mentioned reasons and since **detailed data on total content and leaching is lacking**, it was not possible to estimate the release and spread of pollutants from track ballast and only a general description of emission scenarios was given.

When **re-using** track ballast it can be assumed that pollution from coarse-textured track ballast to a certain extent can be released as a result of surface-wash off. Thus subgrade, subsoil and railway embankments are sinks for all released substances. This will also apply to the subgrade and/or subsoil in unbound applications of track ballast or backfilling. When recovering track ballast in **unbound applications** or as **backfilling** material, respectively, the type of any pollution present in the track ballast will have an influence on the degree of release and spread to the surrounding environment. When **landfilling** track ballast this will also be the case. However, emissions will be limited due to the various protective measures installed at landfill sites.

When track ballast is used as **aggregate in asphalt** production organic pollutants could partly be destroyed in the hot mix production in asphalt plants, whereas metals will be present in the asphalt mix leaving the plant (provided they are not removed prior to this). Similar emission pathways as for not virgin/natural asphalt aggregate during paving and from the final pavement can be assumed, i.e. they will include emissions to air and emissions to soil and groundwater.

Overall it can be said that recovery operations that ensure re-use of track ballast directly on-site or nearby can be said to be favourable, as transportation of virgin material and thus emissions as a result of transportation and quarrying of virgin materials can be minimised. Furthermore, it can be assumed that (highly) water soluble pollution from coarse-textured track ballast to a certain extent is transferred to finer material. For that reason, any recovery operation that

- a) ensures efficient cleaning and removal of any pollutant prior to re-use of material or
- b) avoids or reduces contact to water and hence emission to soil and groundwater/surface water should be preferred if the release and spread of pollutants is to be avoided or minimized.

Moreover, if quality criteria for the re-use (or recycling) of track ballast exist and they are derived from risk assessment, they presumably prevent unacceptable contamination of soil, ground water and surface water as a result of recovery activities. If this is not the case there may be a risk of spreading pollution if track ballast is recovered.

Recovery of gypsum-based construction materials *(EWC 170802)*

Statistics on generated gypsum based waste indicate that it is a small flow compared to other material flows studied in the project, about 0.05-0.06 Mtonnes in Denmark and Norway.

Gypsum based materials are construction products where “closed loop” recycling is possible, which means that the waste is used to make the same product again as compared to recovering it for some down-cycling application. The recovery of plasterboards is relatively easy to carry out while the recovery of other gypsum-based materials is more difficult. There are large differences in recycling practises in the European Union member states and this is also reflected in the statistics for the Nordic countries. Some countries, such as Denmark and Norway report high recycling rates. For Sweden, 2010-data shows relatively low recycling rate (more recent data not available) and in Finland there is currently no market for recovered gypsum due to the generation of FGD (flue gas desulphurisation) gypsum from coal-fired power stations.

Selected recovery operations that were evaluated and compared with respect to emissions and impact on transport include

- recycling to produce gypsum powder that can be used as raw material in the manufacture of new gypsum-based products such as plasterboard and stucco,
- recycling as a raw material in the cement production where gypsum is used to modify the setting characteristics of cement,
- recycling as a nutrient and structural material in compost/agriculture soil treatment.

The most common recovery operation in Denmark is the production of new gypsum-based products, while it is the production of cement in Sweden.

Natural gypsum can be substituted by FGD gypsum and recycled C&D gypsum. This material substitution decreases the rate by which the natural gypsum reserves are exploited. This reduces the impacts on land-use and biodiversity associated with extraction processes. Furthermore, there are potential energy and pollution savings related to material substitution due to shorter transportation. It has been evaluated that the substitution of natural gypsum by recycled gypsum results in a net benefit. The net benefits are assumed to be higher if the recycled gypsum replaces the FGD gypsum, since there is a higher energy consumption associated to the manufacturing of plasterboards from synthetic gypsum due to its higher moisture content.

Resources saved and impact on transports

In general it can be assumed that 1 kg of virgin material is replaced by 1 kg of recycled material/synthetic material and it can be assumed that 1 kg natural gypsum is replaced by 1 kg of natural/recycled gypsum.

To assess the impact on transport and possible energy and pollution savings in detail, the actual geographical location of actual quarrying sites, coal-fired power plants, collection and processing sites for gypsum waste as well as manufacturing sites would have to be known. In this study, the impact on transport distances has been evaluated in relative terms, *e.g.* “Which process causes additional transport?”

Pollution and emissions

Data on pollution content of and release from gypsum-based material is very sparse. However, the release of sulphate (gypsum is calcium sulphate) is generally considered the major problem in relation to contamination of groundwater and surface water.

For all three selected recovery operations applies that:

- Dust emissions may occur during demolition and transport.
- Substances present in the gypsum powder will remain in the new gypsum product or the compost/soil improver, unless they are removed during the processing and production processes.

With respect to the production of **new plasterboards**, there may consequently be a risk of substances accumulating in the production chain, as gypsum is recycled again and again. When recycling gypsum-based materials in **cement production** or in **compost/soil improver**, there is a risk that substances are released to soil and groundwater.

The environmental impact from release of substances from **landfilled** gypsum is reduced by the installation of protective measures such as bottom liners and leachate collection systems.

The results of a recent Danish LCA for handling and treatment of gypsum waste concluded that the production of new plasterboards and the use of gypsum in the production of cement are favourable as compared to the use of gypsum waste in compost. The authors of the study concluded moreover that the environmental impacts of collection and transportation as well as use of residues from the waste gypsum have a significant impact on the results. This means that site specific conditions and transportation distances may be crucial for the assessment. The sensitivity analysis carried out, as part of the study, showed that alternatives for substitution of natural gypsum are very important (*e.g.* natural gypsum from Spain, Germany or from power plants).

Recovery of wood waste (EWC 170201)

Available data from Eurostat states that about 1 206 000 tonnes of non-hazardous wood waste from construction were generated in Denmark, Finland, Norway and Sweden in 2010. In addition, there might be wood waste included in the reported amount of mixed waste from construction, which was about 450 000 tonnes.

According to the Waste Statistics Regulation, all European member states shall report about waste generation and treatment every second year. However, the basis for reporting differs substantially, even between the Nordic countries. The Danish figures are based on data from waste received at registered treatment plants and the Finnish figures are based on waste factors. The Norwegian figures are based on a statistical survey of the municipalities which collect data on construction waste on a yearly basis. Finally, the figures from Sweden are based on a combination of waste factors and a survey of the larger construction companies.

In Denmark, Norway and Sweden the waste reported as wood waste from the construction sector is mainly incinerated with energy recovery, while in Finland parts of the wood waste from construction may go to incineration without energy recovery or to landfill. However, the present Eurostat data does not reveal if wood waste from construction is re-used or material recovered. Consequently, the Eurostat database is not adequate to following up the WFD target about 70 % recovery. Better information about waste flows from “cradle to grave” is required.

Screening LCA

Wood waste from construction and demolition activities may be divided into manufacturing wastage and demolition waste. Manufacturing wood wastage from the construction site may be handled so that contamination and weather exposure is avoided. If so, a pure wood fraction for potential use may be sorted out using a rather simple sorting process. The recycled wood from a demolition site will probably be contaminated in different ways and to a greater extent and needs more processing before use compared to manufacturing wastage. The first target for material recovery should therefore be the **manufacturing wastage**. This fraction will probably also be less contaminated with wood suffering from biological attack.

The market concerning recycled wood products was screened in this study in order to find products that may result in environmental gains. Calculations according to the common ISO standard on LCA (ISO 14044) and European standard on LCA for construction products (EN 15804), showed that wood **composite products** made of pre-consuming waste (*e.g.* manufacturing wastage) will not generate any environmental improvements compared to the substitute. Unfortunately, wood products that use cement as a binder (boards and bats) contain too much cement to be useful as substitutes from an environmental point of view. The cement in these products would generate a protection against mould growth. Two wood based products are defined here as interesting alternatives to be produced from C&D wood waste. The first is a particle board which is already produced from such waste in several countries. The second is a wooden insulation product that could be manufactured from recycled wood. The environmental gain can be achieved, when the recycled wood products substitute gypsum board and mineral wool, in such applications where these substitutions are considered to be possible.

The screening LCA performed here uses two system approaches that are applied in LCA: a) the product approach, also known as attributional LCA, and b) consequential LCA covering a complex system. Attributional LCA is very robust and only includes direct consequences, while consequential LCA also includes indirect effects. More information about different system perspectives when performing an attributional or a consequential LCA may be found in Erlandsson et al (2013).

An evaluation that accounts for indirect environmental effects when the C&D wood waste is removed from the current energy market is also performed. This evaluation assumed that the marginal fuel affected by the recovery of C&D wood waste will be forestry residual wood. The consequences of this, so-called system expansion, includes the carbon storage and the sink effect and their effects on climate change. Such carbon storage is accounted for in the national climate reporting, but there is not

full consensus on how such evaluations shall be transformed and handled from a product perspective. The calculations made follow the principle given by IPCC and are streamlined by PAS 2050 (that is also used in the Product Environmental Footprint, PEF, as suggested by DG Environment). This indirect effect results in an additional gain, illustrating that material recovery of wood as part of a construction work gives larger environmental gains, compared to the removal of forestry recedes (GROT), which is assumed to be the marginal fuel.

In conclusion, according to an attributional LCA, which has a product perspective, the use of C&D wood waste for manufacturing of particle board and insulation bats and then substituting gypsum board and mineral wool, will result in environmental improvements. According to a consequential LCA, a marginal fuel has to be defined. This marginal fuel is the fuel that will replace the current C&D wood waste as fuel source. The most environmental profitable alternative will depend on what fuel is assumed to be the marginal fuel. The selection of a likely marginal fuel will be different if using a short time perspective compared to a longer time perspective (a few decades). The two extreme alternatives are that the marginal fuel is either any fossil fuel or a bio-based fuel. The first alternative will support that wood should be used for energy recovery and the second alternative that there is a potential environmental gain in wood material recovery.

The market acceptance and other technical as non-technical barriers related to the substituting alternatives given here is not part of the study and therefore not accounted for. These issues should be investigated further in future studies.

General discussion and recommendations

With regards to the uncertainties of the current waste and recovery statistics the study has had a scenario-based approach. Various recovery scenarios have been compared in terms of resource management and spreading of pollutants. However, lack of data for material properties raised difficulties for the comparisons. This was due to waste reporting in mixed fractions and the fact that few or no leaching tests had been performed for the C&D waste of interest. It is concluded that in order to report progress regarding the target, an improvement of the statistics on the generation and recovery of waste as well as on pollution content and release is needed.

Regarding consequences for transports, on site re-use or recycling reduces transport work. Furthermore, the impact will vary among countries or even regions since it depends on what supplies there are, which resources are replaced as well as on the distance to recycling facilities. In the Nordic countries, Denmark is different to Sweden, Norway and Finland regarding the resources of high quality crushed rock, which is largely imported to Denmark, *e.g.* from the other Nordic countries.

The consequences for spreading of pollutants will also vary for different recovery operations since emissions (leaching, particles etc.) depend on how and where the waste is recovered. It is emphasised that only limited data on total content and leaching was available and that the data thus are not representative for the C&D waste covered. Furthermore, the results of this study do not represent a risk assessment, but merely describe the potential release/emissions. To get reliable information on the environmental consequences there is a need for risk based assessment. Simplification may lead to wrong conclusions.

Asphalt and track ballast are generally not included in the Nordic waste statistics. Since these materials represent large material flows with a high re-use or recycling rate, including or excluding them in the waste statistics will highly influence the opportunities of attaining the WFD target of 70 % recovery.

Suggestions for the next generation of the 70 % target

The recovery target is an important step towards increased recycling. Based on the project result the following suggestions for improvements of the target are given:

- The wastes covered and the calculation method for reporting progress: Recovery targets should be set individually for different waste types since this would favour recycling of C&D waste in general and not only for high weight materials.
- The recovery routes covered: The recovery targets should distinguish between different recovery routes. Re-use or recycling into new constructions is often favourable due to environmental benefits (*e.g.* minimised transports and savings of virgin resources) even if the recovered materials are contaminated compared to virgin raw materials. Backfilling is currently regarded equal to other recovery routes but there is a risk for “downcycling”, which means that the waste is not used in the most optimal way. Guidelines on management and environmental criteria for recycling into new constructions are important in order to promote the favourable recycling routes, but also to prevent unacceptable release of contaminants as a result of unfavourable recovery routes.

We recommend that the general target of 70 % should be transformed into material specific recovery targets for different waste types reflecting their potential benefits when they are recovered. In addition, we suggest that the Nordic countries put up national material and scenario specific recovery targets so that the most effective recycling routes are promoted and those posing the biggest environmental risk are avoided.

How can the current data be improved?

The current statistics do not allow a quantification of the effects on recovery rate or the environmental effects. In order to report progress regarding the target, an improvement of the statistics is needed. As relevant initiatives and projects are carried out, more information and data on C&D waste may become available.

We propose that a joint Nordic project is set up to specify a standardized procedure and to collect data regarding the content and leaching of contaminants from asphalt, track ballast and mixed concrete/brick waste. This would enable better assessment of potential environmental risks than currently possible.

How to promote use of recycled products in the future?

Most (if not all) of the mineral C&D waste can replace natural aggregates. A better acceptability of recycled materials enhances their economic value and thus their profit margin to natural materials. However, the recycled materials need help to compete with natural aggregates. Some ideas are presented in this report (section 9.4) that would create drivers for the application of recycled products.

1 Introduction

This chapter describes the background, the goal, the prerequisites and the methodology of the project work. It also includes the definitions of abbreviations and terms used in this report. Finally, it identifies the waste fractions studied in the project.

1.1 Background

The [Waste Framework Directive](#) (WFD) includes a target for recovery of construction and demolition waste (C&D waste) which reads

“by 2020, the preparing for re-use, recycling and other material recovery, including backfilling operations using waste to substitute other materials, of non-hazardous construction and demolition waste excluding naturally occurring material defined in category 17 05 04 in the list of waste shall be increased to a minimum of 70 % by weight.” (WFD 2008/98/EC Article 11(2)(b))

The target was added during the final negotiations of the Directive text and thus did not include any consequence analysis.

In 2014 (at the latest) the WFD target is to be evaluated and maybe reinforced. In order to provide a basis for assessing the consequences of the target, the Swedish Environmental Protection Agency launched this project. Co-funding for the project could be raised via the Nordic Council of Ministers since two of the current NCM Nordic Environmental Programme (2009-2012) priorities are

- contribute to better implementation of the EU waste framework directive and
- facilitation of synergies between policies on chemicals, products and waste for the purpose of making circular flows and recycling of waste materials safer and more efficient.

1.2 Purpose and goal

Purpose: To provide the Nordic EPAs a basis for assessing the consequences of the WFD target mentioned above in terms of resource management and spreading of pollutants. This basis will be used

- when focusing the efforts to achieve a flow of materials from the construction and demolition sector with minimal impact on the environment.
- to develop means of control for increased re-use and recycling of C&D waste.
- in further discussions with the EC, to give scientific facts to what effects the 70 % target in the directive might have in the Nordic countries.

Goal: A written report in the Nordic Council of Ministers' (NCM) ANP series of publication, which provides a basis for summarily assessing the environmental impact of the 70% target in the WFD article 11(2)(b). Contents of the report should be:

- Possible future recovery scenarios and recovery scenarios currently available for a selected set of C&D waste.
- Quantification of the resources saved and the impact on the transports involved for each waste depending on how it is recovered.
- Quantification of the potential spreading of pollutants for each recovery option.

1.3 Limits and prerequisites

The following prerequisites were set up for the project work:

- The project focuses on mineral construction and demolition waste and on wood waste. It deals with the following recovery operations: re-use, recycling and other material recovery excluding energy recovery (in accordance with the EC target). However, energy recovery is handled for wood waste.
- Hazardous waste and naturally occurring material are not studied (in accordance with the EC target).

1 Introduction

- Only impacts of the EC target regarding resource management and spreading of pollutants are handled.
- LCA is only made for wood waste, for the other wastes a “life cycle perspective” is used.
- The LCA for wood waste is a so-called screening LCA covering climate impact, acidification, excessive fertilization, ground-level ozone and energy resource management, but not toxicity for example of wood incineration waste.
- Information and data is collected from databases, literature and personal contacts and, for wood waste, from LCA.
- No laboratory tests are made within the project.

Further prerequisites are given in Decision [2011/753/EU](#) (rules and calculation methods for verifying compliance with the target) which refers to Decision [2000/532/EC](#) (list of wastes and hazardous waste) which is amended in [2001/118/EC](#), 2001/119/EC and 2001/573/EC.

1.4 Methodology

A scenario-based approach was chosen for the study. This was due to ambiguities in how to apply the definition of waste, primarily in the demolition sector, which in parts has led to shortcomings in the current statistics and as a consequence considerable uncertainty as to what material types and what quantities are/should be covered to meet the WFD recovery target.

The study has identified significant material flows and has described the effects of various recovery scenarios. Both recovery options of today and recovery options that are possible in the future have been covered. Information and documentation required for the assignment have been retrieved from databases, literature and personal contacts. No laboratory tests of materials have been made in the study.

The following work strategy has been used and the result is described in the report chapters 1-8:

- **Chapter 1 – Identification of material flows and waste types:** Material flows and waste types of importance for achieving the 70 % target set in the WFD were identified with support from the steering group of the project.
- **Chapter 2 – General Nordic legislation and guidance on C&D waste**
- **Chapters 3–7 – For each waste type:**
 - **Description and uses of the waste**
 - **Demolition techniques**
 - **Waste specific legislation and guidance**
 - **Recovery scenarios and resource savings:** A compilation was made of recovery operations currently available and recovery options that are possible in the future. The savings in primary materials resources by replacing primary materials with waste materials were identified and quantified using an appropriate functional unit. A scenario without waste recycling was described including a quantification of the consumption of material resources such management entails.
 - **Impact on transports:** The impact on transports that each scenario entails was estimated and compiled. This evaluation could be done in several ways and some alternatives are shown in the report.
 - **Pollution and emissions:** Information about the pollution content of the various wastes was compiled. This was required for the calculation of pollutants spreading. Potential changes in the quality of the waste as a result of the development of demolition techniques and production methods was identified and forecasted. Emission scenarios describing the spreading of pollutants to air, land and water in connection with various recovery scenarios was compiled. A life cycle analysis perspective was applied which included future use of the waste through renewed recovery. A scenario without waste recycling was described including the spreading of pollutants that would occur from conventional virgin construction products.
 - **Spreading of pollutants:** Spreading of pollutants from different scenarios was calculated for a selected number of substances and normalized to an appropriate functional unit.

– **Comparison between recovery scenarios and the landfill scenario**

- **Chapter 8 – Impact of the 70 % target on wood waste handling:** Energy recovery through incineration of wood waste is not included in the 70 % target. Therefore, the main differences between re-use, recycling and energy recovery were evaluated by means of collecting national data from the Nordic countries and performing a screening LCA.

The study includes information from all Nordic countries. The wood waste was studied by IVL. The remaining four institutes, SINTEF, DHI, VTT and SGI, studied all selected wastes in their country, but in order to achieve a similar description within a waste type, each institute was responsible for one or two particular waste types. This means that each institute collected information and delivered results for one waste type for all Nordic countries and gave national input/comments to the other waste types. Therefore, it cannot always be avoided, that some sections or smaller parts of the report focus on the individual institute's home country and hence national facts and data.

It is important to note that no risk assessment has been made and no LCA was performed except for wood waste. The collected data in this report is given to show the expected release in some different scenarios compared to landfilling.

1.5 Definitions

The report uses several abbreviations and terms which are explained here.

Term	Explanation
Aggregates	Granular material of natural, manufactured or recycled origin used in construction (from CEN/TC 154)
B(a)P	Benzo(a)pyrene
Backfilling	A recovery operation where suitable waste is used for reclamation purposes in excavated areas or for engineering purposes in landscaping and where the waste is a substitute for non-waste materials. (from 2011/753/EU). Backfilling can also be classified as any operation meeting the definition for "recovery" under the WFD, but failing to comply with the specific requirements for recycling (or for preparation for re-use). (from EC Eurostat "Guidance on the interpretation of the term backfilling")
C&D waste	Construction and demolition waste = waste corresponding to the waste codes in Chapter 17 of the Annex to Commission Decision 2000/532/EC, excluding hazardous waste and naturally occurring material as defined in Category 17 05 04 (from 2011/753/EU)
Cement	Clinker that is finely ground in the cement mill together with gypsum to control the setting, iron(II)sulphate to reduce the hexavalent chromium to the trivalent state, supplementary cementitious materials (SCM) (e.g. coal fly ash or ground granulated blast furnace slag), mineral additives/fillers (e.g. limestone)
Cement mortar	A mixture of sand (< 8 mm), cement and water where the ratio of sand to cement is around 3 by weight.
Cement paste	Cement + water + mineral additives like silica fume (< 15 % by weight of cement) + chemical admixtures (< 1.00 % by weight of cement) that either influence the workability (e.g. plasticisers) or the setting and hardening process (accelerators and/or retarders). Cement paste is the "glue" in concrete.
Clinker	The material that is produced from the rotary kiln in a cement plant
CLP	Regulation (EC) No 1272/2008 on classification, labelling and packaging of substances and mixtures

Term	Explanation
Concrete	A mixture of coarse and fine aggregates (approximately in the range of 0.12–20 mm), cement and water where the ratio of coarse and fine aggregates to cement is around 6 by weight
DOC	Dissolved organic carbon
EWC	European waste code
FGD gypsum	Flue gas desulphurisation gypsum
GROT	Forestry residues (“grenar och toppar” which means branches and tree tops)
HDF	High density fibre board
LVL	Laminated veneer lumber
Material recovery	Any recovery operation, excluding energy recovery and the reprocessing into materials which are to be used as fuel. (from 2011/753/EU)
MCCP	Medium chained chlorinated paraffins
MDF	Medium density fibre board
Natural aggregates	Aggregates from mineral sources which has been subjected to nothing more than mechanical processing (from CEN/TC 154)
NPE	Nonylphenols
OC	Organic carbon
OSB	Oriented strain board
PAH	Polycyclic aromatic hydrocarbons. PAH compounds are a class of organic compounds found in fossil fuels (crude oil and coal), fossil fuel conversion products and incomplete combustion of carbon-containing fuels. PAHs are neutral, nonpolar organic molecules that comprise two or more benzene rings arranged in various configurations. PAHs are often reported as the sum of different selected PAH compounds.
PAH-4	A group of PAH-compounds that is reported in some monitoring schemes: Benzo[b]fluoranthene, Benzo[k]fluoranthene, Benzo[a]pyrene and Indeno[1,2,3-cd]pyrene
PAH-7	A group of highly hydrophobic (heavy) PAH compounds: Chrysene, Benzo[b]fluoranthene, Benzo[k]fluoranthene, Benzo[a]pyrene, Dibenz[a,h]anthracene, Benzo[ghi]perylene and Indeno[1,2,3-cd]pyrene. All except Benzo[ghi]perylene are classified as carcinogenic PAH
PAH-12	A group of PAH-compounds that is reported in some monitoring schemes: Phenanthrene, Anthracene, Fluoranthene, Pyrene, Benzo(a)anthracene, Chrysene, Benzo(b)fluoranthene, Benzo(k)fluor anthene, Benzo(a)pyrene, Dibenso(a,h)anthra cene, Benso(g,h,i)perylene and Indeno(1,2,3-cd) pyrene
PAH-16	The most commonly analysed PAH compounds: Naphthalene, Acenaphthylene, Acenaphthene, Fluorene, Phenanthrene, Anthracene, Fluoranthene, Pyrene, Benz[a]anthracene, Chrysene, Benzo[b]fluoranthene, Benzo[k]fluoranthene, Benzo[a]pyrene, Benzo[g,h,i]perylene, Indeno[1,2,3-cd]pyrene and Dibenz[a,h]anthracene
PCB	Polychlorinated biphenyls
PM10	Particulate matter which passes through a size-selective inlet with a 50 % efficiency cut-off at 10 µm aerodynamic diameter
PMA	Polymer modified asphalt

Term	Explanation
PMB	Polymer modified binder
POC	Particulate organic carbon
Pre-consumer waste	Waste generated by manufacturers and processors. May consist of scrap, trimmings and other production residues that were never used in the consumer market
Post-consumer waste	End product that has completed its life cycle as a consumer item and will be disposed of as solid waste
Preparing for re-use	Checking, cleaning or repairing recovery operations, by which products or components of products that have become waste are prepared so that they can be re-used without any other pre-processing. (from WFD)
RA	Reclaimed asphalt
RCA	Recycled concrete aggregates
Recovery	Any operation the principal result of which is waste serving a useful purpose by replacing other materials which would otherwise have been used to fulfil a particular function, or waste being prepared to fulfil that function, in the plant or in the wider economy. Examples of recovery operations are set out in annex II of WFD (from WFD)
Recycled aggregates	Aggregates resulting from the processing of inorganic material previously used in construction (from CEN/TC 154)
Recycling	A recovery operation by which waste are reprocessed into products, materials or substances whether for the original or other purposes (from WFD)
Re-use	Any operation by which products or components that are not waste are used again for the same purpose for which they were conceived (from WFD)
RMC	Ready mix concrete. Is manufactured in a factory or batching plant, according to a set recipe, and then delivered to a work site.
SCCP	Short chained chlorinated paraffins
TOC	Total organic carbon
WFD	Directive 2008/98/EC on waste (Waste Framework Directive)

1.6 Identification of material flows and waste types

C&D waste has been identified as a priority waste stream for re-use and recycling because of the large amounts of waste generated. Since the production of construction materials to a large extent relies on natural resources, improved management of C&D waste would contribute to the effective and efficient use of natural resources. For this reason, the Waste Framework Directive (WFD) requires the Member States of the European Union to take the necessary measures to achieve a minimum of 70 % (by weight) re-use, recycling and other material recovery (including backfilling) of non-hazardous construction and demolition waste by 2020.

Selection of waste types for this study

Based on the rules¹ for verifying compliance with the WFD target for recovery of C&D waste, the following waste types were selected for further study in this project.

- Concrete (EWC 170101)
- Bricks, tiles & ceramics and mixtures of these and concrete (EWC 170102-170103, 170107)

¹ which are given in annex III of Decision 2011/753/EU (see [Annex B](#) of this report)

1 Introduction

- Wood (EWC 170201)
- Bituminous mixtures not containing coal tar (EWC 170302)
- Track ballast (EWC 170508)
- Gypsum-based construction materials (EWC 170802)

The selection represents waste streams of importance to fulfil the WFD target, which is weight based. No glass, plastic, metals or insulation materials were studied and neither were wastes from the mechanical treatment of C&D waste. Furthermore, no mixed wastes other than mixtures of concrete, bricks, tiles and ceramics were studied.

As stated in the WFD text, no waste containing dangerous substances is covered by the target.

Replacement of natural aggregates

Most waste types studied in this report can to some degree substitute natural aggregates (wood and gypsum-based construction materials being exceptions). The most common natural aggregates of mineral origin are sand, gravel and crushed rock.

Aggregates are both an end-product in themselves such as railway ballast and a raw material used in the manufacture of other vital construction products such as ready-mixed concrete (made of 80 % aggregates), pre-cast products, asphalt (made of 95 % aggregates), lime and cement.

Aggregates are produced from natural sources extracted from quarries and gravel pits and in some countries from sea-dredged materials (marine aggregates). Secondary aggregates are usually residues from other industrial processes, like blast furnace slags or incinerator ash.

Recycled aggregates derive from reprocessing materials previously used in construction, including construction and demolition residues. According to the European Aggregates Association (UEPG), recycled aggregates from C&D waste currently account for only 6 % of the consumption of aggregates (Figure 1-1).

Figure 1-1 Use of aggregates (UEPG, 2013a)

Table 1-1 provides estimates for aggregate production in Denmark, Finland, Norway and Sweden. As can be seen the production of recycled aggregates accounts only for a very small proportion in Denmark, Finland and Sweden, respectively, whereas no data is given for Norway.

Table 1-1 Estimates of aggregates production data 2011 (after UEPG, 2013b).

	Denmark	Finland	Norway	Sweden
Total number of producers	400	430	844	985
Total number of extractions sites	500	2500	949	1575
Sand & Gravel extracted	31	36	13	17
Crushed rock produced	0	53	64	58
Marine aggregates produced	13	0	0	0
Recycled aggregates produced	6	1	0	1
Manufactured aggregates produced ^a	2	0	0	6
Total production (<i>million tonnes</i>):	51	89	77	81

^a e.g. slag, fly ash

2 General Nordic legislation and guidance on construction and demolition waste

Nordic waste legislation is largely based on EU legislation and the 70 % target in the EU waste directive will be adapted to the Nordic waste decrees.

2.1 Denmark

The overall Danish rules for recycling of C&D waste are laid down in *Statutory Order no. 1309/2012 on waste*. C&D waste has to be sorted on site and sorting shall be carried out into the following ten waste fractions:

1. Natural stone, *e.g.* granite and flint
2. Non-glazed tiles (bricks and roof tiles)
3. Concrete
4. Mixtures of stone materials, and non-glazed tiles and concrete
5. Iron and metal
6. Gypsum
7. Stone-wool
8. Soil
9. Asphalt
10. Mixture of concrete and asphalt

When sorting at the source of fractions 1 to 4 it must be ensured that all other than mortar and possibly reinforcement iron has been removed and handled separately, and that PCB containing sealant materials have been identified, removed and handled separately. Double-glazed windows have to be removed and handled separately (re-used, prepared for re-use or recycled, and if that is not possible destroyed or landfilled). Sorting on-site may be omitted if the total amount of waste from the construction and demolition operation in question is less than 1 tonne or if the physical conditions render on-site sorting impossible. In those cases sorting may take place at a stationary sorting facility approved by the municipality.

The local councils shall lay down regulations for handling municipal C&D waste and through inspections ensure compliance with the Statutory Order.

Spreading or burying waste requires a specific permit under the Environmental Protection Act. This also applies to C&D waste and other types of aggregates. However, some fractions of unpolluted C&D waste and certain other types of aggregates can be used as a substitute for primary raw materials without a specific permit under the Environmental Protection Act – provided the requirements of Statutory Order no. 1662 of 21 December 2010 are met.

The use of C&D waste for construction purposes is regulated by *Statutory Order no. 1662/2010 on recycling of residual products and soil in building and construction work and on recycling of sorted, unpolluted C&D waste*. The use requires that the waste is sorted into fractions 1–7 listed above. It also requires that the C&D waste is “uncontaminated”. In the Statutory Order, uncontaminated C&D waste is defined as C&D waste for which it with a high degree of certainty can be assumed that the waste does not contain polluting materials or substances to such an extent or of such a nature and concentration that the use of the waste may have an adverse impact on the environment or human health. The waste must hence not contain polluting substances, including substances that can cause contaminating release to soil or groundwater, *e.g.* impregnated wood, PCB containing sealants, tar, soot, remains of paint and lacquer.

C&D waste containing PCB is classified as either hazardous or non-hazardous waste, depending on the PCB content²:

² <http://www.pcb-guiden.dk/graensevaerdier-affald-pcb> - Grænseværdier for byggeaffald

2 General Nordic legislation and guidance on construction and demolition waste

- C&D waste containing above 50 mg PCB/kg is regarded as hazardous waste and shall be destroyed by incineration in a Hazardous waste incinerator or landfilled underground
- C&D waste containing less than 50 mg PCB/kg is regarded as non-hazardous waste. Combustible waste shall be disposed of in incinerators that are authorized to incinerate non-hazardous waste. Other waste or waste which is difficult to handle (e.g. concrete) can be landfilled.

C&D waste considered not contaminated with PCB can be re-used according to specified rules and requires notifications to authorities. The local authorities (the municipality) have the duty to assess whether the waste should be considered not contaminated with PCB. A limit value of 0.1 mg PCB/kg is currently used as a guideline value. However, the Danish authorities are working on establishing a national limit value for PCB in C&D waste.

Concentration levels of PCB are to be determined according to EN 15308. The limit values are specified as PCB-total, i.e. measurement of the seven selected congeners to be included in the analysis according to the standard, multiplied by a correction factor of 5.

2.2 Finland

The general Finnish rules for handling of waste are laid down in the *Finnish Government Decree 179/2012 on waste*. The Decree states that the holder of C&D waste must organise separate collection of waste so that the highest possible proportion of the waste can be prepared for re-use, or otherwise recycled or recovered. Separate collection of C&D waste must be organised at least for the following waste types:

- Concrete, brick, mineral tile and ceramic waste;
- Gypsum-based waste;
- Non-impregnated wood waste;
- Metal waste;
- Glass waste;
- Plastic waste;
- Paper and cardboard waste;
- Soil and waste rock material.

The use of C&D waste for some earth construction is regulated by the *Finnish Government Decree 591/2006 (rev. 403/2009)* given for specific waste streams, e.g. concrete chippings (see further in Chapter 4). The purpose of the decree is to promote, through simplification of the environmental permit system, the use of selected wastes in earth construction fulfilling the requirements defined in the decree. The decree is based on the protection of soil and groundwater. Limit values are given both on total content and on the release of metals and salts both for basic characterization and compliance testing (focusing only on key elements). Besides utilisation according to the decree, the local authorities can also give permission for utilisation of waste through environmental permits for other types of utilisation modes (e.g. noise barrier) especially taken in to account local conditions and material specific properties.

2.3 Norway

In Norway, specific guidelines and mandatory requirements apply in construction, rehabilitation or demolition projects with respect to waste generation and treatment. These are laid down in the Regulation on technical requirements on constructions (Byggteknisk forskrift 2010). Prior to any of the mentioned activities, e.g. demolition of buildings with available area more than 100 m² or if the project generates more than 10 metric tons of waste, a waste management plan shall be prepared. The plan, which needs to be available for inspection, includes calculations and estimates of waste volume generated, material fractions, segregation level, pre-sorting and recovery level and the quantity that needs to be landfilled. In addition, a survey shall be carried out on building elements, installations and similar that may contain hazardous components and a separate environmental restoration plan shall be prepared. Upon completion, the actual achievements in the project (waste generation, recovery level, environmental restoration plan, etc.) should be reported to Norwegian local authorities in order to

receive the certificate of practical completion. In addition, minimum 60 % (by weight) of the waste generated must be source segregated.

The Norwegian general criteria for hazardous waste are described in Avfallsforskriften chapter 11 (appendix 3) and criteria for use of polluted soils in areas with sensitive land use are formulated in Forurensningsforskriften. Although the physical and chemical properties of cement based materials are significantly different from those of soil, the same limits (total contents) are also used for concrete rubble and other slightly polluted mineral wastes, i.e. no restriction on the use for materials with concentrations below the soil criteria and prohibition of use for materials with concentrations that are above the hazardous waste limits. Between these two extremes no specific criteria exist and the fitness for use is evaluated by risk assessment on a case-by-case basis.

For waste generated from road construction, the Norwegian Public Roads Administration has issued waste management guidelines (NPRA, 2012).

2.4 Sweden

European directives have gradually been implemented in the Swedish legislation, and general rules for waste and handling of waste are laid down in the Swedish Environmental Code *Miljöbalken* (SFS 1998:808) chapter 15 and in ordinances made by the Government, e.g. Ordinance on Waste (SFS 2011:927). Management of C&D waste is mainly subject to the general waste legislation or the General rules of consideration in the Environmental Code, chapter 2. However, there are some guidelines and regulations for certain waste fractions that influence C&D waste management as well.

Swedish political control instruments are: the Waste Tax (SFS 1999:673), the Environmental Objectives³, the Swedish Waste Prevention Program (SEPA, 2013a) and the Swedish Waste Management Plan (SEPA, 2012a). The tax on waste deposited on landfill sites was introduced in the year 2000. Since then it has gradually been increased, to about € 50 (435 SEK) per tonne as from January 2006. Some waste categories are exempted, such as excavated soil that is deposited on landfill sites for inert waste. Reclaimed asphalt as well as construction and demolition waste is subject to the tax. The WFD 70 % target for recovery of C&D waste at 2020 has been implemented in the Environmental Objectives, although it is not clear if the definition of the waste types involved is the same for the Swedish target. Improved management of C&D waste has been given priority in the Waste Prevention Program and the Waste Management Plan

Landfilling of organic and combustible waste is banned in Sweden and regulations on sorting out of combustible waste are imposed in the Swedish EPA regulations and guidelines on the handling of combustible and organic waste (SEPA, 2004a). Combustible C&D waste needs not be sorted out at source if circumstances are such that sorting on site is not possible, but shall be performed of site prior to final treatment.

Waste management in the demolition of buildings is regulated in the Building code (SFS 2010:900) and guidance is given by the Swedish National Board of Housing, Building and Planning. An inventory of the generation of hazardous waste is required prior demolition of buildings. The management of both hazardous waste and non-hazardous waste should, in relevant parts, be stated in the inspection plan handed in to local authorities. The inspection plan should include information on: sorting in waste fractions, precautionary actions to prevent environmental and health risks and the final disposal of the waste. An inspection plan should also be submitted in case of renovation. There is no specific regulation on the management of waste during the construction of new buildings or for the demolition, renovation or construction of civil engineering constructions.

Guidelines on sorting of C&D waste have been published by the Swedish Construction Federation (SCF, 2013a) stating that sorting on site should be carried out into at least these nine waste fractions:

- Products and materials for re-use
- Hazardous waste

³ <http://www.miljomal.se/sv/Environmental-Objectives-Portal/>

2 General Nordic legislation and guidance on construction and demolition waste

- Electronic waste
- Wood
- Plastic for recycling
- Combustible
- Scrap and metal
- Fill material
- Waste to be landfilled (sorted on site or mixed for sorting afterwards)

Sorting in fewer fractions requires special motives. Asphalt should be sorted out for recycling purposes or as hazardous waste. Gypsum (not contaminated) should be sorted out for recycling if there is a recycling plant within reasonable distance.

Gypsum-based waste generated in construction, renovation and demolition has to be landfilled separately according to Swedish EPA regulations on landfilling, criteria and procedures for the acceptance of waste at landfills NFS 2004:10 (SEPA, 2004b). There is no definition of gypsum-based waste, no criteria on the content of for example SO_4 that triggers classification of waste as gypsum-based waste and no specific requirements on separate sorting of gypsum-based C&D waste in Swedish regulations. The guidelines of the Swedish Construction Federation recommend separate collection of plasterboards in building production (SCF, 2013b), but not in demolition (SCF, 2013c).

The management of PCB contaminated construction products is regulated in the Swedish Ordinance on PCB (SFS 2007:19) which requires identification of products in buildings and facilities. Decontamination is required prior 2016 if the PCB content exceeds 500 mg/kg. In connection with renovation or demolition, products with PCB content exceeding 50 mg/kg have to be removed.

Waste handling activities that include management, processing and recycling of C&D waste require permit from the competent authority – either licensing by court or county authority or a less extensive notification to local authorities. In the case of recovery of waste in constructions the notification process is endorsed more frequently compared to other recycling routes. (SFS 2013:251).

Permit – either licensing or notification – is also relevant in the production of new construction materials, *e.g.* asphalt plants, gravel pits or rock quarries. Recovery of waste in civil engineering requires notification in case of a minor risk of pollution of land or water and licensing in the case of more than a minor risk. The Swedish EPA has published a handbook on recovery of waste in civil engineering (SEPA, 2010). The handbook provides guidance on when the recovery activity requires notification or licensing and a procedure are put forward for assessment of environmental and health risks. Criteria when recovery can be performed without prior contact with authorities are given for the content and leaching of both hazardous and non-hazardous substances (Table 2-1). These criteria only apply to granular waste and not to products used in monolithic constructions such as asphalt courses. It should be noted though that these guidance values do not have legislative force.

Table 2-1 Guidance values for recycling of waste in civil engineering without prior notification to environmental authorities (SEPA, 2010).

Substance	Content <i>mg/kg TS</i>	Leaching (LS 0.1 l/kg) <i>mg/l</i>	Leaching (L/S 10 l/kg) <i>mg/kg</i>
As	10	0.01	0.09
Pb	20	0.05	0.2
Cd	0.2	0.01	0.02
Cu	40	0.2	0.8
Cr	40	0.2	1
Hg	0.1	0.001	0.01
Ni	35	0.1	0.4
Zn	120	1	4
Cl ⁻	-	80	130
SO ₄ ²⁻	-	70	200
PAH-L	0.6	-	-
PAH-M	2	-	-
PAH-H	0.5	-	-

Swedish Radiation Safety Authority has published a regulation on naturally occurring radioactive material (SRSA, 2011) indicating that contaminated C&D waste, such as lightweight concrete based on alum shale, can be re-used in civil engineering without restrictions due to low radioactivity (activity concentration less than 10 kBq/kg per nuclide in the uranium and thorium chains).

The Swedish Transport Administration (STA) has published technical and environmental guidelines promoting the recycling of C&D waste like asphalt and crushed concrete. Technical specifications on recycling of asphalt in bound courses are given in TRV 2011:082 (STA, 2011a). TRVK Alternativa material (STA, 2013a) describes the technical specifications for asphalt granulates, crushed concrete or blast furnace slag in unbound layers of roads. Specifications for material, construction and quality control are given in TRVKB Alternativa material (STA, 2013b) and guidelines are given in TRVR Alternativa material (STA, 2013c). At present, these documents only cover the three materials mentioned above, but may in the future be extended with sections for other alternative road materials.

STA also applies general environmental requirements on the content of hazardous substances in articles and chemicals. These requirements are not based on risk assessment but on hazard classification schemes related to the CLP⁴ regulation on chemicals. The fundament of these requirements is that materials must not contain hazardous substances that trigger labeling with hazard pictograms according to concentration limit values in the CLP, but other limit values are applied as well. The requirements are continuously updated and identical with the requirements in the BASTA⁵ system and similar with guidelines provided by other Swedish chemical hazard guideline systems such as Sunda Hus⁶). The specific criteria are outlined in the technical guidelines for chemicals TDOK 2010:310 (STA, 2010) and articles and materials TDOK 2012:22 (STA, 2012). For chemicals, an approval scheme and database on approved products is provided by the STA. For chemicals or articles and materials not fulfilling the criteria, a risk assessment and an evaluation according to the product choice principle (Environmental Code, chapter 2) has to be performed prior to use. Both TDOK 2010:310 and TDOK 2012:22 include separate ban lists on substances that must not be present in chemicals or articles and materials (Table 2-2). Excavated soil and rock for use in civil construction are regarded to require more stringent criteria and a guideline dealing specifically with these materials is under development.

⁴ Regulation (EC) No 1272/2008 on classification, labelling and packaging of substances and mixtures <http://eur-lex.europa.eu/Notice.do?mode=dbl&lang=en&ihmlang=en&lng1=en,sv&lng2=bg,cs,da,de,el,en,es,et,fi,fr,ga,hu,it,lt,lv,mt,nl,pl,pt,ro,sk,sl,sv,&val=486098:cs>

⁵ <http://www.bastaonline.se/english/bastaonline/aboutbasta.4.386979f513a1a34373978f.html>

⁶ <http://www.sundahus.se/home.aspx> and the Swedish Chemical Agency PRIO-system (http://www2.kemi.se/templates/PRIOEngframes_4144.aspx)

Table 2-2 The Swedish Transport Administration ban lists on substances in chemicals, articles and materials (STA, 2010; STA, 2012).

Ban list on substances in chemicals TDOK 2010:310		Ban list on substances in articles and materials TDOK 2012:22	
Substance	CAS	Substance	CAS
Acrylamide	79-06-1	Acrylamide	79-06-1
Anthracene	120-12-7	Anthracene	120-12-7
Benzo(a)pyrene	50-32-8	Anthracene oil	90640-80-5/ 90640-81-6/91995-15-2
Arsenic and arsenic compounds*	All	Arsenic and arsenic compounds**	All
Benzene	71-43-2	Benzene*	71-43-2
Bisphenol A	80-05-7	Bisphenol A	80-05-7
Lead and lead compounds	All	Lead and lead compounds**	All
Benzyl butyl phthalate (BBP)	85-68-7	Boric acid	10043-35-3/11113-50-
Dibutyl phthalate (DBP)	84-74-2	Benzyl butyl phthalate (BBP)	85-68-7
Bis (2-ethylhexyl)phthalate (DEHP)	117-81-7	Dibutyl phthalate (DBP)	84-74-2
		Bis (2-ethylhexyl)phthalate (DEHP)	117-81-7
		Diisobutyl phthalate (DIBP)	84-69-5
		Hexabromocyclododecane (HBCDD)	25637-99-4/3194-55-6
		Hexachlorobenzene (HCB)	118-74-1
Cadmium and cadmium compounds	All	Cadmium and cadmium compounds**	All
Short Chained Chloroparaffines (SCCP)	85535-84-8	Short Chained Chloroparaffines (SCCP)	<i>e.g.</i> 85535-84-8/ 63449-39-8
Sodium dichromate	7789-12-0	Chromium (VI) compounds	All
Mercury and mercury compounds	All	Mercury and mercury compounds**	All
		Disodium tetraborate	1303-96-4/12179-43-04-3/1330-43-4
Nonylphenol	25154-52-3	Nonylphenols	25154-52-3/84852-15-3/104-40-5/90481-04-2
4-Nonylphenoethoxylate	26027-38-3	Nonylphenoethoxylate	9016-45-9
		Tris(2-chloroethyl)phosphate	115-96-8
Perfluorooctanesulfonic acid (PFOS)	All	Pentachlorophenol (PCP)	87-86-5/131-52-2
Perfluorooctanoic acid (PFOA)	All	Perfluorooctanesulfonic acid (PFOS)	Carbon chain > C6
		Perfluorooctanoic acid (PFOA)	Carbon chain > C6
		Polybrominated diphenyl ethers (PBDE)	<i>e.g.</i> 1163-19-5/ 32534-81-9/32536-52-0
Tinorganic compounds	All	Tinorganic compounds	All
Chlorofluorocarbons (CFC)	All		
Glycol ethers	110-80-5/109-86-4/111-15-9/110-49-6		
Halon compounds	353-59-3/75-63-8/124-73-2		
Hydrochlorofluorocarbons (HCFC)	All		
Biocides*	All		

* Some uses are exempted from the ban, *e.g.* those according to Directive 2011/65/EU

** Articles and materials where the content of metals is at average natural background values are exempted from the ban.

3 Bituminous mixtures not containing coal tar – “asphalt”

3.1 Description and uses of asphalt

The three major types of asphalt products are asphalt-based paints, roofing asphalts and paving asphalts. Asphalt consists of:

- Aggregates (*e.g.* crushed rock and steel slag)
- Binder
- Additives (*e.g.* rejuvenators, anti-stripping agents)

Bitumen binder is a central ingredient in asphalt due to its adhesive force, flexibility, durability, water resistance and ability to form strong cohesive mixtures with mineral aggregates. Bitumen is the product of the non-destructive distillation of crude oil in petroleum refining. There are a variety of asphalt and bitumen qualities depending on its intended use.

3.1.1 Asphalt-based paints

Waterproofing, damp-proofing or insulation paints represent a very small minority of the use of asphalt and are not subject for recycling activities. They may however contaminate other construction wastes such as crushed concrete if not removed prior to demolition and crushing. They are thus not further considered in this chapter but may be considered as contaminants in relation to other recycled wastes.

3.1.2 Asphalt-based roofing materials

Roofing asphalt in the form of asphalt shingles or felts are made from four basic materials: fiberglass or cellulose felt backing, bitumen, mineral aggregate and mineral filler.

Asphalt shingles can be recycled into paving asphalt or new roofing asphalt (NAHAB, 1998). Roofing asphalts from building waste are predominately recycled into pavements (NAHAB, 1998; Townsend et al., 2007; Tarpaper, 2013). Tarpaper Recycling ApS has patented a recycling technology with focus on recycling into paving asphalts (Tarpaper, 2013). Facilities for recycling are located in Denmark and since 2011 recycling centres are available in Sweden as well.

The available amount of roofing asphalts in Denmark has been estimated to 40 000 tonnes per year (Tarpaper, 2012) and in western Sweden to 5000 tonnes per year (Renova, 2011). The bitumen content in felt roofing material is 5–10 times higher than in road paving asphalt. It has been estimated that recycled roofing bitumen has the potential to replace some 5–10 % of virgin bitumen used for road construction in Denmark (Tarpaper, 2012). Only sorted material, free from contamination of other building waste, is accepted. Alternative final deposition techniques are landfilling or incineration. This study evaluates the recovery of roofing asphalt as a raw material in pavement recycling.

3.1.3 Paving asphalts

Paving asphalt is a mixture of mineral aggregate, bituminous binder (usually 3-7%) and filler. The recycling of Reclaimed Asphalt (RA) pavement is an important and well established measure within road pavement production with a variety of recycling routes. RA is regularly recycled into new asphalt mixtures or used as aggregate in unbound layers.

The structure of an asphalt pavement can be divided into surface (wearing) course, binder course and base course which are overlaying an unbound sub-base layer. Design of the pavement is dependent on the traffic load, the subgrade support capacity, aggregate and binder specifications, course thickness and climate. Nordic conditions with frost, the use of de-icing agents and studded tires (in Sweden, Finland and Norway) put specific quality requirements on aggregate and binders in the surface course. Noise reduction, generation of particles (PM10) and drainage capacity of surface layers are additional upcoming requirements for high performance pavement surfaces.

Production of asphalt mixes

The production of pavements includes both the production of asphalt mixes and the laying of the asphalt. The production of asphalt mixes can be divided into four groups (DIRECT-MAT, 2013):

1. Hot mix asphalt – is produced at temperatures of 140–160 °C.
2. Warm or halfwarm mix asphalt – is produced at temperatures of 80–130 °C. This reduces the amount of energy needed and the emissions to air during production and laying. A major part of the energy needed, for drying the aggregates, is due to the evaporation water. Warm mixes produced below 100 °C are usually termed halfwarm mixes. Two types of technology processes are used for halfwarm mixes; additives (generally microcrystalline wax) or binder foaming processes.
3. Cold mix asphalt – is usually produced at ambient temperature using bitumen emulsions. The production is usually performed in place and only for base or binder course on roads with low traffic loads.
4. Hot in place recycled asphalt – is produced by specialised machines where the road surface is heated, milled and the asphalt mix is recycled in place.

The frequency and use of asphalt additives, bitumen modifiers and manufactured aggregates has varied across regions and time periods. In the past, coal tar was widely used in asphalt products, sometimes at 100 %, sometimes as a mixture with bitumen (EAPA & NAPA, 2011).

Increasing demands on the performance of asphalt pavements have induced that a number of different additives are used in the Nordic countries today (NVF, 2006). Anti-stripping agents are used to improve the adhesion between aggregate and binder in order to prevent rutting, cracking, ravelling and shoving. Two different groups of agents are used, mineral and floating agents. Rejuvenators are used to compensate for the aged and hardened binders in RA. Rejuvenators are an alternative or complement to using softer bitumen binders when recycling RA.

Non-natural manufactured aggregates, such as iron and steel making slag, is used as asphalt aggregate in many countries. The use of these is integrated in the harmonised European standard for aggregates for bituminous mixtures (EN 13043). Steel slag offers advantages such as increased adhesion, friction and durability. Fibres have been used since the 80's to enable the production of binder rich mixtures. The purpose is to prevent run at high proportion of binder.

3.2 Demolition of paving asphalt

Asphalt pavement surfaces are rehabilitated frequently (Table 3-1). In the past, new asphalt layers were often put on top of the existing layers resulting in an increasing thickness of the pavement. This procedure also lead to that older tar contaminated layers have been superimposed by new uncontaminated asphalt layers. The bound base layer of asphalt pavements are not rehabilitated frequently unless the performance of the road has deteriorated severely.

Table 3-1 Maintenance frequency of asphalt pavement surfaces in Sweden (Karlsson, 2013).

Average traffic load per day and traffic lane	Years between maintenance occasions
250	22
750	17
1500	13
3000	12
5000	11
9000	9
13000	7
13000-20000	5

Today milling is often used in maintenance and rehabilitation of asphalt pavements especially for surface layers. Milling is advantageous for RA recycling:

- It is possible to mill layer by layer and sustain the specific quality of binders and aggregates in surface, binder and base course.
- Uncontaminated layers can be separated from contaminated (*e.g.* tar contaminated).
- Milling generates an RA granulate with high homogeneity and aggregate and binder properties suitable for application in new high performance asphalt mixes.
- Prior removal of thick road marking materials by thin layer milling can improve RA quality.

Even though milling can preserve the quality of different pavement layers, some mixing of layers is often inevitable due to the rutting and variations in layer thickness that have been generated over the years.

Excavation of asphalt pavements in blocks is used when the objective is to remove the whole road at the end of its service life or to maintain subsurface installations beneath a paved surface. Excavation is more common on industrial or municipal roads and surfaces than on highways. Excavated blocks need to be crushed to RA and quality-controlled before recycling. Crushing is often performed in campaigns with mobile crushers, and temporary storage at dedicated storage sites is necessary to ensure an amount that is sufficient for treatment. An RA block is a mix of surface layers with older bound layers, and inclusion of unbound aggregates is often inevitable. The lower quality of aged bound and unbound base layers in comparison with the surface layer means that excavated RA is of lower quality and more inhomogeneous than milled RA. It also results in that old tar contaminated layers are mixed with uncontaminated layers. Statistics from Norway indicate that the amount of RA generated as blocks is twice the amount generated by milling (KFA, 2011).

An alternative to recovering asphalt pavements as RA is to mix it into the subgrade in situ by deepmilling and placing a new pavement on top. If binders are added in the process it is often referred to as “cold in place recycling” (DIRECT-MAT, 2013)

3.3 Legislation and guidance regarding asphalt recycling

Here is described the Nordic legislation and guidance specific for recycling of asphalt. If there is no specific legislation, the general waste and environmental legislation will apply. Refer to Chapter 2 for general information on the legislation on C&D waste recycling and to Annex C for European harmonized standards.

3.3.1 Legislation

In Denmark, the recycling of asphalt is dealt with by specific legislation. Asphalt waste can be used for road building purposes etc. without a specific permit under the Environmental Protection Act – provided the requirements of the Asphalt Circular of 15 July 1985 are met. A corresponding interpretive note to the circular states that crushed asphalt may be recycled without permission for use as sub-base and consolidation of the surface of roads, paths, parking lots even if they do not have an “impermeable” top cover (Danish EPA, 2010). Layers of up to 1 m are foreseen, but if the crushed asphalt is used only for sub-bases, the thickness of the layer must not exceed 50 cm. The criteria for use of asphalt are currently under review.

In Finland, a recently approved regulation (Ordinance 846/2012) simplifies the permit system for recovered old asphalt granulates to be used directly in production of new asphalt. It implies that if the recovered asphalt granulate is free from metals, oil, tar and other contamination, it can be re-used after registration and no permit is required.

In Åland, which is an autonomous province in Finland, there are special provincial laws regarding the use of waste in earth constructions. The re-use of asphalt is regulated in Landskapsförordning 79 where the asphalt free from tar is classified by < 70 mg/kg PAH-16 and tar contaminated asphalt is classified by 70-1000 mg/kg PAH-16. The use of clean asphalt in bound layers should not increase emissions that effect health or environment. The use of clean asphalt in unbound applications should

3 Bituminous mixtures not containing coal tar – “asphalt”

be restricted to 0.5 m thick base layers well above the groundwater level and with a distance to drinking water wells of at least 12 m. The use of tar contaminated asphalt is restricted to maximum 0.25 m thick bound base layers in public roads well above the groundwater level and with a distance to drinking water wells of at least 30 m. Recycling in areas sensitive for water pollution (public drinking water protection areas) is not allowed. Cold recycling techniques and an asphalt surface layer that diverts infiltration is required. Crushing should be performed shortly prior to recycling. Temporary storages of clean or tar contaminated asphalt at the recycling site should be limited (4 weeks up to 12 month if precautions are used).

In Norway, the overall management of asphalt waste from buildings and also infrastructure (e.g. roads) is covered by the technical regulation Byggt teknisk forskrift 2010. The Norwegian environment agency has declared that RA recycling sites are subject to waste legislation (KFA, 2013).

In Sweden there is no specific national legislation related to asphalt recycling.

3.3.2 Quality requirements – technical and environmental

The production of asphalt pavements is an area which is covered by harmonized European standards set up by CEN/TC 227/WG 1 Bituminous mixtures, CEN/TC 154/SC3 Bituminous bound aggregates and CEN/TC 336 Bituminous binders. The quality of reclaimed asphalt is handled in EN 13108-8 *Reclaimed asphalt* and EN 12697-42 *Bituminous mixtures – Test methods for hot mix asphalt - Part 42: Amount of foreign matter in reclaimed asphalt*.

EN 13108-8 requires the declaration of

- presence of foreign materials type 1 (cement concrete, bricks, sub-base (natural aggregates), mortar and metal) and type 2 (synthetic materials, wood and plastics).
The amount of foreign material shall be classified either by declaring the content or by specifying category F1 or F2 which corresponds to $\leq 1\%$ or $\leq 5\%$ for type 1 materials and $< 0.1\%$ for type 2 materials.
- type of binder, use of additives and technical binder performance (penetration, softening point or viscosity)

Sampling shall be carried out according to EN 932-1, where the number of samples shall be the quantity of feedstock in tonnes divided by 500 tonnes, with a minimum of five samples.

The asphalt mix homogeneity, as regards aggregate grading and binder properties, has to be tested. If addition of RA is below 10 % in surface layers and $< 20\%$ in base layers, test frequency should be once per 200 tonnes and once per feedstock.

In Denmark, technical guidelines on the use of reclaimed asphalt in unbound application have been published by the Danish Road Directorate (DRD, 2011).

In Finland, technical guidelines for asphalt pavements also cover the use of reclaimed asphalt to replace parts of virgin asphalt (PANK, 2011).

In Norway, the technical specifications for road construction include specifications regarding the use of reclaimed asphalt for asphalt paving (NPRA, 2011) and two categories are defined. The first is crushed asphalt, defined as the grading produced by crushing asphalt waste (blocks or milled off material). The second is recovered asphalt, defined as an unbound base layer or road surfacing where the normal bound asphalt specifications are no longer relevant. Recovered asphalt is used with or without addition of new bitumen. In hot mix production, the asphalt used for base and surface course may contain 25 % and 15 % of crushed asphalt respectively (without any specific adjustment of the asphalt mix recipes).

The Swedish Transport Administration has published technical specifications on the re-use of reclaimed asphalt for road paving (STA, 2011a) and technical specifications for recycling in unbound layers (STA, 2013a). In hot mix production, the asphalt used for binder/base and surface course may contain 30 % and 20 % of crushed asphalt respectively. Higher recycling rates can be used if asphalt

mix recipes are adjusted and the pavement performance is shown to be adequate. Guidelines on RA generation and recovery have been published for both uncontaminated (STA, 2004a) and tar contaminated pavements (STA, 2004b). Guidelines on the recovery of tar contaminated asphalt include recommendations for sampling, identification and chemical analysis of PAH. The following management routes of contaminated pavements are recommended:

- < 70 mg/kg TS PAH-16 – Regarded as uncontaminated. Recovery without restrictions.
- 70-300 mg/kg TS PAH-16 – Recovery in bound or unbound applications. In road applications, a non-contaminated asphalt wearing course should be applied. In non-road applications, placement above groundwater table and measures to prevent infiltration of precipitation must be applied.
- 300-1000 mg/kg TS PAH-16 – Recovery as above but measures to prevent leaching and/or to collect leachates during temporary storage should be applied. Sensitive areas such as water protection areas must be avoided.
- >1000 mg/kg TS PAH-16 – Site specific measures should be applied.

Similar guidelines are used by many municipalities with the exception that the recovery of tar contaminated asphalt should take place only in roads (SALAR, 2004). It should be noted that the STA guidelines on handling of tar contaminated asphalt encourage the recovery without restrictions of materials containing PAH compounds registered on STA’s generic ban lists for chemical substances and exceeding Swedish EPA total content guidelines⁷ on recovery of waste in civil engineering (see Section 2.4 and Tables 2-1 and 2-2).

3.4 Recovery operations

The recycling of Reclaimed Asphalt (RA) pavement is an important and well established measure within road pavement production with a variety of recycling routes. RA is regularly recycled into new asphalt mixtures or used as aggregate in unbound layers. Recycling into new asphalt mixtures and bound courses can be regarded as re-use since the RA is used again for the same purpose for which it was conceived. Recycling into unbound layers, however, is a material recovery technique where the function of the original asphalt binder is not utilized to the full.

Final disposal methods for contaminated RA are landfilling or incineration. In the Netherlands, where the recycling of tar contaminated asphalt is banned, thermal treatment plants have been established where the tar contaminated binder is incinerated and the cleaned aggregates are recycled (BRC, 2013).

Figures on recycling are published on a yearly basis by the European Asphalt Pavement Association EAPA (Table 3-2). The quality of the RA statistics in Table 3-2 varies from country to country and it is also noteworthy that the figures are much higher than in the national waste statistics (SEPA, 2012b; Statistics Norway, 2012). In general, no statistics on RA generation and recycling are kept and figures published by EAPA are highly uncertain as they are based on expert judgement by single persons. One reason for this is that a majority of the RA generation and recycling is not reported as waste handling. Other estimates on RA generation and recycling for the Nordic countries indicate considerably different data on RA generation and recycling, e.g. with a higher fraction of unbound recycling for Sweden and Denmark. The Swedish Transport Administration previously kept statistics on RA management which indicated a much higher degree of recycling in unbound layers (about 40 %).

In Norway, RA statistics are based on detailed monitoring by a non-profit, voluntary association, Kontrollordningen for Asfaltgjenvinning (KFA). The motive for establishing KFA was to provide documentation of the actual usage of recycled materials in order to avoid any possible environmental tax on the usage of asphalt materials. The objective for KFA is to increase, promote and document the recycling of asphalt pavement material. The KFA is financed by collecting a green fee on bitumen sold for use in the road sector. The green fee is currently about 1 US dollar per tonne bitumen. Data from KFA suggests a much higher recycling rate than given in Table 3-2 – well above 95 %. The difference between the figures from KFA and statistics in Table 3-2 is apparently due to in situ

⁷ The SEPA guidance values for total content are relevant for recovery of asphalt in unbound applications, but not in bound applications.

3 Bituminous mixtures not containing coal tar – “asphalt”

recovery of bitumen not being reported as recycling in Table 3-2. A small fraction of RA may be landfilled, but figures on landfilling seem to be very uncertain.

Table 3-2 Statistics on asphalt production and recycling for Nordic countries (EAPA, 2012).

	Denmark	Finland	Norway	Sweden
Number of asphalt plants				
Total	42	55	85	85
Mobile	3	26	11	10
Plants fitted for RA recycling	40 %	45 %	25 %	75 %
Production of asphalt (Mton)				
Hot and warm	4	5	6.7	8.1
Share with added RA	53 %	65 %	10 %	65 %
Warm-Half warm		0.69		0.6
Cold	0		0.037	0.06
Production of reclaimed asphalt, RA	0.6	1	0.726	1.1
Recycling technique				
Hot and warm	80 %		18 %	70 %
Halfwarm (warm)				5 %
Cold			0 %	5 %
Unbound	20 %		62 %	15 %
Supplementary information				
Area ^a (sq km)	42 370	304 000	305 000	410 000
Inhabitants ^a (M)	5.57	5.39	4.95	9.45
Road network ^a (Mm)	73	79	94	583
Paved roads ^a (Mm)	73	64	62	142
Number of non-hazardous landfills	42 ^b			80 ^c
Number of aggregate production sites				2000 ^d
Crushed rock				800
Number of RA-processing sites			150 ^e	

^a The World Bank, 2013

^b Danish EPA, 2012

^c Avfall Sverige, 2012

^d SGU, 2010

^e KFA, 2011.

3.4.1 Selected recovery operations for studies on emissions and impact on transports

All identified recovery scenarios, except cold recycling, were selected for further studies. Cold recycling is not frequently used (Table 3-2) and is thus excluded.

Re-use in stationary and mobile asphalt plants (to produce new asphalt)

Recycling in asphalt plants is a well-established technique and many plants are specially fitted for RA recycling (Table 3-2). Many different techniques have been developed to include RA in hot mix, warm/halfwarm mix or cold mix and it is generally subject to quality control. High quality RA has an economic value. There are often restrictions on how much of RA that can be introduced in the mix due to restrictions of both the plant technology and uncertainties related to RA binder quality and pavement performance. Today there are techniques available to compensate for quality deterioration and to recycle RA to its best use on a site-specific basis (Re-Road, 2012e). Such cases require specific actions and measures for determining RA quality and adjusting asphalt mixture recipes (NPRA, 2011; STA, 2011a).

Re-use through hot in-place recycling (to produce new asphalt)

Hot in-place recycling is performed by specialised machines where the road surface is heated, milled and the asphalt mix is recycled in place. New asphalt is added into the mix or placed on top. The pavement is recycled in a continuous in-place process and no surplus RA is generated. Added asphalt compensates the rutting of the old pavement due to wear and compaction. In some countries hot in-place recycling has been abandoned because of fume emissions and health issues during the heating of the old pavement prior milling (DIRECT-MAT, 2013).

Recycling in new unbound applications

Recycling of RA as unbound aggregate in road base layers or the base layer of other paved surfaces is a common technique that is widely in use (Table 3-2) for which a number of national guidelines are available. Recycling as unbound aggregate requires that RA is collected, stored and crushed to ensure quality control and sufficient amounts available for construction objects.

Backfilling (use as filling material)

RA is sometimes mixed with other recovered materials for backfilling purposes or used in unspecified backfilling as it is. Under this heading we include the use of RA as a surface material on temporary roads at construction sites, parking lots, road edges and gravel roads as these applications seem to be fairly widespread.

3.5 Disposal/landfilling of recovered asphalt

Decision 2003/33/EC⁸ puts restrictions on the TOC content of waste acceptable at landfills. Investigations of the TOC content of RA as a function of the bitumen binder content reveals a relationship close to 1:1 (Re-Road, 2012a). The limit value on 3 % TOC for inert waste landfills thus prevents the disposal of RA on inert landfills since binder content normally exceeds 3 % but does not prevent landfilling at non-hazardous landfills since the binder content normally falls below the 10 % limit for ban on organic waste landfilling.

3.6 Resources saved and impact on transports

The quality of reclaimed asphalt aggregate and binder is lower than that of new asphalt and has to be improved or adjusted by addition of fresh aggregate and binder before re-use in high performance pavement courses. Since recovered asphalt pavements are only a small fraction of the newly produced pavements (Table 3-2) this can be accomplished by adjusting the recipes of the new asphalt mixes. Restrictions are often put on the maximum allowable amount of RA in technical specifications for hot mix production, *e.g.* 20 % in wearing course and 30 % in binder and base course (STA, 2011a), while cold recycling is often performed at nearly 100 %.

Additives used to improve the binder performance (polymers, rubber etc.) may be preserved, at least partially, while it is reasonable to assume that anti-stripping agents and rejuvenators need to be renewed at each recycling opportunity. In Sweden nonetheless, technical specifications do not accept the use of RA in pavements with polymer modified binders, PMB (STA, 2011a).

There is no significant difference in the bulk densities of recycled and virgin raw material and it is thus reasonable to approximate that recycling of asphalt pavements into new bound courses in general saves both aggregates and binder on a 1 to 1 ratio.

In recycling of RA into unbound layers or applications, the presence of binder residues in RA might give the construction a better performance than when ordinary aggregates are used. However, this is normally not taken into account in the pavement design. The bulk densities of virgin aggregates and recycled RA are not significantly different and it is thus reasonable to assume that the technical properties of binders are lost and that recycling of RA into unbound layers only saves aggregates on a 1 to 1 ratio. The binders present in RA in unbound applications are wasted and have to be replaced with virgin binders in new asphalt pavements. A number of LCA studies have shown that recycling of asphalt into unbound layers or applications is thus clearly unfavourable compared to recycling into new bound layers (Re-Road, 2012b; Miliutenko, 2012; Wahlström et al., 2013).

The production of bitumen binders and additives is generally associated with long range transport, extensive emissions and consumptions of non-renewable resources. Production or import facilities for

⁸ Council Decision of 19 December 2002, establishing criteria and procedures for the acceptance of waste at landfills pursuant to Article 16 of and Annex II to Directive 1999/31/EC

binders and additives are few, while aggregates are often produced locally at a large number of sites. In Denmark virgin aggregates for surface and binder course are imported. On roads with very high traffic load for Nordic conditions, like in the Stockholm area, special aggregates with high performance to withstand the wear from studded tires might be used. These aggregates are produced at few sites but the consumption is very small.

3.6.1 Impact on transport

In this section, impact on road transport is discussed related to transport of aggregates, binder and additives used in the production of asphalt pavements. Transport distances may vary considerably from site to site. The aim of the transport assessment is to compare and evaluate the transport work associated with various recycling or disposal alternatives. The assessment focuses on the relative changes, *e.g.* whether the transport work increases or decreases. The total transport distance for an alternative is calculated based on the cumulative sum of all transport distances for sub-processes associated with the recycling or disposal scenario. To perform this assessment a simple model is used. The model is based on the hypothesis that the transport distance to different sites of material handling is proportional to the number of sites available. The more sites there are, the shorter the transport distance will be, *e.g.* the more landfills there are, the shorter the transport distance to a landfill will be.

A number of assumptions are made in this simplified model:

- In order to assess the increase or decrease in transport work, a figure or estimate of the relative abundance of different processing sites is needed.
- Only transports with aggregates, binder, additives and recovered asphalt are considered. Transports needed to produce or process raw materials, *e.g.* transport of fuels etc., are not considered.
- All transports are executed by trucks of comparable performance and the return or one-way frequency is equal for all transports.
- Overseas transports are not considered in the calculations. Transportation by boat is much more efficient than transport by trucks.
- The relative transport distance r for a handling process site can be calculated as $r \propto \frac{1}{\sqrt{n_p}}$ where n_p is the number of production sites and delivery sites for a process. The average transport distance for a handling process will be shorter the more processing sites there are available for a process.
- Recovered waste from construction and demolition needs to be transported to a processing site for preparation, quality control and temporary storage.
- Non-recycled waste is disposed of at non-hazardous landfills.
- For aggregates, the transport distances from demolition sites and to construction sites are assumed to be equal. This assumption is in favour of recycling since it is reasonable to assume that the numbers of new construction sites are more abundant than the numbers of demolition sites and the model is based on the hypothesis that the transport distance to different sites of material handling is proportional to the number of sites.
- It is assumed that backfilling sites are as abundant as construction sites.
- The lower or higher density of some secondary waste aggregates is not taken into account. Lower density can save some transport since a larger volume can be filled on trucks and since reduced load on the trucks reduces fuel consumption. These potential savings are assumed to be of minor importance.

In Sweden, where rock quarries are ample, asphalt plants are generally located at or close to an aggregate production site and the transport distances for aggregates to the plants are negligible. This is usually also the case for mobile plants. Surplus of fillers and high quality aggregates is sometimes needed from outside the quarry. An estimation of the relative road transport distances is given in Table 3-3. The relative number of processing sites has been estimated for Sweden based on information in Table 3-2. It is assumed that these estimations of the relative number of processing sites are relevant also for Norway and Finland where crushed rock aggregate quarries are abundant.

Table 3-3 Swedish scenario – Approximate number of handling sites and calculated relative road transport in asphalt pavement recycling.

Product and process		Fraction of construction material by weight (f)	Relative number of processing sites (n_p)	Relative transport distance to processing site ^a (r)	Weighted relative transport distance ^b
Quarry aggregate unbound application ^c	QA	100 %	1000	0.032	3.2
High quality aggregates and filler production ^d	HAQ	5 %	100	0.1	0.5
Bituminous binder depot/production ^e	BB	5 %	10	0	2
Bitumen modifier depot/production		0.4 %	10	0	0.1
Mineral anti-strip depot/production	MA	1 %	10	0.32	0.3
Amino anti-strip depot/production		0.02 %	1	1	0.02
Asphalt mix plant	AP	100 %	100	0.1	10
RA crushing and storage site ^f	RAP	100 %	200	0.1	7
Non-hazardous landfill	L	100 %	100	0.1	10

^a The relative transport distance r for a handling process site is calculated as $r=1/\sqrt{(n_p)}$.

^b The weighted relative transport distance is calculated as $r \times f$.

^c Reference for unbound constructions

^d Asphalt plants are assumed to be located at aggregate production sites and only transport of a small amount (5 %) of surplus filler or high quality aggregate is assumed.

^e There are about ten bitumen binder depots located along the Swedish coast supplied by boat shipments.

^f The number of RA crushing and storage sites is assumed to be twice the number of asphalt plants, based on the data for Norway in Table 3-2.

Denmark however, imports crushed rock aggregates from Sweden or Norway and it is delivered by boat. The scenario used for Sweden is thus not relevant and a scenario for Denmark is given in Table 3-4. Imported high quality crushed aggregates are mostly used in surface and binder courses while domestic low quality aggregates are used in bound bases. The import of aggregates for asphalt pavements has been estimated to 55 % of the total aggregate consumption in asphalt production (Helk, 2013).

Table 3-4 Danish scenario - Approximate number of handling sites and calculated relative road transport in asphalt pavement recycling.

Product and process		Fraction of construction material by weight (<i>f</i>)	Relative number of processing sites (<i>n_p</i>)	Relative transport distance to processing site ^a (<i>r</i>)	Weighted relative transport distance ^b
Quarry aggregate low quality unbound application ^c	QA	100 %	100	0.1	10
Quarry aggregate low quality asphalt layers ^d	QALq	40 %	100	0.1	4.5
Quarry aggregate high quality asphalt layers ^e	QAHq	50 %	10	0.3	17
High quality aggregates and filler production ^f	HAQ	5 %	10	0.3	1.6
Bitumen binder depot/production	BB	5 %	5	0.4	2.2
Bitumen modifier depot/production		0.4 %	5	0.4	0.2
Mineral anti-strip depot/production	MA	1 %	10	0.3	0.3
Amino anti-strip depot/production		0.02 %	1	1.0	0.02
Asphalt mix plant	AP	100 %	40	0.2	16
RA crushing and storage	RAP	100 %	80	0.1	11
Non-hazardous landfill	L	100 %	40	0.2	16

^a The relative transport distance *r* for a handling process site is calculated as $r=1/\sqrt{(n_p)}$.

^b The weighted relative transport distance is calculated as $r \times f$

^c Reference for unbound constructions

^d Asphalt plants are assumed to be located at aggregate production sites and only transport of a small amount (5 %) of surplus filler or high quality aggregate is assumed.

^e There are about ten bitumen binder depots located along the Swedish coast supplied by boat shipments.

^f The number of RA crushing and storage sites is assumed to be twice the number of asphalt plants, based on the data for Norway in Table 3-2.

Calculation model for cumulative transport

The larger the calculated weighted relative road transport distances, the more important is the cumulative transport. Based on calculated weighted relative road transport distances in Tables 3-3 and 3-4 it is seen that the most important transport processes are the transport of bitumen binders, aggregates, asphalt mix or RA to production, construction or disposal sites. Transport of high quality aggregates, bitumen modifiers, filler and mineral anti-strip agents is of less importance. Transport of amino based anti-strip agents is of negligible importance.

Transport calculations on cumulative sum of all transport distances for sub-processes associated with different scenarios are based on the following assumptions.

- Bitumen modifiers are of low importance and seldom used and are not included.
- Recycling of RA into new pavements avoids the transport of binders while mineral anti-strip agents and a small amount of high quality aggregates need to be added.
- Asphalt plants in Sweden are normally located at aggregate production sites and thus there is no transport of normal aggregates to the plant while asphalt plants in Denmark are supplied by aggregates from off-site sources.

- Recycling of RA into unbound pavements induces the need of replacing wasted binders with virgin binders in new road pavements and for Danish scenarios also the replacement of imported high quality crushed rock aggregates.

In the description of cumulative transport scenarios, acronyms given in Tables 3-3 and 3-4 are used.

Reference alternative – Construction with virgin raw material

The reference alternative assumes that RA is transported to a landfill and that new constructions are built with virgin raw material. In Sweden, asphalt plants are generally located at or close to an aggregate production site and the transport distances for normal quality aggregates to the plants are negligible.

The following transport operations are identified for Swedish pavements:

- Transport of added high quality aggregates and filler to asphalt plant
- Transport of binders to asphalt plant
- Transport of mineral anti-strip agents to asphalt plant
- Transport of asphalt mix to construction site
- Transport of RA to landfill

Cumulative relative transport is $HAQ+BB+MA+AP+L$.

For Danish pavements imported high quality crushed rock and domestic low quality aggregates are transported to the plant and cumulative relative transport is $QALq+QAHq+BB+MA+AP+L$.

The following transport operations are identified for unbound applications:

- Transport of aggregates to construction site
- Transport of RA to landfill

Cumulative relative transport is $QA+L$.

It is assumed that aggregates from a quarry are used in backfilling scenarios as well as in construction scenarios.

Re-use in stationary and mobile asphalt plants (to produce new asphalt)

It is assumed that RA crushing and storage is performed at the asphalt mix plant.

The following transport operations are identified:

- Transport of RA from demolition site to asphalt plant
- Transport of added high quality aggregates and filler to asphalt plant
- Transport of mineral anti-strip agents to asphalt plant
- Transport of asphalt mix to construction site

Cumulative relative transport: $AP+HAQ+MA+AP$.

Re-use through hot in-place recycling (to produce new asphalt)

In hot in place recycling, the recycling train is located at the road site and thus no transport to the plant is needed. The following transport operations are identified:

- Transport of high quality aggregates and filler to construction site
- Transport of mineral anti-strip agents to construction site

Cumulative relative transport: $HAQ+MA$

Recycling in new unbound applications in constructions or backfilling

The number of sites where backfilling or construction is performed is assumed to be equal and thus transport work is assumed to be equal for both backfilling and use in construction. Recycling of RA into unbound pavements induces the need to replace wasted binders with virgin binders in new road pavements. The following transport operations are identified:

- Transport of RA to crushing and storage site
- Transport of RA to construction or backfilling site
- Transport of virgin bitumen binder to asphalt plant

Cumulative relative transport for the Swedish scenario is $RAP+BB+RAP$.

For Danish unbound applications imported high quality crushed rock also needs to be replaced besides wasted binders and cumulative relative transport is RAP+BB+QAHq+RAP.

Recycling in unbound backfilling applications at site

Since the technical requirements for backfilling materials are less strict than for unbound application, transport of RA to a crushing and storage site might not be needed. If both crushing (with a mobile crusher) and backfilling could be performed at the demolition site, the cumulative relative transport would be limited to the replacement of waste binders in the Swedish scenario and the replacement of binders and imported high quality crushed rock in the Danish scenario.

Summary of impact on transport

The cumulative relative transport distances for asphalt recycling are summarised in Tables 3-5 and 3-6. Cumulative absolute transport distances are also given in Tables 3-5 and 3-6 assuming an average transport distance from demolition and construction site to RA processing and storage of 25 km for Swedish scenario and 15 km for the Danish scenario. Transferring of RA to and from processing sites is critical. Recycling that can be performed at site (in place) drastically reduces the transport work both in unbound backfilling and bound applications by about 90 %. In the Danish “at site backfilling” scenario though, high quality imported aggregates are lost in backfilling and need to be replaced, which limits the reduction to about 30 %. In plant recycling of RA in new pavements, the transport work decreases by merely 10 % in the Swedish scenario due to the fact that asphalt plants normally are located at virgin aggregate quarries, which compensates transport of RA to landfill in the reference scenario. In the Danish scenario, the transport savings are estimated to be substantial – resulting in a 40 % reduction as a result of savings of high quality virgin aggregates imported from other Scandinavian countries.

Recycling of RA in unbound applications increases the transport by about 20 % in the Swedish scenario and 60 % in the Danish scenario compared to landfilling. This is due to the need of transferring RA forth and back from a quality control and storage processing site prior to recycling, and the need to replace the bitumen binder (Swedish scenario) and imported high quality crushed rock (Danish scenario) wasted in unbound applications. Reducing the transport work needed for quality control and storage by increasing the number of RA processing sites (RAP) from 200 to 300 sites would get transport work in unbound recycling equivalent to landfilling. In the Danish scenario, increasing the number of RA processing sites can not compensate the transport work associated with the loss of high quality aggregates and recycling in unbound application always results in increased transport work. This is due to the assumption that the unbound reference scenario for Denmark only consumes locally produced “low quality” aggregates while RA recycling in unbound applications partly consumes high quality aggregates.

Table 3-5 Cumulated transport distances for Swedish scenario RA recycling operations.

Option	Cumulative relative transport distance	Transport distance ^a (km)	Relative change versus reference scenario
<i>Reference production of virgin pavement ^b</i>	22	56	Ref scenario
Recycling in plant	21	52	-7 %
Recycling hot in place	0.8	2	-96 %
<i>Reference production unbound application ^b</i>	12	31	Ref scenario
Recycling in unbound application	16	39	19 %
Backfilling at site	2	4	-88 %

^a Total transport distance. Assuming an average transport distance from demolition and construction site to RA processing and storage of 25 km.

^b See Table 3-3.

Table 3-6 Cumulated transport distances for Danish scenario RA recycling operations.

Option	Cumulative relative transport distance	Transport distance ^a (km)	Relative change versus reference scenario
<i>Reference production of virgin pavement^b</i>	54	53	Ref. scenario
Recycling in plant	34	32	-38 %
Recycling hot in place	1.9	2	-96 %
<i>Reference production unbound application^b</i>	26	24	Ref. scenario
Recycling in unbound application	42	40	57 %
Backfilling at site	20	19	-30 %

^a Assuming an average transport distance from demolition and construction site to RA processing and storage of 15 km.

^b See Table 3-4.

3.7 Pollution content and release from reclaimed asphalt

The pollution content of asphalt and release of substances during management and processing will depend upon the raw material used (aggregate, binder and additives). Aggregates are usually crushed rock or gravel. The most common binder, bitumen, is refined by distillation of selected crude oils. It is a mixture of paraffinic and aromatic hydrocarbons and heterocyclic compounds containing sulphur, nitrogen, and oxygen. Bitumen is not classified as hazardous. Different additives are used to improve the performance of asphalt (see appendix D). There is a variety of asphalt and bitumen qualities depending on the intended use. A number of different additives are used in the Nordic countries today (NVF, 2006).

Pollution can also result from contamination of asphalt in the form of accidental spills, road markings or diffuse pollution from road traffic.

3.7.1 Sources of pollution

Here is discussed the intended or unintended constituents of asphalt and RA and their potential to release hazardous substances.

Fumes

A well-known health related hazard relevant for both new and recycled materials are fumes generated during the heating and laying of asphalt pavements (Montelius, 2011; NIOSH, 2000). Additives to bitumen binders can have an effect on the quantity and quality of asphalt fumes (Schindlbauer & Hödl, 1993) as well as on its hazardous properties (Lindberg et al., 2008; Väänänen et al., 2006). The former use of coal tar as an additive to form a binder for asphalt dictated PAH exposure (Burstyn & Kromhout, 2003). This has forced ban or restrictions on recycling of RA contaminated with tar. Laboratory studies have revealed that recycling of RA does not seem to effect the total emission of fumes but additives or contaminants like tar will change the quality of the fumes (Re-Road, 2012a).

Mixing and paving temperature is one of the prime parameters that affect the emission of asphalt fumes. For pure bitumen binders the emission of benzene-soluble particulate matter (BSM) (EAPA & NAPA, 2011) and PAH (Brandt et al., 2000) increases nonlinearly by a factor 2 per 12 °C. However, lower temperature dependencies have been shown for tar-contaminated binders (Hugener et al., 2007). Evident temperature dependency has been found also in occupational exposure (Cavallari et al., 2012).

The complex chemical composition of fumes makes it difficult to identify the specific component(s) responsible for adverse health effects (Montelius, 2011). The complexity of asphalt fumes exposure results in major difficulties when to evaluate asphalt quality and RA recycling in relation to bitumen fume health effects. A commonly adopted threshold limit value specific for asphalt fumes is the benzene-soluble particulate matter (BSM) at 0.5 mg/m³ time-weighted average (EAPA & NAPA, 2011). Negative health effects have been noted at 0.16 mg/m³ BSM (Montelius, 2011) where binder

additives might have contributed. Observations of acute irritation in workers from airborne and dermal exposures to asphalt fumes and aerosols and the potential for chronic health effects, including cancer, warrant continued diligence in the control of exposures.

Coal tar

In the past, coal tar has been used as binder in asphalt. The use of coal tar can be related to the development of the road sector. Early development of the road network meant a large-scale use of tar. The phase-out also differs from country to country. Coal tar has been used in paving asphalts in Sweden until 1974, in Denmark until 1975, in Finland until the 1960's and in Norway until 1960 (EAPA, 2011). In Sweden, coal tar has been used over most parts of the country and it is estimated that there are 10-16 Million tonnes of tar contaminated asphalt pavements only in the state roadwork (Lindgren & Friberg, 2009). The road network in Norway was built earlier than in Sweden and tar contaminated RA is less abundant even though tar contamination has been recorded for single objects (KFA, 2010). A mapping of the PAH content of asphalt pavements in three Norwegian cities revealed tar contamination in 3 out of 20 pavement samples (NGU, 2006). The selection of samples was biased towards older and presumed contaminated pavements and the results are thus not representative for average pavements. It has been estimated that 90 % of the paved roads in Norway were paved after the use of tar-asphalt ceased in 1965 (Jørgensen, 2013). If the selected samples in the mapping are representative for older roads it means that about 1.5 % of all Norwegian roads may be contaminated by tar in the old underlying bound layers. Monitoring of PAH content in RA from RA-processing sites (1500 samples) by a simple qualitative test⁹ has revealed the presence of tar contaminated RA in 1 out of 150 sites (KFA, 2010). No information on the frequency of tar contaminated pavements in Finland or Denmark has been found.

Coal tar and coal tar asphalt is classified as hazardous waste (EWC 170301*) and thus out of the scope for the WFD recycling target. In recycling, RA from pure bitumen based pavement layers is mixed with tar contaminated layers. There is no specific guideline in the WFD on how to differentiate RA classified as hazardous from non-hazardous RA and different guidelines related to restrictions in recycling of tar contaminated asphalt are used in Europe. The complex and variable composition of coal tars and the changes in composition due to ageing when present in old pavements make it impossible to quantify the coal tar content in asphalt samples based on chemical analysis. The presence of PAH-16 and carcinogenic PAH compounds is often used as indicators to support the hazard assessment of tar contaminated pavements. PAH-16 contents at up to 15–23 % have been estimated for coal tar (Hugener et al., 2007; KFA, 2010).

The Swedish EPA has published guidance on the hazard classification of asphalt stating that asphalt can be classified as non-hazardous if the content of PAH-16 or PAH-7 (carcinogenic PAH) is below 300 and 100 mg/kg respectively (SEPA, 2013b). A limit of 1000 mg/kg PAH-16 is used by some local Swedish authorities (SALAR, 2004) and has also been proposed by STA (STA, 2004b). In Norway, a regulatory limit of 100 mg/kg B(a)P applies (Forurensningsforeskriften) based on a harmonised European classification for B(a)P in CLP (EC nr 1272/2008, Annex VI). Due to the co-occurrence of other carcinogenic substances in coal tar a lower limit of 50 mg/kg B(a)P is applied for heavy complex coal-derived substances in the CLP (EC nr 1272/2008, Annex VI Part 3 Note M) which is in line with harmonised classifications for asphalt waste in European regulation on shipment of waste (EC 1013/2006). This limit seems to be the most appropriate limit today and is used throughout the report as a limit for hazardous waste. Based on PAH analysis of Swedish tar contaminated RA, a limit of 50 mg/kg B(a)P seems to correlate to PAH-7 at about 250 mg/kg and PAH-16 at about 1000 mg/kg. An upper limit at 1000 mg/kg PAH-16 is also used in Åland (Landskapsförordning 79) for tar contaminated asphalt that is recycled. The inclusion of more stringent ecotoxic classification rules in CLP, where M-factors are used to give highly toxic substances more weight in ecotoxic classification of mixtures, might trigger more stringent hazardous waste classification limits for PAH content. How the M-factors should be applied in case of waste classification is still uncertain though (EC, 2013).

⁹ The method used is a paper chromatography of bitumen extracts where the presence of tar is quantified by ocular examination (Ministerie van de Vlaams Gemeenschap, 2006). The detection limit has been estimated to 5 % of tar in bitumen which is equivalent with circa 600 mg/kg TS PAH-16 in the asphalt pavement (assuming that the fraction of PAH 16 in tar is 20 % and the binder content of the pavement is 6 %).

Modifiers

The mechanical properties of the binder and asphalt can be improved by addition of modifiers. Modifiers are often used in applications with high demands such as airfield runways and high volume roads or in porous asphalt and often in surface layers. There is a wide variety of polymers but Styrene-butadiene-styrene (SBS) elastomers or ethylene-vinyl-acetate (EVA) plastomers are most common in the Nordic countries. The content of polymer is normally in the range of 2-9 % of the binder. The use of polymer modifiers to the binder (PMB) or the asphalt mix (PMA) is still limited in the Nordic countries but increasing. In Finland, about 1 % of the total asphalt production contained PMB with 4-9 % SBS in 2004 (NVF, 2006). PMB are usually used in hot mix asphalt and sometimes at elevated temperatures. Work health related exposure measurements performed in Sweden did not identify any changes in exposure to fumes (total fumes) due to the use of PMB but rather as response to increased paving temperature. Laboratory testing of asphalt mixes with RA at 145 °C RA also reported small changes in relation to total fumes while significant changes in the fume quality – an increase in emissions of BSM – were recorded for mixes containing SBS (Re-Road, 2012a).

Crumbed rubber from recycled tires can be used as an alternative to PMB in asphalt mixes. A three-year project to promote the use of asphalt rubber was performed in Sweden during 2007-2009 (STA, 2013d). The content of crumbed rubber in the binder was 17-20 %. The presence of hazardous substances (PAH, aniline, aldehydes, bensotiazols) in tire rubber has risen environmental and health concerns related to fume emissions and leaching. Emission of bensotiazols in asphalt fumes (Previa, 2010) and leachates (Larsson, 2008) are typical for rubber modified asphalt. Higher emissions of hazardous substances in fumes were registered – at least partly due to elevated mixing and paving temperatures (Previa, 2010). Conclusions from the Swedish project were that exposure was well below occupational exposure limit values and the use of asphalt rubber did not cause specific actions related to work health or the environment. There is on-going research to further study the work-health related exposure to fumes from asphalt rubber (Albin, 2013). Standardisation of crumbed rubber from recycled tires is in progress (CEN/TC 366).

Commercial wax, but also polyphosphoric acid, is sometimes added to bitumen or asphalt mixtures as so-called flow improvers to reduce the asphalt mixing temperature in order to decrease energy consumption and fume emissions, but also to improve workability (Edwards, 2005). The content of wax can be up to 6 % but is normally 3 % of the binder. In the Nordic countries waxes have been used mostly in Denmark (NVF, 2006).

Naturally occurring asphalt/bitumen ("crude bitumen") can be used as modifier added at a rate of 5-10 % of the binder to stiffen the asphalt. Previously used natural Trinidad-asphalt caused work-health related problems and has been replaced with Gilsonit. Gilsonit is used in most Nordic countries, but on a limited scale (NVF, 2006).

Anti-stripping agents

Two different groups of agents are used, floating and mineral agents (NVF, 2006). Floating agents are amino based fatty acid derivatives. They are used in hot and cold mix asphalts at a rate of 0.2-0.5 % and 0.8-1.5 % of the binder respectively. Amino based agents are classified as hazardous substances due to ecotoxic, irritant and sometimes corrosive properties. When used in asphalt mixtures they are immobilised and not released significantly to the environment. When these agents were introduced in Sweden in the 80's, complaints on acute irritation in workers were recorded. Studies on work-health related exposure from asphalt fumes in Sweden have not been able to confirm significant emissions in asphalt fumes. Amino based anti-stripping agents are commonly used in the Nordic countries.

Mineral anti-stripping agents are hydrated lime or cement and sometimes fly ashes, blast furnace slag or electro arc furnace slag. Mineral agents are added at a rate of 1-2 % to the mix (ca 25 % of the binder) and also act as a filler. They are used both in hot and cold mixes. The high pH of these agents gives them irritant and sometimes corrosive properties. Leachates generated from RA can get a high pH as a result of these agents. Mineral anti-stripping agents are used frequently in Sweden and Denmark but less in Norway, Finland and Island (NVF, 2006).

Rejuvenators

Early types of rejuvenators were classified as hazardous, contained solvents and increased the emission of asphalt fumes (NVF, 2006), but nowadays there are alternatives based on petroleum or vegetable oils that are non-hazardous. Rejuvenators are added to RA at a rate of 6-15 % of the binder content (Tyllgren, 2010). Rejuvenators may still increase emissions of fumes but since the use of rejuvenators allows asphalt laying at lower temperatures, which generates a lower amount of fumes, the outcome may be a zero-sum game. Without the use of rejuvenators, a procedure of overheating RA rich and stiff mixes is sometimes used to compensate for laying problems.

Steel slag and track ballast

Steel slag as asphalt aggregate is often used on surfaces with high wear (roundabouts and high volume roads). Disadvantages are dark pavement surface and high transport costs due to the high density of steel slag. Leaching of hazardous metals, *e.g.* chromium, has been identified as a possible concern, but leaching from paved surfaces is assumed to be limited due to the immobilisation of the slag in the asphalt matrix (NVF, 2006). In the Nordic countries, steel slag has been used locally in Denmark and Sweden.

Recovered track ballast can substitute virgin aggregates in some asphalt applications, see Section 6.4.

Fibres

Fibres used in asphalt pavement are based on cellulose or, less common, fibreglass. The fibres do not have any function in the final pavement, just during laying. When these agents were introduced in Sweden in the eighties, complaints on acute irritation in workers were recorded (NVF, 2006). Studies on work-health related exposure to asphalt fumes in Sweden have not been able to confirm significant impact of fibre additives but rather an impact related to elevated laying temperatures. Fibres are commonly used as additives in binder rich mixtures, *e.g.* porous asphalt, at a rate of 3-5 % of the binder. The former use and presence of asbestos fibres in RA has been observed in other countries but not in the Nordic countries.

Sulphur

Sulphur-Extended asphalt was introduced in the early seventies due to an assumed surplus of sulphur from industrial sources, but the interest ceased in the eighties when sulphur prices increased.

Sulphur:binder mass ratios of 20:80 to 40:60 were used. When asphalt is heated during mixing and paving, emissions of S, H₂S and SO₂ will cause smell and may cause eye irritation. Temperatures below 150°C are recommended to stay below occupational threshold limits. The presence of sulphur in asphalt has proven to cause corrosion of concrete constructions (Jonsson, 2000). Sulphur-Extended asphalt may have been used in some demonstration objects in the Nordic countries.

Recycled roofing asphalt

The presence of PAH and asbestos in roofing asphalt shingles and felts has been given attention in the USA where asphalt shingle recycling is in practice (Townsend et al., 2007). Testing for asbestos is required in many states before allowing the recycler to process the material. The results for over 27 000 samples indicated asbestos at detectable levels in approximately 1.5 % of the shingle samples analysed. The low number is consistent with the fact that the use of asbestos ceased in the late seventies or early eighties, and that the typical lifetime of a shingle is 12-25 years. Many of the detections were attributed to the presence of mastic and not the asphalt shingle. Over time, the amount of asbestos contaminated shingles should decrease even further. The natural PAH content in bitumen and leakage during storing of recovered asphalt shingle has not proven to cause any concern (Townsend et al., 2007). Tar contaminated roofing asphalts have been shown to increase the emission of PAH in asphalt fumes (NIOSH, 2000). The use of biocides (copper) in asphalt shingles to prevent algae growth is noted by Townsend et al. (2007). Other compounds containing Zn and Ti are also used to prevent growth of algae and moss and B compounds (borates) are used as flame retardants in some roofing felts (Oberender, 2013). No data on the frequency and extent of asbestos, tar biocides or flame retardants as additives has been found for either virgin or recovered roofing asphalts in the Nordic countries.

Contamination by external sources

Asphalt pavements and RA can be contaminated by external sources (road markings, emissions from traffic, accidental spills) during the use of the pavement.

In Norway, studies on contamination by PCBs have been performed (NGU, 2006). These studies were initiated because of findings of PCB in isolated samples of asphalt from cities and during the reclamation of the Fornebue airfield extensive contamination by PCBs was discovered in the airfield track pavements. In the latter case, the former use of PCB containing bond coats was identified as the cause. Bond coats are applied at the old pavement surface to improve the adhesion when new pavement layers are applied. Contamination was also found in some surface layer samples and deemed associated to road markings. Subsequent surveys performed on city road pavements (NGU, 2006) revealed PCB7 contamination at 50-80 µg/kg in one out of 63 samples. Analysis of the different pavement layers of the contaminated sample identified the top layer (from year 2000) as the source of contamination but did not reproduce the content from the first analysis but rather much lower values (15 µg/kg). A duplicate sample from the same pavement showed no contamination. Traces of PCB were identified in ten other samples. The traces were below the level of quantification at 3 µg/kg for individual PCB7 compounds. No correlation between the presence of PCB and road markings was observed.

The metal content of asphalt pavements was examined in 71 pavement samples from Norwegian cities (NGU, 2006). It seemed to be related to the metal content of the asphalt aggregates and not significantly influenced by other sources such as emissions from traffic.

Road markings

Short chained and medium chained chlorinated paraffins (SCCP and MCCP) have previously been used in paints for road markings (Zarogiannis & Nwaogu, 2010; Fiedler, 2010). Efforts to replace SCCPs have started several years ago due to intensified regulatory pressures (Zarogiannis & Nwaogu, 2010; Järnberg et al., 2005). In Scandinavia, thermoplastic markings, that do not contain chlorinated paraffins, have been used rather than paints (Zarogiannis & Nwaogu, 2010). No records of test of SCCP content in RA from Nordic countries have been found, but based on the facts above, SCCP and MCCP are not assumed to be present in RA from the Nordic countries.

Elevated levels of toxic metals arsenic (As), antimony (Sb) and lead (Pb) in glass beads for road markings imported to Europe were given attention in the mid 2010's and initiated actions to limit the content. Harmonised European standards for glass beads in road markings (EN 1424 and EN 1423) now include requirements on declaration of the As, Sb and Pb content in the glass beads in either of two classes:

- Class 0: no value requested;
- Class 1: ≤ 200 ppm (mg/kg)

In Finland, a limit value has been set for As saying that no road marking glass beads with As content exceeding 200 mg/kg (dry weight) may be used. The arsenic content should be determined by an independent testing laboratory (EC, 2007). The test method is not specified in the regulation, but since there are harmonised product standards that specifies the As content it can be argued that these are normative. The Swedish Transport Administration applies weight in the evaluation of tenders based on the declared content of arsenic in glass beads for road markings.

The yearly amount of glass beads used has been estimated to 6500 tonnes in Sweden (IVL, 2007) and 3600 tonnes in Norway (Klif, 2010). In Sweden, about 85 % is used by the Swedish Transport Administration (IVL, 2007). Analysis of the metal content of Norwegian asphalt pavement samples with or without road markings did not indicate any contamination by metals induced by road markings (NGU, 2006). The content was probably related to the metal content in the asphalt aggregates and was on an average at 6,1 mg/kg for lead and 1,8 mg/kg for As. The fraction of road marking glass beads in RA can be estimated by equation 3a.

$$f_{GB-RA} = \frac{GB \times f_{PS} \times f_{remov}}{RA} \quad (3a)$$

3 Bituminous mixtures not containing coal tar – “asphalt”

where

- GB is the annual consumption of glass beads,
- f_{PS} is the fraction of added road markings and glass beads that is still present at road surfaces when they are removed,
- f_{remov} is the fraction of paved road surfaces that is removed each year,
- RA is the annual production of RA (Table 3-2).

Based on assumed f_{PS} in the order of 0.1 and f_{remove} in the order of 0.01 the fraction of glass beads in RA is estimated to be about $1 \cdot 10^{-5}$. This estimation is based on the annual RA and glass beads statistics assuming that the major part of the markings applied are lost due to abrasion during the use of the road and separate removal of the road markings¹⁰. The content of As, Sb and Pb in glass beads is very variable and maximum concentrations at 200, 1500 and 300 mg/kg respectively have been recorded (Wik, 2007; IVL, 2007). Average values are well below these values and does probably not exceed the average content in aggregates by two orders of magnitude even if imported glass beads with elevated levels of metals are used. The small fraction of glass beads from road markings included in RA will thus not have any significant influence on the content of metals from RA.

3.7.2 Substances selected for studies of release

Besides tar, there is a number of old and new additives, special raw materials or possible external contamination sources that could cause emissions of hazardous substances from asphalt pavement production. None of these are chosen for studies on release because of:

- Very limited use and thus very little effect on the overall asphalt recycling (e.g., track ballast, steel slag, modifiers, rejuvenators)
- Very small fraction of RA (e.g. pavements contaminated by road markings or external sources)
- Expected low release due to immobilisation in the asphalt (e.g. fibres and amino based anti-stripping agents)
- Lack of data on content and release (biocides and flame retardants in roofing felts).

A more extensive use of additives and recycling of roofing felts could trigger the need for assessment of effects on recycling depending on the likely recycling routes.

The metal content of asphalt pavements and RA is dictated by the aggregates used and recycling is not expected to change the emissions compared to the use of virgin rock or gravel aggregates. Alternative aggregates such as steel slag are used very infrequently and not included in the study.

PAHs

Tar that previously has been used in pavements may become activated in the recycling process. PAH is a major component of tar. The PAH content of virgin asphalt pavements is about 0.5-4 mg/kg PAH-16 (Birgisdottir, 2007; Legret et al., 2005; Re-Road, 2012c). This is well below the guideline values of 70-100 mg/kg for “not contaminated” asphalt which are used in several Nordic countries. Data from Norway also suggests that the majority of older pavements will contain less than 10 mg/kg PAH-16 (NGU, 2006). For some old pavements and RA the content does exceed 10 mg/kg (NGU, 2006; Re-Road, 2012c) even though they are not obviously contaminated by tar, probably due to a higher PAH content in older bitumen binders or external contamination.

Tar contaminated RA is generated when the whole pavement is demolished and the PAH content can vary between 100 and 5000 mg/kg TS depending on the amount of tar used in the original pavement layer and the dilution generated when contaminated layers are mixed with uncontaminated in the recycling process. Tar contaminated asphalt pavements often exceed 300 mg/kg PAH-16 while 1000 mg/kg TS is seldom exceeded (Wik, 2004). There is fewer data on the leaching of PAH and emissions of PAH in fumes.

PAH is selected for further studies of release. There are no reliable statistics available on the PAH content for different types of RA generated in the Nordic countries. It is evident though that tar has

¹⁰ There are recommendations that road markings should be removed prior recycling of asphalt pavements since they can reduce the technical performance of recycled RA (DIRECT-MAT, 2013)

been used extensively in Sweden but less in Norway. The composition of PAH compounds in Swedish RA samples is about the same in spite of large variations in the total content of PAH-16 in the samples (from 54 to 4200 mg/kg). This indicates that the source of PAH is similar for all samples and that it is probably tar contaminated pavement layers. An estimate of the PAH content in RA for Sweden and Norway is given in Table 3-7.

Table 3-7 Estimated content of PAH in Swedish and Norwegian RA.

Total content (mg/kg)	Type of RA	Fraction of yearly generated RA	
		Norway	Sweden
< 10	Surface milling ^a	35 %	40 %
< 10	Excavated blocks or milling ^{ab}	50 %	20 %
10-100	Excavated blocks or milling	12 %	25 %
100-300	Excavated blocks or milling	2 %	10 %
300-1000	Excavated Blocks or milling	1 %	5 %
Calculated average amount of PAH-16 in RA ^c		20 mg/kg	70 mg/kg

^a RA not contaminated by PAH from old or recycled tar contaminated pavement layers.

^b “Excavated blocks” includes recycling of RA by deep milling

^c The average amount is calculated based on the mid of the “Total content” interval and the fractions of uncontaminated or contaminated RA.

The emissions of PAH are expected to be proportional to the PAH content of the asphalt (see Section 3.8).

3.8 Emission scenarios and calculated release/spread of pollutants

The following emissions are handled: release of particles, release of fumes and leaching.

Release of particles

During recycling of RA, particles can be released as a result of airborne fugitive dust emissions and through spillages and losses during the production, handling and storing of RA. The amount of particles being released can vary greatly and depends on how the production, handling and storing of the RA is performed and on the actions taken to counteract the release of particles. In addition, the temporal and site specific weather conditions, such as rain fall and wind speed, also influence the emissions. The emission of fugitive dust is related to the amount of fines (silt fraction) in the material handled. The binder content of RA prevents the formation of fines and the fugitive airborne dust emissions from RA handling are expected to be negligible compared to other particle transfer routes. The low fraction of fines in RA results in less fugitive dust emissions by orders of magnitude compared to when virgin aggregates are handled.

A study on the release of larger particles settling at the roadside during milling of road surfaces showed that the release tentatively could be grouped into three different categories (Re-Road, 2012a).

- Low release of RA: 0.01 kg per m² milled surface or 0.01 % of milled RA mass. This release can be achieved with ambitious sweeping actions and at favourable weather conditions (*e.g.* low wind speed < 4 m/s)
- Medium release: 0.05 kg per m² milled surface or 0.05 % of milled RA mass. This release is normally achieved with moderate sweeping actions and at normal weather conditions (*e.g.* wind speed < 8 m/s)
- High release: 0.5 kg per m² milled surface or 0.5 % of milled RA mass. This release is achieved with failing sweeping actions or substantial involuntary or accidental spillage or conscious dumping and at extreme weather conditions (*e.g.* wind speed > 10 m/s).

During milling operations sweeping is normally fairly ambitious due to technical requirements on the milled surfaces if new pavements are applied directly or if traffic is let on to the surfaces prior application of a new pavement. A release of RA corresponding to 0.05 % of the milled RA mass (0.05

3 Bituminous mixtures not containing coal tar – “asphalt”

kg per m² milled surface) is assumed as a conservative estimate for normal milling operations of pavement surfaces. There is no data available on the spillages and losses during demolition by excavation of pavement blocks. Since there are no technical incentives to prevent loss in the case of block excavation it seems reasonable to assume that losses of RA particles can be substantial, corresponding to a release of 0.5 % of the excavated mass.

Monitoring of Swedish sites for temporary storage of tar contaminated RA has shown that spillages and losses do occur but these losses have not been quantified. At asphalt plants or permanent storage sites the loss is probably substantially less since it can be expected that continuous cleaning of surfaces prevents the build-up of losses. A loss of 0.05 % of the mass handled at temporary storage sites is assumed as a conservative estimate, while the loss at permanent storage sites is assumed to be negligible. Particle loss during paving or landfilling is assumed to be negligible.

In Sweden, Finland and Norway a substantial wear of the pavement surface is caused by the use of studded tires. The wear is proportional to the traffic load (and fraction of studded tires) and for roads with a large traffic load, 9 000 vehicles per day and lane, the wear is in the order of 0.5 mm per year. This corresponds to a release of about 1.2 kg/m², y (density of asphalt is about 2400 kg/m³). If studded tires are uncommon (as in Denmark) the wear of the surface is significantly lower and the release is estimated to 0.012 kg/m², y.

Emissions of PAH associated with particle transport are proportional to the content of PAH in the particles released. It is assumed that the composition of PAH in particles released is about the same as in the asphalt pavement. This assumption is supported by studies on the binder/aggregate proportions of released particles (Re-Road, 2012a). For recycling activities, the average yearly release has to be adjusted taking the recycling frequency into account. Average yearly PAH-emissions due to particle release are given in Table 3-8 assuming a PAH-16 content of 100 mg/kg for tar contaminated asphalt pavements. The release of particles and PAH due to wear of the paved surface by studded tires is the most important process, while release during demolition is at least two orders of magnitude lower. Studies of the availability of PAH in RA particles are few and uncertain but indicate that the long term availability is high (Roskam & Comans 2009, Re-Road 2012c).

Table 3-8 Assumed release of PAH by particle transport.

Process	Particle release ^a	PAH content ^b (mg/kg)	PAH release	Frequency ^c of process	Average yearly PAH release
Surface milling	5•10 ⁻⁴ kg/kg 0.05 kg/m ²	100	0.05 mg/kg 5 mg/m ²	1/10 years	0.005 mg/kg 0.5 mg/m ²
Block excavation	5•10 ⁻³ kg/kg 0.5 kg/m ²	100	0.5 mg/kg 50 mg/m ²	1/50 years	0.01 mg/kg 1 mg/m ²
Temporary storage site	5•10 ⁻⁴ kg/kg 0.05 kg/m ²	100	0.05 mg/kg 5 mg/m ²	1/50 years	0.001 mg/kg 0.1 mg/m ²
Permanent storage site	0	-	-	-	-
Paving	0	-	-	-	-
Pavement surface wear Sweden ^d	0.012 kg/kg 1.2 kg/m ²	100	1.2 mg/kg 120 mg/m ²	1/1 year	1.2 mg/kg 120 mg/m ²
Pavement surface wear Denmark ^d	1.2•10 ⁻⁴ kg/kg 0.012 kg/m ²	100	0.012 mg/kg 1.2 mg/m ²	1/1 year	0.012 mg/kg 1.2 mg/m ²
Landfilling	0	-	-	-	-

^a Emissions given are calculated based on recycling of a 4 cm thick layer of asphalt pavement with a density of 2400 kg/m³.

^b A PAH-16 content of 100 mg/kg is assumed for tar contaminated RA.

^c The frequency of surface milling for a traffic load of 9 000 vehicles per day and lane. Block excavation and temporary storage prior recycling is assumed to be performed at the end of the expected lifetime of the road which is assumed to be 50 years.

^d Pavement surface wear is calculated for a traffic load of 9 000 vehicles per day and lane and assumes a high frequency of studded tires in Sweden and a low frequency in Denmark.

Release of fumes

The release of PAH in fumes is highly dependent upon the recycling technique chosen and is poorly understood. Emissions to air are negligible in cold recycling or recycling in unbound applications where recycling temperatures are low. If hot recycling in an asphalt plant is used, the emissions of PAH in fumes is expected to be influenced by the amount of PAH in the RA (Re-Road, 2012a). The emissions of asphalt fumes can be allocated to the following activities:

1. from the asphalt plant dryer and mixer
2. from storage in silo
3. from transfer of asphalt to trucks
4. from loaded trucks prior to departure from plant and during transport to paving site.
5. from transfer of asphalt to paver
6. from paver hopper area
7. from paver screed area
8. from newly paved surface

Approximately 90 % of the fume emissions from the recycling activities occur at the plant (activities 1–3), the larger part of it being emitted through the plant stack. Emissions from the asphalt plant are closely related to the emissions from the burner and the combustion of fumes from the dryer/mixer. Plant technology and management are more important for these emissions than the quality of RA and raw material. Emissions of fumes during the paving activities (activities 4–8) are closely related to the quality of the raw material and mix temperature (Re-Road, 2012a). As a simplified approach and based on the results from the Re-Road project, it is assumed that the emission of PAH in fumes is proportional to the amount of PAH in the hot asphalt mix and that the fraction of PAH in total fumes (measured as TOC) is 0.1 % in the case of virgin raw material. Total emission of fumes is assumed to be about 35 mg TOC per kg asphalt (Re-Road, 2012a) which at a PAH-16 content of virgin asphalt of 5 mg/kg corresponds to about 0.035 mg/kg PAH-16 for the whole recycling process. Assuming a PAH-16 content of 100 mg/kg for tar contaminated RA corresponds to an increase of the PAH content and emissions by a factor 20 resulting in PAH-16 emissions at about 0.7 mg/kg asphalt mix or 70 mg/m² for a 4 cm thick pavement surface layer.

An alternative technique to hot in plant recycling for surface pavements is hot in-place recycling. The high temperature achieved for the pavement surface in hot in-place recycling results in massive fume generation. Occupational exposure assessment of hot in-place recycling shows a substantial increase in the exposure to bitumen fume and PAH, but no data on the emissions per mass of RA is available (Re-Road, 2012a). As a simplified approach it is assumed that PAH emission from hot in-place recycling is 100 times higher than from hot in plant recycling. This would correspond to 7 g/m² in case of a surface pavement contaminated by PAH up to the same degree as chosen for the hot in plant recycling above.

Emissions of PAH associated with fumes are assumed to be proportional to the PAH content of the pavement, but the average yearly release has to be adjusted taking the recycling frequency into account. Average yearly PAH emissions due to release of fumes are given in Table 3-9 assuming a PAH-16 content of 100 mg/kg for tar contaminated asphalt pavements. Comparing data in Table 3-9 with data for PAH emissions related to particle release in Table 3-8 indicates that release in fumes is an important pathway for emissions of PAH.

Table 3-9 Estimated release of PAH in fumes at a PAH-16 content of 100 mg/kg in asphalt.

Process ^a	PAH-16 emissions in fumes ^b	Frequency ^c of process	Average yearly PAH release
Hot in plant recycling	0.7 mg/kg 70 mg/m ²	1/10 years	0.07 mg/kg 7 mg/m ²
Hot in-place recycling	7 mg/kg 700 mg/m ²	1/10 years	7 mg/kg 70 mg/m ²

^a The whole recycling process is included (from the asphalt plant dryer and mixer, during transfer to paving site and during paving).

^b Emissions given as mg per m² are calculated based on recycling of a 4 cm thick layer of asphalt pavement with a density of 2400 kg/m³.

^c The frequency of surface renewal for a traffic load of 9 000 vehicles per day and lane.

Leaching

Two different leaching processes are relevant for the PAH emissions from RA:

- emissions associated with percolating water and
- emissions to surface run-off controlled by diffusion from the paved surface.

The hydrophobic nature of PAH compounds makes the assessment of leaching a challenge.

Standardized test methods for leaching have in general been developed for inorganic substances and are not validated for organic substances.

Test methods for percolation aim at maintaining equilibrium conditions but are very sensitive to release of hydrophobic PAH associated with particles (Re-Road, 2012c) and biodegradation. Test methods for diffusion are sensitive to adsorption of PAH on the test apparatus and biodegradation (Larsson, 2001). Evaluation of leaching data from the Re-Road project suggests that equilibrium modelling works well in explaining the release of PAH from RA if release associated with particulate organic carbon (POC) or dissolved organic carbon (DOC) in the leachate is accounted for (Wik, 2013). Equilibrium partitioning is also supported by data in a few peer review papers (Brand & De Groot, 2001; Roskam & Comans, 2009).

The percolation equilibrium partitioning model used to evaluate the Re-Road test data uses the following equation:

$$C_{\text{tot}} = C_w * (K_{\text{bit}} * f_{\text{bit}} + L/S + K_{\text{DOC}} * \text{DOC} * L/S + K_{\text{bit}} * \text{POC} * L/S) \quad (3b)$$

where

K_{bit} and K_{DOC} are partitioning constants for bitumen and DOC. It is assumed that POC is small bitumen particles.

f_{bit} is the fraction of bitumen in the pavement

C_{tot} and C_w are the concentrations of PAH in the sample and in the water phase of the leachate.

The total leached concentration is expressed as the sum of the concentration dissolved in water and bound to POC and DOC using equation 3c.

$$C_{\text{leach}} = C_w * (L/S + K_{\text{DOC}} * \text{DOC} * L/S + K_{\text{bit}} * \text{POC} * L/S) \quad (3c)$$

Data on partition constant K_{bit} is calculated by fitting experimental leaching data to equation 3b based on average bitumen content in asphalt of 6 %. Calculated partitioning constants and the composition of individual PAH compounds for tar contaminated asphalt are given in Table 3-10. The K_{bit} partitioning constant calculated for tar asphalt is about one order of magnitude lower than partition constants calculated for pure uncontaminated bitumen (Brand & De Groot, 2001) and judged to be a conservative estimate.

Table 3-10 PAH compound specific partitioning constants and ratios for tar contaminated asphalt pavements.

PAH	Log K_{DOC}^a	log K_{Bit}^b	Fraction of PAH-16 in tar contaminated asphalt ^c
Naphthalene	3.13	3.7	3.2 %
Acenaphthylene	3.34	3.9	0.3 %
Acenaphthene	3.71	4.3	2.4 %
Fluorene	3.97	4.6	4.2 %
Phenanthrene	4.29	4.9	25.6 %
Anthracene	4.47	5.1	6.3 %
Fluoranthene	4.97	5.5	20.2 %
Pyrene	4.75	5.8	14.8 %
Benz[a]anthracene	5.7	6.7	6.1 %
Chrysene	5.6	6.6	4.3 %
Benzo[b]fluoranthene	5.91	6.91	3.6 %
Benzo[k]fluoranthene	5.9	6.9	2.1 %
Benzo[a]pyrene	5.92	7.5	3.8 %
Dibenz[a,h]anthracene	6.29	7.9	0.3 %
Benzo[ghi]perylene	6.01	7.6	1.2 %
Indeno[1,2,3-cd]pyrene	6.37	8.0	1.6 %

^a $K_{DOC} = K_{OC} * 0.24$. Data on K_{OC} from Verbruggen (2012).

^b Calculated from Re-Road leaching data (Re-Road, 2012a). Organic particulate matter released in leachates from RA is assumed to be binder particles.

^c Calculated from Re-Road total content data (Re-Road, 2012a).

The PAH concentration in the leachate can be calculated using the leaching model in equations 3b and 3c and data in Table 3-10. Field data on leaching from storage piles indicates that particles released and transported by the leachate are important routes for emissions of PAH-16 (Re-Road, 2012a; Re-Road, 2012d). Based on these observations, it is assumed that POC content in leachates of 40 mg/l is reasonable when modelling the leaching from storage piles.

To calculate the emission from a construction, data on the dimensions of the construction is needed together with data on the service life and the amount of percolation water (see appendix E). Field data from storage stockpiles indicates that the percolation from RA storage piles is limited to about 10 % of net precipitation during the first four months when there is a build-up of the water content in the pile. Thereafter, the percolation increases and can reach up to 50 % of net precipitation (Norin & Strömvall, 2004). Percolation through asphalt pavements has been reported at 9-6 % of rainfall for Europe (Re-Road, 2012a). For Nordic countries, where a substantial part of the precipitation can be snow that is cleared from the road, this is assumed to be a conservative estimate. For granular materials in the sub-base of a paved road, infiltration in the road shoulders increases the percolation. An average percolation rate in the sub-base of twice the rate through the pavement is assumed reasonable. To estimate the release, an average annual precipitation of 700 mm and net precipitation of 300 mm is assumed.

The following parameters were chosen to calculate leaching emissions:

- Storage pile: height 2 m, density 1800 kg/m³, average lifetime of storage one year, calculated¹¹ percolation of 0.11 m corresponds to an L/S of 0.03 l/kg during storage for one year.
- Pavement on road: height 0.2 m for bound base layer and 0.04 m for surface layer, density 2400 kg/m³, average yearly percolation of 0.05 m/y corresponds to an L/S of 0.087 l/kg, y or 8.7 l/kg for a service life of 100 years. Leaching calculated separately for the surface layer corresponds to an L/S of 0.52 l/kg, y or 52 l/kg for a service life of 100 years.
- Sub-base in paved road: height 0.5 m, density 2200 kg/m³, average yearly percolation of 0.10 m/y corresponds to an L/S of 0.091 l/kg, y or 9.1 l/kg for a service life of 100 years.
- Backfilling: height 1 m, density 1800 kg/m³, average yearly percolation 0.3 m/y corresponds to an L/S of 0.17 l/kg, y or 17 l/kg for a service life of 100 years.

¹¹ Calculation based on data from Norin & Strömvall (2004) and net precipitation of 300 mm as (4 months*0.1*300 mm/year + 8 months*0.5*300 mm/year) / 12 months

3 Bituminous mixtures not containing coal tar – “asphalt”

Yearly PAH emissions due to leaching are given for four scenarios in Table 3-11 assuming a PAH-16 content of 100 mg/kg for tar contaminated RA. The yearly amount of leached PAH is about the same for the different scenarios, but since leaching during temporary storage is an infrequent process relevant for a limited period it will be approximately one order of magnitude lower than the leaching from constructions when distributed over the entire life cycle.

Table 3-11 Modelled yearly emissions of PAH, leached from asphalt recycling scenarios at a PAH-16 content of 100 mg/kg in asphalt.

PAH	Total content (mg/kg)	Storage pile ^b	Leaching per year (µg/kg) ^a		
			Pavement	Road sub-base	Backfilling
Naphthalene	3.2	0.37	0.86	0.90	1.66
Acenaphthylene	0.3	0.02	0.05	0.05	0.10
Acenaphthene	2.4	0.11	0.17	0.18	0.33
Fluorene	4.2	0.15	0.17	0.17	0.32
Phenanthrene	25.6	0.69	0.49	0.51	0.93
Anthracene	6.3	0.16	0.08	0.08	0.15
Fluoranthene	20.2	0.44	0.09	0.09	0.17
Pyrene	14.8	0.32	0.04	0.04	0.08
Benz[a]anthracene	6.1	0.13	0.003	0.003	0.005
Chrysene	4.3	0.09	0.002	0.002	0.004
Benzo[b]fluoranthene	3.6	0.07	0.001	0.001	0.002
Benzo[k]fluoranthene	2.1	0.04	0.001	0.001	0.001
Benzo[a]pyrene	3.8	0.08	0.0003	0.0003	0.0006
Dibenz[a,h]anthracene	0.3	0.01	0.00002	0.00002	0.00003
Benzo[ghi]perylene	1.2	0.03	0.0001	0.0001	0.0002
Indeno[1,2,3-cd]pyrene	1.6	0.03	0.0001	0.0001	0.0002
Σ PAH 16	100	2.7	1.9	2	3.7
L/S Ratio	l/kg	0.031	0.087	0.091	0.17

^a Emissions include PAH bound to DOC in the leachate. A DOC content of 5 mg/l was assumed.

^b Emissions include PAH bound to colloidal bitumen particles (POC) in the leachate. A POC content of 40 mg/l was assumed.

PAH leached from asphalt shows a distribution of PAH compounds that are markedly different from the composition in the pavement (total content). Heavy and carcinogenic PAH compounds (Chrysene, Benzo[b]fluoranthene, Benzo[k]fluoranthene, Benzo[a]pyrene, Dibenz[a,h]anthracene, Benzo[ghi]perylene) are retained in the asphalt matrix and leached in very low amounts unless there is a release of colloidal organic bitumen particles such as in the case of storage piles. Comparing leaching data in Table 3-11 with data for fume emissions in Table 3-9 and data for particle release in Table 3-8 indicates that leaching is the emission route of least importance.

Emissions to surface run-off controlled by diffusion from the paved surface can be calculated by equation 3d (Birgisdottir et al., 2007).

$$M_{\text{tot}} = 2 * \rho * C_a * f_{\text{wet}} \sqrt{t * D_e / \pi} \quad (3d)$$

where

M_{tot} is the calculated emission controlled by diffusion,

ρ is the density,

C_a is the concentration of PAH available for emission,

f_{wet} is the fraction of time when the surface of the pavement is wetted,

t is the time interval and

D_e is the effective diffusion coefficient.

Laboratory tests at 20 °C on non-contaminated asphalt pavements¹² found effective diffusion coefficients in the range of 10^{-12} to 10^{-14} m²/s (Birgisdottir et al., 2007) and tests on recycled tar

¹² PAH-16 content below 4 mg/kg

contaminated asphalt pavements¹³ found effective diffusion coefficients in the range of 10^{-15} to 10^{-16} m²/s (Larsson, 2002). Effective diffusion coefficients in the bitumen binder matrix for hydrophobic organic compounds with a molecular weight equivalent with PAH-16 (130-280 g/mol) is in the range of 10^{-14} to 10^{-13} m²/s at 20 °C (Karlsson, 2002). Effective diffusion from the surface of the matrix is expected to be substantially less than diffusion within the binder matrix, due to physical and chemical retention factors. The effective diffusion coefficients given by Birgisdottir et al. (2007) are thus deemed unrealistically high. Data from the study by Larsson (2002) suggests however that emissions of heavy and carcinogenic PAHs are controlled by equilibrium partitioning rather than diffusion. This is valid after a limited contact time (about one day) at a leachate area ratio (L/A) of 82 l/m². A surface leaching process might thus retain heavy and carcinogenic PAH-compounds more in the asphalt matrix than lighter PAH compounds equivalently with leaching associated with percolation.

The emission rate controlled by diffusion is uncertain due to lack of reliable data, but also because diffusion rates are highly dependent on environmental factors such as climate (e.g. wetted time and temperature) and pavement design. Estimated yearly PAH emissions from the pavement surface based on equation 3d are given in Table 3-12 assuming a f_{wet} of 35 % for paved roads (Birgisdottir et al., 2007) and a PAH-16 content of 100 mg/kg for tar contaminated asphalt and 90 % availability.

Table 3-12 Estimated release of PAH by diffusion from paved surface at a PAH-16 content of 100 mg/kg in asphalt.

Effective diffusion coefficient	Average yearly PAH release (mg/kg)	Average yearly PAH release ^a (mg/m ²)
10^{-15}	0.16	16
10^{-16}	0.050	5

^a Emissions given as mg/m² are calculated based on recycling of a 4 cm thick layer of asphalt pavement with a density of 2400 kg/m³.

Comparing calculated data for surface run-off (Table 3-12), leaching (Table 3-11), fume emissions (Table 3-9) and particle release (Table 3-8) indicates that diffusion from the pavement surface could be an important emission route.

The methods to estimate emissions of PAH-16 are linearly related to the content of PAH and the exposure (e.g. L/S) for all different emission pathways – particles, fumes and leaching. Emissions for different exposure scenarios are thus easy to estimate by linear extrapolation of the emissions given above, as long as the content of PAH in the exposed construction is not depleted. The density of recycled RA will vary depending on the recycling option, and in many real life recycling scenarios RA is often added as a minor fraction of the new construction (e.g. bound pavement layers). Since emission models are linear in relation to pollutant content and exposure, extrapolation will still be valid if emissions are normalised to the mass of RA recycled. For leaching and fumes it is expected that the composition of PAH compounds emitted is enriched with regard to the lighter PAH compounds, while heavier and carcinogenic PAH compounds are more retained in the asphalt matrix.

3.8.1 Backfilling of recovered asphalt

Backfilling scenarios for RA are low performance applications such as landscaping or noise barriers, sometimes at the demolition site. In these cases RA is often mixed with other low quality secondary materials and therefore it is assumed that no future monitoring or recycling is performed with RA used in backfilling scenarios.

A special type of backfilling scenario is the use of RA granulates as a surface layer on gravel roads and areas. The RA is attractive in this application since its low proportion of fines is considered to

¹³ PAH-16 content at 4000-5000 mg/kg

3 Bituminous mixtures not containing coal tar – “asphalt”

reduce dust emissions from gravel surfaces. Research on the dust suppression of RA granulates as a surface layer on gravel roads however, shows poor results (Edvardsson K, 2010).

Emissions associated with backfilling are *a)* particle emissions during demolition and storage and *b)* leaching during storage and use. Block demolition and temporary storage sites are the most probable routes prior backfilling since block demolition generates RA of lower quality than milling. Emissions for a service life of 100 years are presented in Table 3-13 and Figure 3-1.

Table 3-13 Backfilling scenario – Landscaping. Estimated release of PAH per kg recycled asphalt. Based on Table 3-8 (particle release) and Table 3-11 (leaching data). PAH-16 content in recycled asphalt pavement is 100 mg/kg. Service life is 100 years.

Backfilling scenario: Landscaping		Emission through particles (mg PAH / kg RA)		Emission through leaching (mg PAH / kg RA)	
Years	L/S	Demolition Block excavation	Storage Temporary	Storage	Use Percolation
1	0.17	0.5	0.05	0.0027	0.0037
100	17	0.5	0.05	-	0.37

Figure 3-1 Backfilling scenario – Landscaping. Cumulative release as function of time and L/S. Based on Table 3-8 (particle release) and Table 3-11 (leaching data). The release from different release mechanisms is added. Service life is 100 years.

The most important emission processes are particles released during demolition and leaching during use. Leaching during storage is the least important process. The released amount is only a small fraction of the amount in the original RA, about 1 %. Most of the PAHs will remain in the backfilled construction and any environmental or health risks will depend on the land use scenario for the backfilling in its future use.

Emissions associated with RA granulate as a surface layer on gravel roads are presented in Table 3-14 and Figure 3-2.

Table 3-14 Backfilling scenario – Granular surface layer on gravel roads.

Estimated release of PAH per kg RA. Based on Table 3-8 (particle release) and Table 3-11 (leaching data). PAH-16 content in recycled asphalt pavement is 100 mg/kg. Service life is 100 years.

Backfilling scenario: Surface layer on gravel roads		Emission through particles (mg PAH / kg RA)		Emission through leaching (mg PAH / kg RA)	
Years	L/S	Demolition Block excavation	Storage Temporary	Storage	Use Percolation
1	6.2	0.5	0.05	0.0027	0.14
100	620	0.5	0.05	0.0027	14

Figure 3-2 Backfilling scenario – Granular surface layer on gravel roads. Cumulative release as function of time and L/S. Based on Table 3-8 (particle release) and Table 3-11 (leaching data). The release from different release mechanisms is added. Service life is 100 years.

The yearly L/S for a surface layer is estimated to 6.2 l/kg based on a layer thickness of 0.05 m and an infiltration of 560 l/m² and year. The total L/S value for a service life of 100 years is 260 and the major emission processes is leaching during use. The leached amount is a substantial fraction of the amount in the original RA, about 14 %. Surface wear is an additional relevant release process for this scenario but there are no data available on the release rate of particles. It seems reasonable though to assume that a major part of the RA granulates on a gravel road could be emitted during a service life of 100 years and that essentially all of the PAHs will be released from the construction.

3.8.2 Recycling of RA in asphalt plants

Recycling of RA in stationary or mobile asphalt plants is a well-established method where RA is used in high performance applications such as bound base layers or surface layers of pavements. RA is often added as a fraction to new asphalt mixes. Asphalt in bound layers is expected to be recycled into new asphalt mixes repeatedly. Emissions are particle emissions during demolition; storage and surface wear, leaching during storage and use and finally fume emission during production and paving with hot mixes. Emissions are calculated for surface layer re-use assuming a PAH content of 100 mg/kg, demolition by surface milling and recovery at a permanent hot mix production plant. The “in use” scenario is a Swedish road with an average traffic load of 10 000 vehicles per day and traffic lane and a maintenance frequency of every 10th year. Emission for a service life of 100 years is presented in Table 3-15 and Figure 3-3.

3 Bituminous mixtures not containing coal tar – “asphalt”

Table 3-15 Re-use scenario – In plant, surface layer. Estimated release of PAH per kg RA. Based on Table 3-8 (particle release), Table 3-9 (fume emissions), Table 3-11 (leaching data) and Table 3-12 (surface run-off emission). PAH-16 content in recycled asphalt pavement is 100 mg/kg. Service life is 100 years.

Re-use scenario In plant		Emission through particles (mg PAH / kg RA)		Emission through fumes (mg PAH / kg RA)	Emission through leaching (mg PAH / kg RA)		
L/S ^a	Years	Milling	Wear		Storage	Use Percolation	Run-off
0.087	1	0.05	1.2	0.7	0.0027	0.0019	0.16
0.87	10	0.1	2.4	1.4	0.0054	0.019	1.6
1.7	20	0.15	3.6	2.1	0.0081	0.038	3.2
2.6	30	0.2	4.8	2.8	0.011	0.057	4.8
3.5	40	0.25	6	3.5	0.014	0.076	6.4
4.4	50	0.3	7.2	4.2	0.016	0.095	8
5.2	60	0.35	8.4	4.9	0.019	0.11	9.6
6.1	70	0.4	9.6	5.6	0.022	0.13	11.2
7.0	80	0.45	10.8	6.3	0.024	0.15	12.8
7.8	90	0.5	12	7	0.027	0.17	14.4
8.7	100	0.55	13	7.7	0.030	0.19	16

^a L/S is calculated for a surface and bound base pavement layer 0.24 m thick since the percolating precipitation is expected to be in equilibrium throughout the asphalt pavement.

Figure 3-3 Re-use scenario – In plant, surface layer. Cumulative releases as function of time and L/S. Based on Table 3-8 (particle release), Table 3-9 (fume emissions), Table 3-11 (leaching data) and Table 3-12 (surface run-off emission). The release from different release mechanisms is added. Service life is 100 years.

The most important emission processes are particles released due to studded tires surface wear, emissions in fumes during recycling and leaching from the surface by diffusion. If emissions by surface wear and diffusion were directed towards road run-off water, they would correspond to a concentration in road run-off water at about 200 µg/l and 30 µg/l PAH-16. Particle loss during milling, or leaching by percolation are processes of low importance. The emitted cumulative amount PAH for a life time of 100 years is a substantial fraction, about 40 %, of the amount in the original RA. This is not unrealistic considering that the emission model assumes a surface renewal every 10th year. The continuous renewal of an asphalt surface layer for 100 years may not be technically realistic due to degrading of the binders and the aggregates. The modelled traffic load is high and represents only a very small fraction (about 1.5 %) of Swedish roads (STA, 2011b). If the traffic load is reduced by a factor of 10, the emissions by surface wear will be equally reduced while the maintenance frequency and related emissions (milling, fumes and storage) will be reduced by a factor of only 1.5 (Table 3-1). In this case, surface wear will not be as significant and fume emissions and surface leaching will dominate, which is also the case for a Danish road where surface wear is assumed to be 100 times lower than in Sweden (Table 3-8). The calculated emitted amount PAH-16 in these cases will be about 25 % (25 mg/kg) of the total content in recycled RA for a life time of 100 years. The emission factors

calculated for fumes and leaching from the pavement surface by diffusion are highly uncertain and thus also the total emissions.

3.8.3 Hot in-place recycling of asphalt

Hot in-place recycling of RA is a well-established method where RA is used in high performance applications in surface layers of pavements. A high percentage of RA is re-used in the new pavement. Hot in-place recycling is expected to be performed repeatedly. Emissions are particle emissions related to surface wear, leaching during use and fume emission during recycling and paving. Emissions are calculated for surface layer re-use assuming a PAH content of 100 mg/kg. The “in use” scenario is a Swedish road with an average traffic load of 10 000 vehicles per day and traffic lane and a maintenance frequency of every 10th year. Emission for a service life of 100 years is presented in Table 3-16 and Figure 3-4.

Table 3-16 Re-use scenario – Hot in-place recycling, surface layer. Estimated release of PAH per kg RA. Based on Table 3-8 (particle release), Table 3-9 (fume emissions), Table 3-11 (leaching data) and Table 3-12 (surface run-off emission). PAH-16 content in recycled asphalt pavement is 100 mg/kg. Service life is 100 years.

Re-use scenario: Hot in-place		Emission through particles (mg PAH / kg RA)	Emission through fumes (mg PAH / kg RA)	Emission through leaching Use (mg PAH / kg RA)	
L/S ^a	Years	Wear		Percolation	Run-off
0.087	1	1.2	7	0.0019	0.16
0.87	10	2.4	14	0.019	1.6
1.7	20	3.6	21	0.038	3.2
2.6	30	4.8	28	0.057	4.8
3.5	40	6	35	0.076	6.4
4.4	50	7.2	42	0.095	8
5.2	60	8.4	49	0.11	9.6
6.1	70	9.6	56	0.13	11
7.0	80	10.8	63	0.15	13
7.8	90	12	70	0.17	14
8.7	100	13	77	0.19	16

^a L/S is calculated for a surface and bound base pavement layer 0.24 m thick since the percolating precipitation is expected to be in equilibrium throughout the asphalt pavement.

Figure 3-4 Re-use scenario – Hot in-place recycling, surface layer. Cumulative releases as function of time and L/S. Based on Table 3-8 (particle release), Table 3-9 (fume emissions), Table 3-11 (leaching data) and Table 3-12 (surface run-off emission). The release from different release mechanisms is added. Service life is 100 years.

The most important emission process is fumes during recycling. The second most important are particles released due to studded tires surface wear and leaching from the surface by diffusion. Leaching by percolation is a process of low importance. The emitted cumulative amount of PAH-16 for a life time of 100 years is larger than the total amount in the original RA. This is unrealistic, although a high emission can be expected considering that the emission model assumes that the surface is recycled every 10th year. The reason for this unrealistic value is that the emission factors calculated for fumes and leaching from the pavement surface by diffusion are highly uncertain and thus also the total emissions. The continuous renewal of an asphalt surface layer for 100 years by hot-in place recycling may not be technically realistic due to degrading of the binders and the aggregates. Decreasing the assumed traffic load to 1000 vehicles per day and traffic lane will give lower emissions for fumes and surface wear. However, the calculated emitted amount of PAH-16 from recycled RA in these cases will still be high, about 70 % of the total content (70 mg/kg) for a life time of 100 years.

3.8.4 Recycling of recovered asphalt in road base layers

Cold recycling of RA in road base layers is a well-established method. RA can be added as a fraction to new asphalt mixes for bound layers and at high percentage in unbound base layers. Recycling into unbound layers is a material recovery technique where the binder properties in the RA raw materials are not fully utilised. Quality requirements are less strict for base layers compared to surface layers and therefore low quality crushed RA from asphalt block excavation is regularly used in these applications. Base layers of a road are not regularly maintained and recycled unless in case of construction failure, demolition of the road or maintenance of subsurface installation. Asphalt block excavation and storage at permanent storage sites prior recycling is expected to be the most frequent recycling technique. Emissions are *a)* particle emissions during recycling and storage and *b)* leaching during storage and use. Emissions are calculated for recycling in unbound base layer assuming a PAH content of 100 mg/kg. A maintenance frequency of every 33rd year is assumed. Emissions for a service life of 100 years are presented in Table 3-17 and Figure 3-5.

Table 3-17 Recycling scenario – Road base layer. Estimated release of PAH per kg RA. Based on Table 3-8 (particle release) and Table 3-11 (leaching data). PAH-16 content in recycled asphalt pavement is 100 mg/kg. Service life is 100 years.

Recycling scenario: Road base layer		Emission through particles	Emission through leaching	
L/S	Years	(mg PAH / kg RA) Demolition Block excavation	(mg PAH / kg RA) Storage	(mg PAH / kg RA) Use Percolation
0.091	1	0.5	0.0027	0.002
3	33	1	0.0054	0.066
6	66	1.5	0.0081	0.13
9.1	100	2	0.011	0.2

Figure 3-5 Recycling scenario – Road base layer. Cumulative releases as function of time and L/S. Based on Table 3-8 (particle release) and Table 3-11 (leaching data). The release from different release mechanisms is added. Service life is 100 years.

The most important emission process is particles released during demolition. Leaching during storage and leaching by percolation is of low importance. The released amount of PAH-16 is only a small fraction of the amount in the original RA, about 2 %. Essentially all PAHs will remain in the construction after a service life of 100 years and any environmental or health risks will depend on the land use scenario for the construction in its future use.

3.8.5 Disposal/landfilling of RA

Emissions related to landfilling are particle emissions during recycling and leachate from the landfill. Emissions are calculated assuming a PAH content of 100 mg/kg. Emission for a service life of 100 years is presented in Table 3-18. The following parameters were chosen for the calculation of cumulative release: total infiltration through top cover system 5 m per year; landfill height 10 m and density 1800 kg/m³. The service life of 100 years corresponds to an L/S ratio of 0.28 l/kg. The leaching to the environment (cumulative release) has been calculated by linear interpolation of data in Table 3-11 to the L/S relevant for a landfill.

Table 3-18 Landfill scenario. Estimated release of PAH per kg RA.

Based on Table 3-8 (particle release) and Table 3-11 (leaching data). PAH-16 content in recycled asphalt pavement is 100 mg/kg. Service life is 100 years.

Landfill scenario		Emission through particles (mg PAH / kg RA) Demolition		Emission through leaching (mg PAH / kg RA) Use
L/S	Years	Block excavation	Milling	Percolation
0.0028	1	0.5	0.05	0.00006
0.28	100	0.5	0.05	0.006

The most important emission process is particles released during demolition. Leaching from the landfill is of low importance. The released amount PAH-16 is only a small fraction of the amount in the original RA, < 0.5 %. Essentially all PAHs will remain in the landfill and any environmental or health risks will depend on the landfill scenario after 100 years.

3.9 Comparison of scenarios

The diversity of recycling routes for RA that are available and used to day complicates the assessment of emissions and a simplified approach has been used by selecting a set of recycling scenarios

3 Bituminous mixtures not containing coal tar – “asphalt”

representing a range of different options. The assessments are performed for a service life of 100 years, while the large scale use of asphalt on roads is not older than about 60 years and RA recycling is not older than about 15 years. Development of new recycling techniques and recycling routes are likely and the estimated long term emissions highly uncertain. Emissions pathways for recycled tar contaminated RA are related to the

- Loss of particles during demolition and storage as well as through surface wear of the pavement by studded tires
- Emission of PAH in fumes during hot recycling
- Emission of PAH in surface run-off and in percolating water during storage or from the construction.

The emissions of PAH-16 in tar contaminated RA are expected to be basically proportional to the content of PAH in the asphalt. For leaching processes and emitted fumes, the PAH emitted has a distribution of PAH compounds that is different from the composition in the pavement. Heavy and carcinogenic PAH-compounds (PAH-7) are retained in the asphalt matrix. A high availability is expected for PAH-16 in asphalt.

Emissions estimated for a selected set of scenarios are presented in Table 3-19. Recycling scenarios where RA are recycled in the surface course of a road result in high emissions related to wear of the surface, leaching in surface run-off and emission in fumes (paved roads) or in percolating precipitation (gravel roads). A large fraction of the PAH-16 in recycled RA is emitted in the hot in plant recycling scenario (40 %), while essentially all PAHs are emitted in hot in-place recycling or gravel road surface recycling. Recycling scenarios where RA is recycled in subsurface constructions result in low emissions, about 1-2 % of the PAH-16 content. The most important emission pathway is particle loss during demolition (0.5-2 %), but also leaching in percolating precipitation (0.2-0.4 %). Essentially all PAHs will remain in the subsurface construction after a service life of 100 years and any environmental or health risks will depend on the land use scenario for the construction in its future use. Leaching during temporary storage prior recycling is the emission process with the lowest impact on the total emission.

Table 3-19 Compilation of estimated PAH-16 emission in various recycling scenarios. Emission for a service life of 100 years is given as a fraction of PAH content in recycled RA.

	Recycling scenario					
	Hot in plant surface course ^a	Hot in-place surface course ^a	Gravel road surface	Bound road base layer	Back- filling	Land- filling
L/S of the construction: (100 years)	8.7	8.7	620	9.1	17	0.28
Particles						
Demolition	0.6 %		0.5 %	2 %	0.5 %	0.5 %
Surface wear	13 %	13 %	~85 %			
Storage			0.05 %		0.05 %	
Leaching						
Storage	0.03 %		0.003 %	0.01 %	0.003 %	
Percolation	0.2 %	0.2 %	14 %	0.2 %	0.4 %	0.006 %
Surface run-off	16 %	16 %				
Fumes	8 %	77 %				
Total emission	40 %	~100 %	~100 %	2 %	1 %	0.5 %

^a Calculated for a traffic load of 10 000 vehicles per lane and year. Reducing the traffic load to 1000 vehicles per lane and year will reduce the total emission by about 30 % for hot in plant recycling and 10 % for hot in-place recycling.

The uncertainties associated with the estimated emissions are large. Data support for PAH emissions in fumes as a function of tar contamination is scarce and probably highly variable depending on the

equipment and technology used. Very few data is available for leaching in surface run-off and the quality is low due to uncertainties in the model description of the emission process. Data on the release of particles due to surface wear by studded tires is available, but highly variable depending on technical and meteorological conditions. Data on particle release related to surface wear of gravel roads, demolition or storage is very scarce or of very low quality. Leaching in percolating water is supported by a fair amount of laboratory data and a valid model description of the emission process, but emissions can be strongly influenced by environmental conditions in the field. **It is noteworthy that the most important emission processes (fumes, particle release and surface run-off) are the processes where the uncertainty is the largest.**

Estimation of yearly PAH-16 emission from Swedish tar contaminated asphalt pavements

There is no updated information on the actual frequency of tar contamination and the PAH-16 content in recycled RA for the Nordic countries. In general, RA recycling techniques result in mixing contaminated layers of bitumen with unpolluted and virgin asphalt or aggregates, resulting in a more diffuse, dispersed and diluted pollution. This makes the future detection of contamination in the field or by screening methods more difficult.

RA with PAH-16 content below 70-100 mg/kg is regarded as uncontaminated by guidelines in the Nordic countries. In Sweden, where tar contamination is most frequent, it has been estimated that there are 10-16 Million tonnes of tar contaminated asphalt (most likely in road base layers), only in the state road network (Lindgren & Friberg, 2009). The municipal paved road network amounts to about 40 % of the state road network and assuming that the frequency of contamination is the same in both networks the total amount of *tar contaminated pavements* would add up to 14-22 Million tonnes. This corresponds to a 5-7 % fraction of the total stock of asphalt pavements in Sweden that has been estimated to 300 Million tonnes (Miliutenko et al., 2013). The fraction of *tar contaminated RA* in Sweden has been estimated to 15 % in Table 3-7 and the total RA generation to about 1.1 Mtonnes in Table 3-2. The higher fraction of tar contamination in RA compared to the fraction in the total stock is in line with the mixing of contaminated and uncontaminated pavement layers in the recycling process and the resulting content of PAH-16 can be expected to be lowered in the contaminated RA.

Based on a limited set of data for contaminated pavement layers (STA, 2004) an average PAH-16 content of 500 mg/kg for tar contaminated pavements and 100 mg/kg for tar contaminated RA is assumed as a conservative estimate. These figures can be used to estimate the total PAH-16 emission from Swedish tar contaminated pavements. Tar contamination of the pavement stock occurs most likely in bound base layers, and based on the emission rate given for percolation in Table 3-20, the yearly average emission of PAH-16 from tar contaminated pavements by leaching from road base layers is estimated to 140-200 kg per year.

Table 3-20 Estimated PAH-16 emission from tar contaminated asphalt in Swedish pavements.

Emission process	Stock in base layers (Mtonnes)	PAH-16 content (mg/kg)	Emission factor ^a	Yearly emission (kg)
Percolation from bound base layers	14-22	500	0.002 %	140-200

^a Calculated based on data in Table 3-19.

If tar contaminated RA is recycled into road base layers, these emissions will increase by merely 3 kg/per year due to emissions in the demolition and recycling process (Table 3-21).

Table 3-21 Estimated PAH-16 emission from tar contaminated asphalt depending on recycling route. Assuming that 0.16 Mtonnes of RA with a PAH content of 100 mg/kg is recycled annually.

Recycling scenario	Dominating emission process	Emission factor ^a	Yearly emission (kg)
Bound base layer	Demolition	0.02 %	3
Hot in plant surface course	Fumes + surface wear + surface run-off	0.4 %	60 ^b
Gravel road surface	Surface wear + percolation	~1 %	160 ^b

^a Yearly emission factors are calculated based on data in Table 3-19.

^b The yearly emissions will accumulate due to the fact that the emissions from road surfaces are much higher than from bound base layers.

However, if contaminated RA from road base layers is recycled into surface layers, the emissions will increase every year by, on average, 60-160 kg. If the total stock of tar contaminated asphalt from road base layers, 7-11 Mkg, was recycled into road surfaces, the yearly emissions of PAH-16 would be in the order of 30 000 to 100 000 kg per year. Accordingly, the transfer of even a small percentage of the stock of contaminated unbound layers to road surface layers yields a significant increase in diffuse PAH emissions over time. The yearly recycled amount of asphalt is very small compared to the total stock and the share coming from contaminated pavements is even smaller. This means that it will take several hundred years before all contaminated asphalt contained in paved roads has been recovered.

Backfilling and landfilling scenarios

Backfilling will give emissions of the same magnitude as recycling into road base layers, more in percolation but less in future demolition and recycling (Table 3-19). However, while direct exposure or ecological risks due to elevated levels of PAH-16 are unlikely in road base layers they pose a potential risk in backfilling scenarios. A change of land use for a road will undoubtedly trigger demolition and recovery of the road pavement, whereas a backfilled area can change land use without any prior action.

Landfilling will give emissions that are significantly lower than in any other scenario, with the totally dominant emission being associated with particle release related to the demolition of the pavement prior landfilling and leaching being very low.

It should be noted that LCA based studies on RA re-used into new asphalt mixes and bound layers have shown that re-use is advantageous. This is due to the savings of bituminous binder and the emission associated with virgin binder but also with aggregate production and transport (Re-Road, 2012b; Miliutenko, 2012; Wahlström et al., 2013).

4 Concrete waste

4.1 Description and uses of concrete

Concrete is produced by mixing cement and mineral additives with fine aggregate (sand), coarse aggregate (gravel or crushed stone) and water. Small amounts of chemical admixtures are often added to improve workability in the fresh state. Freshly mixed concrete is a plastic material which makes it possible to produce structures of widely different shapes and forms. After a few hours it will stiffen, and upon time it will gradually gain high strength. The setting and hardening result from a chemical reaction between mainly the cement, mineral additives and water (hydration). To avoid any confusion the material components can be defined as shown here:

- **Clinker** = the material that is produced from the rotary kiln in a cement plant.
- **Cement** = clinker that is finely ground in the cement mill together with gypsum to control the setting, iron(II)sulphate to reduce the hexavalent chromium to the trivalent state, supplementary cementitious materials (SCM) (e.g. coal fly ash or ground granulated blast furnace slag), mineral additives/fillers (e.g. limestone).
- **Cement paste** = cement + water + mineral additives like silica fume (< 15 % by weight of cement) + chemical admixtures (< 1.00 % by weight of cement) that either influence the workability (e.g. plasticisers) or the setting and hardening process (accelerators and/or retarders). Cement paste is the "glue" in concrete.
- **Cement mortar** = a mixture of sand (< 8 mm), cement and water where the ratio of sand to cement is around 3 by weight.
- **Concrete** = a mixture of coarse and fine aggregates (approximately in the range of 0.12–20 mm), cement and water where the ratio of coarse and fine aggregates to cement is around 6 by weight.
- **Aggregates** = crushed stone, gravel and sand that is used in mortar and concrete.

The chemistry in concrete is dominated by the cement (with or without SCM) and the mineral additives compared to the relatively inert natural aggregate phase. The natural aggregates in concrete may partly be substituted with recycled aggregates like crushed concrete debris or coarser ash fractions. The former case is very limited today whereas different ash fractions are more frequently used. In concrete production today the addition of high quality coal fly ash is normal practice as it reduces the cement addition per cubic metre concrete produced. In cases where the reactivity of the ash is limited, it may partly replace the fine aggregate fraction in concrete and thus decrease the inertness of the aggregate part.

Concrete is the most frequently used building material globally and it may be delivered freshly prepared at the building site produced in ready-mixed concrete (RMC) plants or it may be delivered in the form of precast element concrete products. Concrete is widely used in the building and construction industry. Specification, performance, production and conformity of concrete are given in EN 206-1. The total production of ready-mix and precast concrete elements in the Nordic countries in 2003 and 2011 are given in Table 4-1.

Table 4-1 Concrete production in the Nordic countries in 2003 and 2011 (million m³).

Country	Ready-mixed concrete production		Precast element concrete production	
	2003 ^a	2011 ^b	2003 ^a	2011 ^c
Denmark	2.2	2.1	0.41	not estimated
Finland	2.3	3.0	not estimated	not estimated
Norway	2.4	3.5	0.32	0.63
Sweden	3.2	3.3	0.56	not estimated

^a Jónsson (2005)

^b Statistics from European Ready Mixed Concrete Organization

^c Data received from Norwegian Precast Element Association

4.2 Demolition of concrete

The life-time of concrete may vary from a few decades to thousands of years depending on the durability of concrete and the aggressiveness of the surrounding environment. Other important factors influencing the lifetime are the need for re-construction, the needs of the consumer (that are constantly changing) and the desire for more attractively designed buildings. In some cases, 10 years old concrete buildings are demolished due to demands and wishes to raise new and more modern designed buildings (and also with increased functionalities). The market economy (real estate costs, rental income, etc.) makes it cheaper to raise a new building rather than reconstruct or partly deconstruct the old building and this is often encountered in developing countries. In Norway the expected lifetime of concrete buildings is 50-100 years.

The figures in Table 4-1 only give an indication of the annual new concrete mass (ready mixed and precast element production). Since the actual lifetime of a concrete building will vary, the concrete production must be seen in relation to the actual demolished concrete mass over a time period to give reasonable estimates of the concrete rubble generation trends in the coming years.

However, exact data for concrete rubble generation is difficult to obtain. Official statistics in the Nordic countries differ in how the specific fractions in the heavy C&D waste are differentiated (Table 4-2). There is no official figure for solely concrete waste in Norway and the statistics include bricks and other heavy building materials in the same calculation (Statistics Norway, 2012). The statistics for Norway do not currently include the waste generated from sole construction works (*e.g.* road sector). Only wastes generated from removal of roads (construction of new roads), landscaping, etc. in relation to construction, renovation and demolition of buildings are included. In 2004, the material recovery (including backfilling and landfill cover) for C&D waste and concrete rubble in Norway was 15 % and 25 % respectively. In 2011, the recovery increased to 57 % and 88 % for the same waste categories.

The Swedish statistics use the term “mineral waste from the construction sector: concrete, stone, etc.” (SEPA, 2012b). According to the latest report, 150 000 tonnes are landfilled whereas 300 000 tonnes are classified as “unknown treatment” resulting in a recovery level of 53 % for the heavy C&D waste.

One part of the uncertainty in the official statistics is the quantity of waste that is not registered. For concrete rubble this includes unauthorised disposals (*e.g.* backfilling operations). Regarding ready-mix concrete, surplus concrete (fresh) in the mixer truck needs to be handled properly and there are examples of illegal dumping of fresh concrete. To what extent unauthorised disposal occurs is difficult to estimate. However, in the case of fresh concrete waste, the illegal dumping is considered to be low compared to the total produced concrete volume. In urban areas, large focus is put on recovery options. These options for surplus concrete may include; casting into usable concrete blocks; feedstock into RCA production after casting into elements at the aggregate plant; mix with the next delivery if the properties are matching.

Table 4-2 Generation and material recovery of concrete rubble in the Nordic countries (million tonnes).

Country	2004		2011	
	Generation	Recovery ^a	Generation	Recovery ^a
Denmark ^b	1.12	100 %	1.36 ^f	100 %
Finland ^c	No data	No data	1	> 50 %
Norway ^d	0.59	25 %	0.64	88 %
Sweden ^e	No data	No data	1.03	53 %

^a Includes the material recovery and other recovery operations, as defined in WFD.

^b Concrete rubble includes concrete

^c Concrete rubble includes mainly concrete and bricks

^d Concrete rubble includes mainly concrete and bricks. Data calculated from Statistics Norway.

^e Concrete rubble includes mineral waste from the construction sector (concrete, stones, etc.). Data from SEPA (2012).

^f Data from 2009 (www.mst.dk)

It is emphasised that demolished concrete may be further processed into recycled aggregates that fulfil the requirements for aggregates to be used in, for instance, road sub-base. The treatment technology for demolished concrete has advanced the past 10 years and includes different techniques for separation, crushing and treatment with water. In countries with scarce population (*e.g.* Norway), the

number of demolition projects is less predictable than in countries with large metropolitan cities (*e.g.* Tokyo). The demolition industry has therefore converted from stationary to mobile crushing plants in Norway. The recovery rates given in Table 4-2, include the operations relevant for concrete rubble and are material recycling, final cover at landfills and backfilling. From this we can deduce that the demolished concrete is in reality not necessarily processed further before being used in some of the recovery operations.

4.3 Legislation and guidance regarding recycling of concrete

In this section the Nordic legislation and guidance specific for recycling of concrete is described. If there is no specific legislation, the general waste and environmental legislation will apply. Refer to Chapter 2 for general information on the legislation on C&D waste recycling and to Annex C for European harmonized standards.

Nordic legislation contains regulation on both organic and inorganic substances. Concrete is highly inorganic in nature and the issue regarding possible release of organic substances is due to contamination from other sources (*e.g.* PAH and PCB). In addition, chemical admixtures are added during concrete production. However, the applied quantities are small and in general less than 1.00 % by cement weight (see chapter 4.1). In a recent Swedish study the ecological risk was evaluated regarding increased leaching of chemical admixtures from recycled concrete and concrete waste (Hansson & Norin, 2013). It was concluded that no precautionary measures are suggested to reduce emissions from recycled concrete or concrete landfills.

Denmark

In order to be recycled as aggregates C&D waste must fulfil both application-specific functional and environmental criteria. Some Danish quality criteria for recycling of crushed concrete aggregates as unbound road base materials are presented below. The Danish Road Institute has produced guidelines for the use of crushed concrete aggregates as unbound road base materials (DRD, 2004). This guideline sets quality criteria for the crushed concrete in terms of elasticity, particle size distribution, strength/resistance to abrasion and purity (*i.e.* content of other materials than concrete). Based on purity, the crushed concrete is split into three different quality groups (Table 4-3). It should be noted that these purity criteria are purely functional, and they are not approved by the Danish EPA in relation to environmental protection.

Table 4-3 Danish purity requirements for crushed concrete used as unbound road base (from DRD, 2004).

Fraction	Group		Content (% by weight)		
	Name	Constituents	Quality A	Quality B	Quality C
> 8 mm	"Concrete"	Concrete and natural materials	≥ 98	≥ 95	≥ 80
	"Tiles"	Tiles, lightweight concrete, expanded clay and mortar	≤ 2.0	≤ 5.0	≤ 20
	"Asphalt"	Crushed asphalt	≤ 2.0	≤ 2.0	≤ 2.0
	"Other, non-harmful"	Glass, porcelain, hard plastic, iron and similar hard materials	≤ 2.0	≤ 5.0	≤ 20
	"Other, harmful"	Wood, paper, light insulation materials* (styropor, polyurethane, etc.), soft plastic, slags, etc.	≤ 0.5	≤ 1.0	≤ 2.0
*Light insulation material alone			≤ 0.02	≤ 0.02	≤ 0.02

Finland

The Finnish Government Decree 591/2006 (rev. 403/2009) on the recovery of some waste materials in earth construction was approved in 2006 and revised in 2009. The purpose was to promote, through simplification of the environmental permit system, the use of selected wastes in earth construction

4 Concrete waste

fulfilling the requirements defined in the decree. The decree is material specific and covers besides concrete waste¹⁴ also fly ashes¹⁵ and bottom ashes¹⁶ from combustion of coal, peat and wood-based material. Concrete chippings refer to waste made of dismantled concrete structures or concrete waste from new buildings and the concrete industry by crushing the material into grains with a maximum diameter of 150 mm (content of bricks max. 30 %).

Under the scope of the decree are the following applications (professional use):

1. Public roads, streets, bicycle lanes, pavements and areas directly connected to these necessary for road maintenance or traffic, excluding noise barriers;
2. Parking areas;
3. Sports grounds as well as routes in recreational and sports areas;
4. Railway yards as well as storage fields and roads in industrial areas, waste processing areas and air traffic areas, roads, streets, pavements and areas directly related to those.

The decree shall only apply to earthworks in accordance with the approved street plan, action plan for a public area, permit or notification), excluding forest roads. Technical specifications and environmental requirements are listed in Table 4-4.

Table 4-4 Finnish technical specification and environmental requirements for earthworks (Finnish Government Decree 591/2006, rev. 403/2009).

Aspect	Requirement
Construction specification:	
max. height	150 cm; however, only a necessary quantity of waste shall be used
protection (pavement)	two categories: covered and paved
distance to groundwater	not specified: the structure containing waste shall not be in contact with groundwater
Distance to water well	The minimum distance of the structure containing waste from a well or spring intended for use as household water shall be 30 m
Use on groundwater area	No
Exposure route included in environmental impact assessment	Main focus on the release of harmful compounds to water
Key properties regulated	Limit values given to leaching (CEN/TS 14005, EN 12457-3) and content

The decree is based on the protection of soil and groundwater. Limit values are given both on total content and on the release of metals and salts both for basic characterization and compliance testing (focusing only on key elements). The limit values for leaching and total content in the concrete based C&D waste for earth construction purposes are shown in Table 4-5.

¹⁴ (EWC codes 101314, 170101 and 191212)

¹⁵ (EWC codes 100102, 100103, 100117 and 100124)

¹⁶ (EWC codes 100101 and 100115)

Table 4-5 Finnish limit values for content and release from C&D waste used in earth construction (Finnish Government Decree 591/2006, rev. 403/2009).

Harmful substance	Limit value, mg/kg of dry matter			Limit value, mg/kg of dry matter		
	Basic characterizations			Quality control investigations		
	Content ^{a1}	Leaching ^{a2} (L/S = 10 l/kg) CEN/TS 14405		Content ^{a1}	Leaching ^{a3} (L/S = 10 l/kg) Batch test EN 12457-3	
		Covered structure	Paved structure		Covered structure	Paved structure
PCB ^b	1.0			1.0		
PAH ^c	20					
Hydrocarbons ^d	500					
DOC		500	500			
Antimony (Sb)		0.06	0.06			
Arsenic (As)	50	0.5	0.5	50		
Barium (Ba)		20	20			
Cadmium (Cd)	10	0.02	0.02	10	0.02	0.02
Chrome (Cr)	400	0.5	0.5	400	0.5	0.5
Copper (Cu)	400	2.0	2.0	400	2.0	2.0
Mercury (Hg)		0.01	0.01			
Lead (Pb)	300	0.5	0.5	300	0.5	0.5
Molybdenum (Mo)		0.5	0.5			
Nickel (Ni)		0.4	0.4			
Vanadium (V)		2.0	2.0			
Zinc (Zn)	700	4.0	4.0	700		
Selenium (Se)		0.1	0.1			
Fluoride (F ⁻)		10	50			
Sulphate (SO ₄ ²⁻)		1000	6000		1000	6000
Chloride (Cl ⁻)		800	800			

^{a1} Determination of metal content: acid digestion and microwave assisted digestion in compliance with standard EN 13656 or digestion with aqua regia in compliance with standard EN 13657.

^{a2} For determining the leaching of dangerous substances, the up-flow percolation test shall be used in compliance with draft standard CEN/TS 14405.

^{a3} For quality control purposes, the two-stage batch test in compliance with standard SFS-EN 12457-3 can also be used.

^b Total quantity of congeners 28, 52, 101, 118, 138, 153 and 180. For the determination of PCB, the method to be used shall comply with draft standard EN 15308.

^c Total amount of compounds (anthracene, acenaphthene, acenaphthylene, benzo(a)anthracene, benzo(a)pyrene, benzo(b)fluoranthene, benzo(g,h,i)perylene, benzo(k)fluoranthene, dibenzo(a,h)anthracene, phenanthrene, fluoranthene, fluorene, indeno(1,2,3-c,d)pyrene, naphthalene, pyrene, chrysene).

^d Hydrocarbon fractions C10-C40. Test method: EN 14039

The determination of harmful substances included in the waste and leaching from it shall be conducted, in the first instance, by using standardised and, secondarily, other methods found adequate in terms of sensitivity of detection, accuracy and repeatability.

An important part of the implementation of the decree concerns quality control. For this purpose a Finnish standard (SFS 5884) was developed. The standard (published in 2001) defines the procedure and requirements for production control of secondary materials and it was used as the base for the decree. The standard specifies the basic requirements for a production control system, technical and environmental classification of crushed concrete products, technical and environmental properties to be monitored as well as sampling and monitoring methods. Requirements for the management of deviations and reporting are also presented. The standard is material specific and the waste is classified into three classes by raw material and technical properties.

The scope and the base of the standard 5884 are as follows:

- Waste from selective demolition (origin of material known)
- Professional operation
- Material classification according to technical and environmental properties

Examples of quality parameters in the standard are:

4 Concrete waste

- purity of received waste, which should be checked visually both when material is received and at different stages of manufacturing process
- grain-size distribution and frost susceptibility
- compressive strength
- purity of produced material (amount of organic impurities and the amount of brick)
- environmental acceptability based on leaching of the compounds that are most likely to exceed applicability criteria for the application in question.

Norway

According to the Norwegian Pollution Control Act (Section 32) concrete waste must be treated in an authorised waste plant. The exceptions are when recycling or beneficial use can be adopted. These operations are evaluated against the risk for pollution and are regulated as follows:

1. A permit is given under the provision of Section 11, Paragraph 1;
2. A permit is not needed as the operation does not cause any significant pollution under the provision of Section 8, Paragraph 3;
3. An authorisation is given through a specific regulation.

Note that if a permit is not necessary, the contractor in reality still needs to document that the operation does not cause any pollution in order to have the sound basis for the evaluation.

Thus, for recycled concrete aggregates (RCA) in unbound applications (*e.g.* back-filling, sub-base of road, etc.), these requirements need to be fulfilled because the total concentrations of some metals in RCA often exceed the soil criteria for sensitive land use, as shown in Table 4-6. In addition, the limits for the solid concentration of PCBs are 0.01 mg/kg and 50 mg/kg for sensitive land use and hazardous waste, respectively. Concentrations of PCBs are sometimes found orders of magnitude above 0.01 mg/kg in paints and mortar attached to the concrete and the PCB contaminated parts are therefore required to be removed prior to demolition.

Table 4-6 Total concentration limits issued by the Climate and Pollution Agency in Norway and concentrations obtained in processed concrete and masonry rubble (mg/kg).

Substance	Total concentrations limits		Concentration in RCA ^a
	Sensitive land use	Hazardous waste	
As	< 8	> 1000	1.0-6.4
Cd	< 1.5	> 1000	< 0.72
Cu	< 100	> 25000	9.3-150
Cr total	< 50	> 25000	49-147
Hg	< 1	> 1000	0.01-0.07
Ni	< 60	> 2500	14-107
Pb	< 60	> 2500	4.5-185
Zn	< 200	> 25000	34-553
SO ₄ ²⁻	Not specified	Not specified	2995-8923 ^b
PAH	< 2	> 2500	0.02-19
PCB-7	< 0.01	> 50	< 0.004-0.14

^a Metal concentration in 12-20 different RCA (recycled concrete aggregates) samples (compilation of data from Engelsen et al., 2007; and Bianchini et al., 2005).

^b Released content at pH < 2 from five sample batches (Engelsen et al., 2009).

Sweden

The Swedish guidance documents include both functional criteria specific for the application and environmental criteria. The technical requirements are specified in the Swedish Transport Administration's specification for crushed concrete used as road material, which was introduced in 2004 and revised in 2013 (STA, 2013a,b,c). To be relevant as road material the concrete waste must consist of at least 50 % by weight of concrete and may include a maximum of 10 % lightweight concrete and 1 % other contaminants. The specification classifies the concrete waste into four quality classes according to purity and resistance properties. Design parameters and recommended use are

attributed to the quality classes. The absence of dangerous substances shall be proved through a certificate prior to the demolition. Relevant substances are PAH, PCB, CFC, asbestos and mercury.

4.4 Recovery operations

The relevant recovery operations for concrete rubble are:

- Re-use
- Recycling
- Backfilling

Re-use of concrete is rare in the Nordic countries. There are only a few examples where concrete buildings are stripped and the structure is used for another purpose while maintaining the majority of the structure. Another option is re-use of concrete elements which is also rarely practised in the Nordic countries. This concept may be more relevant when buildings are planned and designed for dismantling after service life.

Recycling of concrete often consist of size reduction and sieving in order to obtain a certain particle size distribution. There are several scenarios for recycling which may be differentiated in unbound use and bound use. The most important user scenarios today are:

Recycling in unbound applications:

- Road construction
- Utility trenches
- Parking lots
- Backfilling at demolition sites

Recycling in bound applications:

- Concrete production

In the unbound user scenarios, the recycled concrete replaces aggregates in most cases. The level of processing of the concrete rubble is dependent on the technical requirements for the natural aggregates used in a specific scenario (*e.g.* bearing capacity, particle size and grading). The same technical requirements may also be applied on the recycled aggregates.

The most demanding application of concrete rubble is to use it in concrete production because the aggregates need to fulfil a range of material quality criteria (*e.g.* organic, chloride and alkali content). Moreover, the fine fraction of recycled concrete aggregate may cause high water absorption in the final concrete. Although the latter application is considered to be high-grade recycling, recycled aggregates from concrete rubble are insignificantly used in the production of ready-mixed and precast-element concrete in the Nordic countries. In Norway, only the concrete waste produced internally (left-over) at a concrete plant is recycled into new concrete. More than 95 % of the recovered concrete rubble is therefore applied in unbound user applications.

The most common backfilling scenarios are filling of empty lots after buildings are demolished (reclamation purpose) and landscaping (engineering purposes). Note that it is debatable whether the waste used for the final cover of landfill is recycling (material recovery) or backfilling by landscaping (other recovery). The quantity of concrete rubble that is backfilled without a permit or without sufficient documentation (see Section 4.3) is unknown.

Sweden does not separate between backfilling/terracing/final cover disposal and recovery activities such as road construction. To recover waste for construction purposes according to the guidelines the following must be fulfilled: a) The measure should have a specific purpose. b) The waste that is recovered must have adequate technical characteristics compared to the material it is intended to replace and c) It can be recycled without risk of harmful environmental and health effects.

4.4.1 Selected recovery operations for studies on emissions and impact on transports

Four recovery operations were selected for further studies: Recycling as aggregate in unbound road layers (sub-base); Recycling as aggregate in concrete production, Backfilling as final cover on landfills and finally Backfilling as filling material at the demolition site.

Recycling as aggregate in unbound road layers (sub-base)

In this application, the recovered concrete needs to fulfil the technical specifications given by the road authority (*e.g.* NPRA, 2011). Normally this can be achieved by processing the concrete rubble in the modern crushing facilities available today. Using processed concrete rubble in road construction is one of the most frequently used recovery scenario in many countries around the world.

Recycling as aggregate in concrete production

In order to use processed concrete rubble in the production of new concrete, the recycled concrete aggregates needs to fulfil the requirements for natural aggregates. Since the material source largely varies (*e.g.* variation of the concrete quality in service life and since concrete rubble is likely to be generated at several demolition sites), the challenge is to ensure a stable production of RCA with respect to technical properties. This recovery scenario is therefore considered to be challenging but regarded as a high value recycling application.

Backfilling (use as final cover material on landfills)

This is the application where most of the recovered concrete in Norway is used today. The typical use has been in the drainage layer. In addition, it is used in relation to road foundations and trenches at the landfill sites.

Backfilling (use as filling material at demolition sites)

In this application, the scenario conditions (*e.g.* thickness, distance to groundwater, etc.) vary to a large extent compared to the conditions in the other recovery scenarios.

4.5 Disposal/landfill of concrete waste

Landfilling is the least favourable disposal alternative for concrete rubble. Release data originating solely from concrete rubble at a landfill site is difficult to obtain due to contributions from other fractions of C&D waste that typically are landfilled together with concrete. However, according to the Landfill Directive, uncontaminated concrete rubble (EWC 1701 01) is acceptable at landfills for inert waste without testing. In case of contamination, testing is required.

In laboratory testing and geochemical speciation modelling studies, the release behaviour of concrete rubble has been investigated extensively. For most of the parameters, the leaching performance complies with the Norwegian criteria for inert waste, provided that the testing is conducted according to the given standards. Some examples are shown in Table 4-7. It should be noted that the leaching tests performed according to the prescribed standards (EN 12457 and prEN 14405) are in many cases conducted at high pH (> 11) due to the crushing of the samples. The release at these pH values differs from the release at mildly basic pH.

Table 4-7 Leaching (mg/kg) from mixed and uniform concrete waste at L/S of 10.

Substances	Mixed 1 ^a	Mixed 2 ^a	Mixed 3 ^a	Uniform ^b	Landfill Inert waste ^c
As	0.003	0.057	0.034	< 0.02	0.50
Cd	< 0.0002	< 0.0002	< 0.0002	< 0.001	0.04
Cr total	0.09	0.46	0.23	0.09	0.5
Cu	0.10	0.24	0.20	0.01	2
Ni	0.01	0.10	0.06	< 0.01	0.4
Pb	0.001	0.001	0.006	< 0.007	0.5
Zn	0.24	0.06	0.12	0.01	4
SO ₄ ²⁻	not measured	not measured	not measured	9.60	1000
Sum PCB-7 ^c	0.17	0.04	0.19	not measured	1

^a Data collected from Engelsen et al., 2012a.

^b Data collected from Engelsen et al., 2010.

^c Total concentration (mg/kg); in concrete waste samples and Norwegian landfill criteria for inert waste.

4.6 Resources saved and impact on transports

4.6.1 Sub-base application scenario

The depth of a sub-base layer may vary in the range of 40-100 cm. It can be noted that the sub-base may consist of layers with different materials (*e.g.* 50 cm of a 100 cm layer is replaced by recycled aggregates).

The quantity of primary aggregates (X) saved when producing a sub-base of 1 m³ with recycled aggregates processed from concrete rubble (RCA) may be estimated according to the following equations:

$$X = \frac{(\text{mass of primary aggregates used if no recycling}) - (\text{mass of primary aggregates used when recycling is applied})}{\text{mass of RCA used}} \quad (4a)$$

$$X = \frac{(\rho_{\text{bulk, primary aggregates}} \times V_{\text{primary aggregates without recycling}}) - (\rho_{\text{bulk, primary aggregates}} \times V_{\text{primary aggregates with recycling}})}{\rho_{\text{bulk, RCA}} \times V_{\text{RCA used}}} \quad (4b)$$

If 1 m³ of primary aggregates is replaced by 1 m³ of RCA, the Equation (4b) simplifies to:

$$X = \frac{\rho_{\text{bulk, primary aggregates}}}{\rho_{\text{bulk, RCA}}} \quad (4c)$$

Equation (4a) expresses the mass of primary aggregates saved per kg of RCA used. In a compacted sub-base, the bulk densities of the primary aggregates and the RCA are approximately 1800 kg/m³ and 1500 kg/m³, respectively. Consequently, as much as 1.2 kg of primary aggregates are saved when 1 kg of RCA are used in the sub-base according to (4c) or 1800 kg of primary aggregates are saved for each square meter of highway constructed with a sub-base depth of 1 meter.

Transport is an important parameter to evaluate when RCA is considered to be used in the sub-base. There are a number of local conditions that influence the impact on transportation. These may include whether a mobile or a stationary processing plant is available, the location of the virgin material source, the sub-structure material, etc. However, in several cases a more simplified and generalised evaluation is needed as a first assessment of the impact on transports.

A simplified assessment may be conducted for the Oslo urban area by treating the distance as the master variable and divide it into the following sub-distances:

- Distance from demolition site to processing plant (DS-PP): 0-20 km.
- Distance from processing plant to construction site (PP-CS): 0-20 km.
- Distance from the natural aggregate source to construction site (NA-CS): 0-20 km.
- Distance from demolition site to landfill (DS-LF): 0-50 km.

4 Concrete waste

To evaluate the impact on transports, the amount of total transport can be estimated by summing the sub-distances. An assessment in a qualitative way will be to decide whether the total transport will decrease, increase or stay at the same level when the material recovery increases. Maximum sub-distances from the demolition site to the processing plant and the landfill were chosen to be 20 km and 50 km, respectively. This is considered to cover the majority of the locations in urban areas. The distance from the stone quarry to the construction site was selected to be in the range of 0-20 km.

The boundary at a distance of 0 km represents the following cases:

- DS-PP = 0, processing plant is located at the demolition site (*e.g.* mobile plant).
- PP-CS = 0, demolition and construction site is at the same location (*e.g.* concrete pavement is recycled on-site) and the processing plant is located at the demolition/construction site (*e.g.* mobile plant).

A simplified assessment is shown in Table 4-8 and it can be seen that the total transport will decrease in most of the cases. Green cells and negative numbers indicate a decrease in transport, while red cells and positive numbers indicate an increase in total transport. In several cases mobile processing plants are located at the same area as the crushing plant for natural aggregates due to the fact that permits are already in place (noise, dust, etc.). In such cases NA-CS equals out PP-CS and the total transport will decrease when the location of the landfill is more than 20 km away from the demolition site.

Table 4-8 Sub-base application scenario – total transport (km) estimated for different locations of demolition, construction, natural aggregate and landfill sites. Positive numbers (red cells) indicate an increase in the total transport.

Distance from demolition site(DS) to:			Distance from processing plant to construction site (PP-CS)								
			0			10			20		
			NA-CS			NA-CS			NA-CS		
			5	10	20	5	10	20	5	10	20
Processing plant (PP)	Landfill (LF)										
	0	10	-15	-20	-30	-5	-10	-20	5	0	-10
	0	20	-25	-30	-40	-15	-20	-30	-5	-10	-20
	0	50	-55	-60	-70	-45	-50	-60	-35	-40	-50
	10	10	-5	-10	-20	5	0	-10	15	10	0
	10	20	-15	-20	-30	-5	-10	-20	5	0	-10
	10	50	-45	-50	-60	-35	-40	-50	-25	-30	-40
	20	10	5	0	-10	15	10	0	25	20	10
	20	20	-5	-10	-20	5	0	-10	15	10	0
	20	50	-35	-40	-50	-25	-30	-40	-15	-20	-30

NA = Natural aggregate source

4.6.2 Concrete aggregate application scenario

Certain chemical and physical properties need to be fulfilled for aggregates that are to be used in concrete production. Important parameters like for instance chloride and alkali content are described with upper limit values. Moreover, physical properties like water absorption, particle grading and size are important to be declared before use. These parameters will also apply for recycled aggregates. As the source material (*i.e.* heavy C&D waste) is heterogeneous, the material properties of RCA will vary and a more demanding declaration system is required, which will limit the use in concrete production. The heterogeneity of the source material will have proportionally lower influence on the required physical and mechanical properties for aggregates used in road construction. Thus, more challenges are faced when RCA is used in concrete production.

However, there is an increased focus globally on the high-value recycling for concrete and technical specifications on how to produce concrete with recycled concrete aggregates have been developed in many countries (*e.g.* Japan, New Zealand and Norway). In for instance Norway, the technical guideline allows a substitution level up to 20 % of the aggregates in the fraction of 4-32 mm by recycled aggregates for concrete grade C55 and up to 30 % for concrete grade C35 (Norwegian Concrete Association, 1999). Concrete mix designs (C35) that include RCA are shown in Table 4-9.

Table 4-9 Concrete mix design using recycled aggregates from C&D waste (RCA) for grade C35. (kg/m³)

Material	Mix		
	0 % RCA ^a	30 % RCA ^b	60 % RCA ^a
Sand 0-8 mm	939	939	939
Crushed stone 8-11 mm	347	347	174
Crushed stone 11-16 mm	347	87	0
Crushed stone 16-25 mm	173	173	173
Recycled aggregates 10-20 mm	0	260	520
Binder (slag cement)	319	319	319
Silica	20	20	20
Plasticiser (Scancem P)	2	2	2
Superplasticiser (Scanflyt-2)	2	2	2
Water	192	192	192

^a From Lahus et al. (2002)

^b Designed according to Lahus et al. (2002)

C35 is a commonly used concrete grade. If 30 % replacement is chosen, it can be seen from Table 4-9 that 30 % by mass of the natural coarse aggregates are replaced per cubic meter. According to this mix design, 260 kg of natural aggregate will be saved per cubic meter concrete produced. Around 2.4 million tonnes of ready-mixed concrete was produced in Norway in 2003 (Table 4-1) and around 60 % was in the strength class C25-35. This gives a theoretical saving potential of approximately 370 000 tonnes of natural aggregates for this application scenario.

The change in total transport in this scenario can be estimated using the same principles as in the previous case (sub-base). It is assumed that the processing plant cannot be placed at the demolition site(s) as the requirements for declaration is higher when RCA is used in new concrete. Therefore it is assumed that a stationary plant is located on the same area as the plants producing natural aggregate. This simplifies the estimation as the distance from the processing plant to the concrete plant will be the same for RCA and natural aggregates and the total transport depends only on the difference between the location of the landfill and the processing plant, as shown in Table 4-10.

Table 4-10 Concrete aggregate application scenario – total transport (km) estimated for different locations of demolition, concrete plant, natural aggregate and landfill sites. Positive numbers (red cells) indicate an increase in the total transport.

Distance from demolition site(DS) to:			Distance from processing plant to concrete plant (PP-CP)					
			10			20		
			NA-CP			NA-CP		
			5	10	20	5	10	20
Processing plant (PP)	Landfill (LF)	10	5	0	-10	15	10	0
		10	-5	-10	-20	5	0	-10
		10	-35	-40	-50	-25	-30	-40
		20	15	10	0	25	20	10
		20	5	0	-10	15	10	0
		20	-25	-30	-40	-15	-20	-30

4.6.3 Backfilling (final cover materials on landfills)

Since the construction site is at the landfill, the total transport will increase for this recovery scenario compared to using the concrete rubble in the sub-base or in new concrete production. This is most likely valid in urban areas (e.g. Oslo and Stockholm). Increased transport will be the result if the distance from demolition site to the landfill site is shorter than the two scenario sites mentioned above.

4.6.4 Backfilling (filling material at demolition site)

In this recovery scenario, the total transport will decrease because the material is used on site and will otherwise have to be transported to a landfill. A comparable scenario is when concrete rubble is processed on site (e.g. demolition of highway bridges) and the recycled concrete aggregates are used in new road construction at the same site.

4.7 Pollution content and release from recovered concrete

A compilation of total concentration data of organic and inorganic substances in RCA batches was given in Table 4-6. Compared to the soil protection criteria in Norway for sensitive land use, the values for Cr, Cu, Ni, Pb, Zn, PAH and PCB exceed these criteria. However, risk assessment based criteria have shown that the contents normally found in processed C&D waste are acceptable in for example road construction (Engelsen et al., 2012b). In addition, the compiled datasets do not exceed the Finnish criteria for earth work construction given in Table 4-5.

Release criteria are normally also given in addition to content criteria. The specific tolerable doses at the recipient (*e.g.* fresh water quality for Zn) dictate the maximum release at the source by reverse calculation (*i.e.* transport term). Thus, the source criteria can be calculated in several ways and the level of sophistication varies significantly. This includes a simplified calculation taking into account the mixing of pore water that causes dilutions. A far more sophisticated approach would be to consider the predominating bio chemical reactions (sorption, mineral precipitation, complex formation with organic substances and biologic degradation) in the transport term covering both the unsaturated and saturated zone by numerical models (*e.g.* Modflow). In the regulatory framework in Norway, the standard tool is based on simplified calculations.

The input data in the transport calculations mentioned above is usually release data which may originate from a number of different leaching tests. One of the most common methods for granular materials is the column test (CEN/TS 14405). A simplified substitute for the column test may be the batch test (EN 12457-3) conducted at different LS ratios (0.1-10). However, concrete is a high alkaline material which is subject to decrease in pH by carbonation and the carbonation level in concrete rubble varies to a great extent. This is due to a number of factors including the type of exposure of concrete in service life (indoor, outdoor, sheltered or unsheltered), the concrete quality and the metrological conditions. The pH dependent characteristic of the different substances will therefore be useful to consider additionally. The substances selected in Figure 4-1 illustrate that they behave completely different as function of pH. At field site, this means that for instance the initial sulphate leaching from recycled aggregates produced from fully carbonated material is significantly higher than from the carbonation-free material.

Figure 4-1 Leaching from concrete rubble as function of pH for partly carbonated (A-C) and carbonation-free (D and E2) samples. (Engelsen et al., 2009 and 2010).

A compilation of leaching data from RCA is given in Table 4-11 including different L/S ratios and also leaching data in the mildly basic region. The data shown in this pH region discloses the binding effects to the cement hydrate phases (*e.g.* Aft) at own pH and the effect of leaching when these phases are degraded at pH below 10 by carbonation. This cannot be seen in the leaching test at own pH even at a low L/S ratio. In Table 4-11 this is in particular the case for sulphate and chromium. Regarding the leaching of PCB-7, the results demonstrate the low solubility of these organic substances and are in compliance with earlier studies.

Table 4-11 Leaching data compiled from four separate sample batches.

Substance	Released concentration (mg/L)					
	L/S 0.6 pH 12.7-12.9	L/S 1.1 pH 12.7-12.9	L/S 5.3 pH 12.5-12.7	L/S 10.5 pH 12.6	L/S 10 ^a pH 12.6	L/S10 ^a (pH 8-10)
As	< 0.003-0.003	< 0.003	< 0.003	< 0.003	< 0.03	0.004
Cd	< 0.0002	< 0.0002	< 0.0002	< 0.0002	< 0.0002	< 0.0002
Cr	0.031-0.102	0.025-0.060	0.016-0.035	0.016-0.034	0.024-0.049	0.15-0.36
Cu	0.008-0.027	0.005-0.018	0.001-0.003	0.001-0.003	0.001-0.002	0.001-0.003
Ni	< 0.001-0.008	< 0.001-0.004	< 0.001	< 0.001	< 0.001	0.003-0.073
Pb	< 0.001-0.003	< 0.001-0.002	< 0.001	< 0.001	< 0.001-0.001	< 0.001-0.002
Zn	0.004-0.034	0.004-0.018	0.005-0.067	< 0.001-0.002	< 0.001	< 0.001
SO ₄	4.9-31	5.7-17	2.3-5.9	1.8-6.2	1.8-3.0	596-673
PCB	< 0.0012 ^b					

^a Compiled from two separate full pH dependent tests (Engelsen et al., 2010)

^b Data from three sample batches and the test is conducted at L/S = 1.5 (Engelsen et al., 2012a)

In a longer time perspective, quantification of the leaching studied in the laboratory and at field sites, identification of the processes governing leaching and the development of proper framework for assessing the environmental risk related to leaching will become increasingly important. This is due to the fact that the cement and concrete produced today and demolished in the future are increasingly produced with alternative raw materials and waste derived fuels (*e.g.* concrete with partly substitution of both binder and natural aggregates). The majority of the cement consumption today are blended cements containing coal based ash, slag from pig iron production and limestone which have different cement chemistry than what is the case with normal Portland cement. Therefore, the understanding of the leaching processes for Portland cement concrete may form the basis for the leaching process identification in concrete with different binders and aggregates.

Based on the results given in Tables 4-6 and 4-11, the cumulative release is calculated for Cr, SO₄²⁻ and Zn. Cr and Zn are readily found in concentrations above the soil quality criteria for sensitive land use in Norway and the leaching of sulphate is increasing significantly when pH is decreased by carbonation.

4.8 Emission scenarios and calculated release/spread of pollutants

This section describes emission scenarios for each of the four selected recovery operations for concrete waste, showing the potential for spreading of pollutants to air, land and water. A scenario without waste recycling (landfilling) is also described.

4.8.1 Recycling of recovered concrete in unbound road layers

The following three emission scenarios have been studied (Figure 4-2):

- during storage,
- during construction
- from the final pavement.

Figure 4-2 Emission pathways from recycling of recovered concrete in road sub-base: a) storage; b-c) construction; d) final site.

Emission pathways during storage

The emission pathways during storage are dependent on a number of conditions including storage time, time of the year (temperature and humidity), the amount of precipitation, particle size and the quality of the recycled aggregates. During storage, leaching to soil and ground water as well as particle emission to air are relevant to evaluate.

In the road sub-base, coarse aggregates are normally used (20/100 mm) which limits the release to soil and groundwater and also the emission of dust to air. This is due to the small specific surface area, compared to coarse aggregates that contain also smaller grain size fractions (*e.g.* 8/16 mm), and the low content of fine particles. It can also be assumed that good quality concrete are crushed and used for the purpose (maximum 5 % bricks). The storage time is dependent on the location of the processing plant.

In a road scenario, it can be assumed that mobile plants are producing RCA on-site which minimizes the storage time. It may therefore be concluded that the impact through release to soil and groundwater and emission of dust is negligible.

Emission pathways during construction

During construction, the relevant pathway is dust emission to workers and the environment. There is limited number of relevant studies on this issue. One study was conducted at a stationary recycling plant in Oslo where the dust emission was studied during the discharge of 0/10 mm of C&D waste from the truck (Uggerud, 2002). An increased concentration of dust particles ($< 10 \mu\text{m}$) was determined 3 meter away from the discharge point. The measured dust concentrations exceeded the criteria for the working environment. However, it should be noted that conservative conditions were selected (near discharge point and fine particles). Furthermore, toxic metals, PCB and PAH were determined in the collected dust particles and the normalised concentrations did not exceed the criteria for the working environment.

At the sub-base construction site, the particle size is coarser leading to less dust emission. In addition, moistening the RCA is also normal during dry periods. Under such conditions the risk for inhaling air with increased dust particles is assumed to be low.

Emission pathways from the final sub-base

The relevant exposure pathway for the final sub-base is release to soil and groundwater. The total emission load to the environment (cumulative release) can be calculated using the leaching data in Table 4-11. It was chosen to show the release for different L/S ratios and at different pH domains. The following parameters were chosen: infiltration = 30 cm/y; depth of sub-base = 50 cm; compacted density = 1850 kg/m³. The results are illustrated in Figure 4-3. The service life of 100 years corresponded to an L/S ratio of 10.8 l/kg.

Figure 4-3 Cumulative release as function of time and L/S in a road scenario (sub-base material). A constant source term is used as input to the model at mildly basic and high pH. The curves with and without markers represent the cumulative leaching where the input data used is derived from a batch leaching test at L/S ratio of 0.6 and 10, respectively..

4.8.2 Recycling of recovered concrete in new concrete

The following three emission pathways have been evaluated:

4 Concrete waste

- during storage of RCA
- during production of concrete with RCA
- during the use of concrete containing RCA in service life.

Emission pathways during storage

The emission pathways during storage are dependent on a number of conditions including storage time, time of the year (temperature and humidity), the amount of precipitation, particle size and the quality of the recycled aggregates. During storage, leaching to soil and ground water and particle emission to air are relevant to evaluate.

In ready-mix concrete production both coarse aggregates (8/22 mm) and finer fraction (0/8 mm) are used. Unlike the unbound user scenarios, recycled aggregates are stored under more controlled conditions at a RMC plant. Normally the aggregates are stored in open bunkers with roof and in closed silos before mixing. The risk for release to soil and groundwater is therefore low. The emission of dust to air may increase during handling of fine aggregate fraction in dry periods. However, an increased use of RCA in ready mixed concrete is easiest to achieve by using recycled coarse aggregate fraction (less demanding). The emission of dust to air is therefore evaluated to be limited.

Emission pathways during production of concrete with RCA

The emissions during RMC production are evaluated to be insignificant as the mixing of aggregates, water and cement is conducted under closed conditions.

Emission pathways from concrete containing RCA in service life

The relevant exposure pathway for concrete buildings and structures is release to soil and groundwater. The release may be controlled by surface run-off, diffusion or solubility control depending on the exposure conditions (heavy rainfall, submerged in water, etc.). However, the release potential from concrete with RCA is not believed to increase during service life compared to normal concrete. This is based on the fact that the natural aggregate fraction is only partly replaced by RCA. Furthermore, high quality concrete rubble and consequently low release potential needs to be used in RMC production.

4.8.3 Backfilling (use of recovered concrete as final cover material at landfill)

The specific scenario is to use the concrete rubble as a drainage material in the top cover system of a landfill (Figure 4-4). In a non-hazardous landfill, the hydraulic conductivity of drainage material should be beyond 10^{-4} m/s (Wahlström et al., 2008). This means that the top soil (> 1 m) will determine the extent of infiltration in the beneath drainage layer.

Figure 4-4 Construction layers in surface cover of landfills (Norwegian Environment Agency, 2003)

Emission pathways during the use as drainage layer

The relevant exposure pathway for this recovery scenario is release to soil and groundwater. Emission to air is not relevant to consider (Figure 4-4). The total emission load to the environment (cumulative release) can be calculated using the leaching data in Table 4-11. It was chosen to show the release for different L/S ratios and at different pH domains.

The following parameters were chosen for the calculation of cumulative release: infiltration through top soil cover = 40 cm/y; depth of drainage layer = 100 cm; compacted density = 1850 kg/m³. The results are shown in Figure 4-5. The service life of 100 years corresponds to an L/S ratio of 22 l/kg.

It is emphasised that calculations are simplified. It assumes near equilibrium conditions (at different pH and L/S ratios) which is a conservative assumption. However, under normal landfill conditions with coarse aggregates in the drainage layer, the release mechanisms is most likely diffusion controlled with intermittent wetting. The release will to some extent also be influenced by the chemical composition of the infiltration water which in turn is influenced by the chemical and physical composition of the top soil cover.

4 Concrete waste

Figure 4-5 Cumulative release as function of time and L/S in the landfill cover material scenario, applying a constant source term as input to the model at mildly basic and high pH. The curves with and without markers represent the cumulative leaching where the input data used is derived from a batch leaching test at L/S ratio of 0.6 and 10, respectively.

4.8.4 Backfilling (filling material at demolition sites)

In this recovery scenario, the field site conditions will vary significantly including the thickness and permeability of the top cover material, the thickness of the backfilling layer, the location of the backfilling (e.g. next to cellar wall or an isolated pit or a low lying area at the demolition site).

Emission pathways during the use as filling material at demolition sites

The relevant exposure pathway for this recovery scenario is release to soil and groundwater. Emission to air is not relevant to consider as it is assumed to be covered.

The total emission load to the environment (cumulative release) is not calculated due variations in the scenario conditions.

However, if it is assumed that backfilling with concrete rubble is placed above groundwater level, the net infiltration through the backfilling layer will be determined by the permeability of the top cover material. Hence, the cumulative release will most likely be higher than in the road sub-base scenario and also compared to the scenario where recycled concrete aggregates are used in new concrete production.

Compared to the backfilling scenario at landfill, the backfilling scenario at demolition site needs to be more defined in order to make an evaluation.

4.8.5 Disposal/landfilling of concrete waste

The specific scenario is to deposit the concrete rubble as a waste material on a landfill. The concrete rubble will therefore be placed below the final surface cover system of a landfill (Figure 4-4). Thus, the concrete rubble is placed beneath the impermeable mineral layer (or an artificial sealing).

Emission pathways during the landfilling of concrete rubble

The relevant exposure pathway for this non-recovery scenario is the release to soil and groundwater. Emission to air may occur while the waste is placed at the landfill but will be limited by the daily cover of waste. (Figure 4-4). The total emission load to the environment (cumulative release) is difficult to assess as it depends on the total infiltration through the top cover system and the type and life time of barrier used at the bottom of the landfill. However, the maximum infiltration through the top cover system in Sweden is 5 cm/year (i.e. annually 50 L/m²). This may be used as the maximum annual infiltration through the concrete rubble.

Furthermore, two principal cases may be discussed. The first case includes an infinite timeframe for the landfilling anticipating that the waste will not be removed. In this case it can be assumed that the source is exhausted and the total chemical concentrations may be used in the evaluation rather than the release data. The other case will be to consider the same timeframe as for the sub-base and the drainage layer scenario.

In order to compare the release with the other recovery scenarios, the calculations are conducted for a timeframe of 100 years. The total emission load to the environment (cumulative release) can be calculated using the leaching data in Table 4-11. It was chosen to show the release for different L/S ratios and at different pH domains. The following parameters were chosen for the calculation of cumulative release: total infiltration through top cover system = 5 cm/y; depth of concrete waste layer = 1000 cm; compacted density = 1850 kg/m³. The results are shown in Figure 4-6. The service life of 100 years corresponded to an L/S ratio of 0.27 l/kg.

4 Concrete waste

Figure 4-6 Cumulative release as function of time and L/S in a landfill scenario (no recovery). A constant source term is used as input to the model at mildly basic and high pH. The curves with and without markers represent the cumulative leaching where the input data used is derived from a batch leaching test at L/S ratio of 0.6 and 10, respectively.

4.9 Comparison of scenarios

The total emissions have been evaluated for the use of concrete waste in the following three recovery scenarios; as road sub-base material, as coarse aggregate in ready mix concrete production and as drainage layer in the top cover system of a landfill. In addition, landfilling of concrete rubble (non-recovery scenario) was also evaluated.

In Figure 4-7, the calculated cumulative release is compared for the recovery scenarios with the **unbound** use of concrete rubble. The landfilling option is also included. It can be seen in the figure that a lower release of Cr, Zn and SO_4^{2-} was calculated for the road sub-base scenario than for the drainage layer scenario (in top cover at landfill). It was not possible to calculate the leaching from the backfilling at demolition site due to the variation in construction practice (soil cover depth, distance to ground water, subgrade, etc.).

Figure 4-7 Release of Cr, SO₄²⁻ and Zn in two scenarios: sub-base (water infiltration is low) and landfill drainage layer (water infiltration 400 mm/year) compared to landfilling. Each scenario represents upper and lower cumulative release value regarding pH and L/S used in laboratory tests.

5 Bricks, tiles & ceramics – and mixtures of these and concrete

5.1 Description and uses of bricks, tiles & ceramics

Bricks are normally produced from sand, clay and lime (in some products).

Floor tiles are typically made of ceramic or stone. They can be clay tiles or slate tiles, when used as roofing tiles.

Generally the term ‘ceramics’ (ceramic products) is used for inorganic materials (with possibly some organic content), made up of non-metallic compounds and made permanent by a firing process. In addition to clay based materials, today ceramics include a multitude of products with a small fraction of clay or none at all. Ceramics can be glazed or unglazed, porous or vitrified (JRC, 2007). The main steps in the manufacture of ceramic products are the same for all types of ceramic products: mining/quarrying of raw materials and transport to the ceramic plant, storage of raw materials, preparation of raw materials, shaping, drying, surface treatment, firing and subsequent treatment (JRC, 2007).

The ceramics industry produces a large variety of different products for different applications with different properties (*e.g.* frost resistance). The major sectors which are based on the ceramics manufactured are as follows (JRC, 2007):

- wall and floor tiles
- bricks and roof tiles
- table- and ornamental ware (household ceramics)
- refractory products
- sanitary ware
- technical ceramics
- vitrified clay pipes
- expanded clay aggregates
- inorganic bonded abrasives

Focus in this report is on bricks & tiles from the following applications:

- building bricks and blocks,
- roof tiles such as extruded tiles and pressed tiles
- floor tiles and walls for covering
- paving bricks
- bricks from chimney pipes.

5.2 Demolition of bricks, tiles & ceramics

An essential prerequisite for utilization is selective demolition, which ensures that the material does not contain excessive amounts of dangerous substances or other impurities that may reduce its environmental and technical suitability or totally hinder its use. Bricks, tiles and ceramics should not be recovered from constructions contaminated by industrial activity, spills or combustion processes (*e.g.* chimneys). Typical impurities or contaminants harming utilisation are asbestos, mineral wool, concrete (critical for some applications), mortar (critical for some applications), heavy metals and polycyclic aromatic hydrocarbons (PAH). Furthermore, the quality of the bricks and tiles (*e.g.* properties of building bricks are different from lightweight bricks) may influence the technical properties of the material (*e.g.* frost resistance), thus limiting later possibilities for utilisation if not carefully sorted on-site (Huuhka, 2010; Monier, 2011)

Bricks and tiles can be directly re-used when free of dangerous materials and damage. For this purpose the bricks and tiles need to be manually dismantled, sorted and cleaned (*i.e.* bricks separated from

mortar and tiles from grout and adhesives). The dismantling and cleaning processes are labour intensive. Attention needs also be paid to the dust formation during the cleaning process. Especially cement rich mortars are difficult to remove (Huuhka, 2010; Monier, 2011). Generally, new bricks have a higher durability than old bricks and also other properties (*e.g.* dimensions, density, frost resistance, water uptake, resistance for heat loss in buildings) are better controlled, because the manufacture process can be more efficiently controlled today (Bisgaard, 2013).

For recycling of bricks and tiles the rubbles are crushed, sieved and often handled together with concrete rubble. Typically the same kind of machineries and handling steps are used for bricks as for concrete waste: sieving and magnetic separation (COVAM, 2006)

5.3 Legislation and guidance regarding recycling of bricks, tiles & ceramics

Here is described the Nordic legislation and guidance specific for recycling of bricks, tiles & ceramics. If there is no specific legislation, the general waste and environmental legislation will apply. Refer to Chapter 2 for general information on the legislation on C&D waste recycling and to Annex C for European harmonized standards.

Finland

The Finnish Government Decree 591/2006 (rev. 403/2009) on recovery of some waste materials in earth construction covers the use of concrete chipping (EWC codes 10 13 14, 17 01 01 and 19 12 12), also including bricks in the concrete waste to a maximum content of 30 %.

Denmark

No specific guidelines for bricks, tiles and ceramics.

Norway

No specific guidelines for bricks, tiles and ceramics.

Sweden

No specific guidelines for bricks, tiles and ceramics.

5.4 Recovery operations

Undamaged bricks and tiles can be used as such in constructions, typically in renovation of old constructions. However, the bricks and tiles need to be properly cleaned, because the excess mortar dust can inhibit the adhesion between mortar and bricks and lead to weaker masonry, depending on the mortar composition.

Bricks, tiles and ceramics are usually recycled as crushed material and mixed with concrete. The largest markets are to replace natural aggregates, especially in road works. Furthermore, crushed ceramic, mixtures of tiles, bricks and concrete rubbles can be used as an underlay and drainage material. However, typically the materials are used in low-grade applications or as a filling material, *e.g.* in noise barriers (Covam study, 2006; Mäkelä & Höynälä, 2000). Crushed glazed and unglazed tiles are currently also used as a raw material in rock wool in Denmark (Tommy Bisgaard, 2013).

A research project carried out by TBE (European Tiles & Bricks Association) in the late 1990s concluded that several applications are possible for recycled clay building materials (Table 5-1).

Table 5-1 Possible applications for recycled clay building materials (TBE, 2012).

Application	Specification
Fill material in pipe trenches	Crushed clay bricks and other masonry can be used to level and fill pipe trenches. A fine grain size of around 0-4 mm is normally used for pipe trench material and this can mostly comprise crushed masonry material.
Aggregates for in-situ and precast concrete and mortars	Coarser particles can be used for other applications (e.g. aggregates in concrete and mortars).
Road constructions	As crushed material in light-traffic roads.
Tennis court sand	<p>Sand for surfacing tennis courts is produced by crushing red bricks and roof tiles. The requirements for sand and other materials used in tennis courts may be laid down in standards and specifications stipulated by tennis governing bodies. The main requirements concern water permeability, grain size distribution, surface shear stability and a satisfactory proctor test. The fine surface layer is laid over courser-grained layers that can comprise crushed clay brick matter.</p> <p>The material is produced by crushing in hammer mills to a grain size of 0-2 or 0-4 minimum. The process is most efficient, when it occurs at brick or tile factories where there is an abundance of scrap material. Different bricks will give different qualities and colours of tennis sand. Clinker quality will have numerous benefits: Better water drainage, Unique colour, Greater density (less wind scatter), No moss problems.</p>
Plant substrates	Crushed bricks and tiles can also be used to form substrates for growing plants. The material may be mixed with other substances used in plant production, e.g. composted organic materials. This material is especially suited for green roofs. Flat roofs are covered with a dense polymer membrane and overlaid with 10-30 cm of the crushed brick and tile material.

The brick waste amount produced yearly in Finland is around 70 000 tonnes (Mäkelä & Höynälä, 2000). Undamaged bricks can be brought to regional recycling centres for later re-use, but currently the volumes are rather small (Kojo & Lilja, 2011). Bricks and tiles are mainly recycled as crushed and mixed with concrete and used in low grade road applications. The amount of crushed bricks must, however, due to technical reason be limited to maximum 30 %. In Denmark, the amount of re-used or recycled brick waste in 2007/2008 statistics was 240 000 tonnes (Danish EPA, 2010). No information was found for the other Nordic countries.

5.4.1 Selected recovery operations for studies on emissions and impact on transports

Backfilling (use as filling material in noise barriers etc.)

Waste material not suitable for use in road constructions can be used as filling materials, e.g. in noise barriers. Typically technical quality requirements for the material are not very high. Therefore the waste composition may vary greatly and the filling material is often a mixture of several waste fractions from a building. Noise barriers are either covered or isolated depending on the local situation and the environmental properties of the waste material used.

Recycling as aggregate in unbound road layers (sub-base)

Crushed bricks can be used in unbound road layers, e.g. sub-bases, but require mixing with concrete waste to achieve necessary geotechnical properties.

Recycling as aggregate in concrete and mortar production

Crushed bricks can be used as aggregate in concrete. However, according to the European standard for concrete (EN 206) the amount of coarse recycled aggregated (e.g. bricks) can generally replace at

maximum 30 % of the natural aggregate in concrete. Generally the produced concrete is used in non-supporting structures, e.g. indoor constructions and pavement blocks.

5.5 Disposal/landfilling of bricks, tiles & ceramics waste

According to the Council Decision 2003/33/EC on waste acceptance criteria for landfilling, a single waste stream consisting of non-contaminated bricks and tiles (not painted or covered with materials containing dangerous substances in significant amount) can be disposed at landfills for inert wastes without any testing. However, the waste producer needs to verify the purity of the waste. In practice, this means that the generated waste needs to be almost free from e.g. mortar and concrete. Consequently, the bricks and tiles are probably not disposed of at landfills for inert waste.

Unsorted construction and demolition waste containing bricks, tiles, ceramics and typical renovation waste, is generally landfilled on landfills for non-hazardous waste. The renovation waste is made up of a mixture of concrete, insulation materials, bricks, tiles and small amounts of wood. Less wood is expected in the future due to disposal ban of organic materials at landfills. Due to increasing landfill costs, more and more efforts are put on avoiding landfilling. In this report it is assumed that the release of sulphate and a few specific metals from bricks contributes to the major release, even if the waste contains other materials. Not many data were found for calculation. Calculations presented in Section 5.8.4 are partly from new bricks that have been crushed prior to testing.

5.6 Resources saved and impact on transports

The production of bricks and tiles is a highly energy intensive process. The biggest environmental benefits are gained by substituting new bricks and tiles with re-used undamaged and clean bricks and tiles. This would also save extraction of natural materials, landfill areas and to some extent transportation of raw materials (especially if bricks and tiles can be re-used on-site or near the construction to be demolished).

Currently crushed bricks and tiles seem to be primarily used in low-grade earth applications (an exception is recycling of unglazed tiles in rock wool production in Denmark). The benefits are saves of materials in the applications and avoidance of landfilling. In general for low grade use, transport distance by lorry is limited to 10-20 km, due to economic reasons.

When brick waste is recycled as road sub-base, it is used together with the concrete waste and can theoretically replace 30 % of primary aggregates used in the sub-base. When brick waste is used as backfilling at demolition sites, it also replaces virgin materials in the proportion 1:1 by volume. When recycled concrete aggregate is used to produce new concrete, maximum 20-30% by mass of the primary coarse aggregate is saved per cubic meter concrete produced.

The impact on transports is similar as presented for concrete waste.

5.7 Pollution content and release from recovered bricks, tiles & ceramics

Available data on content and leaching is shown in Tables 5-2 and 5-3. Table 5-3 also contains leaching results from old demolition waste which can be assumed to represent demolition waste containing bricks due to less sorting of bricks and concrete in the past.

The total content of harmful metals and organic compounds reported in Table 5-2 are low, typically at the same range as threshold values used for the assessment of clean soil. Small amounts of PCB might be found in the mixed concrete brick waste, if the sealing compounds have not been removed carefully before demolition.

In Table 5-3 it is notable that crushed bricks leach higher amounts of sulphate and vanadium and lower amounts of chromium than concrete waste. No data was available for crushed tiles, only for non-crushed tiles data from leaching tests (tank test) are available. The leaching of metals and sulphate from non-crushed tiles was very low and the leaching from tiles is therefore neglected in the calculations.

In the literature, no leaching data has been reported for blocks (monolithic materials) containing crushed bricks. Leaching data for monolithic bricks is published, but these data is not seen as relevant for cement blocks containing crushed bricks.

Table 5-2 Content data for crushed bricks and demolition waste.

Substance	Total content				
	Crushed bricks ^a	Mixture concrete/bricks ^a	Mixture concrete/bricks ^b	Mixture concrete/bricks ^b	Ceramic products ^c
	N=1 (mg/kg)	N=3 (mg/kg)	N=41, Median (mg/kg)	N=41, Range (mg/kg)	N=1 (mg/kg)
Arsenic	4	4	3.2	1-21,5	26
Barium					
Cadmium			0.20	0.01-0.7	6
Chromium			27.2	5.4-82	
Copper	36	23	15.7	5.1-114	
Mercury	0.23	0.25	0.01	0.005-0.35	
Molybdenum					5
Nickel	15	13	19	4.2-60.3	
Lead	38	67	16	4,9-255	
Vanadium					122
Zinc	140	68	93.8	23.8-635	
Sulphate					
PAH-EPA 16			0.7 (N= 33)	0.01-340 (N= 33)	
PCB-sum of 7 congeners			0.0085	<0.001-1.2	

^a Baun & Hjelmar, 2003.

^b SFT, 2003.

^c Aalbers et al., 1996.

Table 5-3 Leaching data (released amounts and concentration, respectively) for crushed bricks and demolition waste.

Substance	Release					
	Crushed bricks ^a N=1 at L/S 2 (mg/kg)	Mixture concrete/ bricks ^a N=2 at L/S 2 (mg/kg)	Demolition waste ^b N=1 at L/S 10) (mg/kg)	Crushed bricks ^a N=1 at L/S 10 (mg/kg)	C ₀ at L/S 0.1 ^c (mg/l)	□ (l/s) ^c (l/kg)
Arsenic	<0.05	<0.05		1.3	0.54	0.20
Barium	<0.05	<0.05		0.09		
Cadmium	<0.001	<0.001	<0.004	<0.005		
Chromium	0.2	0.1	0.5	0.09	0.052	0.38
Copper	0.05	0.1	0.24	0.01		
Mercury	<0.0002	<0.0005				
Molybdenum				0.31		
Nickel	0.02	0.04		0.02		
Lead	<0.002	<0.002		0.005		
Vanadium				2.3	5.26	0.45
Zinc	<0.02	<0.02	<0.1	0.02		
Sulphate	100	340	1300	3200	1550	0.53
PAH-EPA 16						
PCB-sum of 7 congeners						

^a Baun & Hjelmar, 2003.^b Wahlström et al., 2000.^c See explanations in Section 5.8.

According to VTT's experience, the origin of clay highly influences the total content and also leachability of some metals (*e.g.* arsenic and sulphate). The reported values for especially arsenic are quite high according to VTT's experience. It should also be noted that leaching results for crushed bricks in Table 5-3 are only for one test data.

5.8 Emission scenarios and calculated release/spread of pollutants

Emissions to soil and groundwater for each of the selected recovery operations are calculated based on data in Table 5-3. Calculations are based on the use of the following equation:

$$E = (C_0/\kappa)(1 - e^{-(L/S)\kappa}) \quad (5a)$$

where

E is the amount of a substance (in mg/kg), released over the period of time it takes for L/S to increase from 0 L/kg to the selected L/S value.

C_0 is the initial peak concentration of the substance in the leachate at L/S 0.1 (mg/l).

κ is a substance specific value and may be estimated from column test (see calculated values in Table 5-3).

Note that the use of the κ model above may lead to slightly different values than measured values because of curve fitting.

5.8.1 Backfilling of recovered bricks, tiles & ceramics

Brick waste arising from demolition typically contains mortar, plaster and concrete. It can be anticipated that this mixture leaches lower amounts of metal than presented in Table 5-3 due to the concrete content.

The emission to environment can be estimated from equation (5a). Assumptions in the scenario calculation are as follows: layer thickness 1 m thick, infiltration rate 300 mm/year, L/S 20 and time

horizon 100 years. The total content can also here be used to describe the upper limit for future environmental burden.

5.8.2 Recycling of bricks, tiles & ceramics in unbound road layers

The following three emission scenarios have been studied: During demolition; during storage; from use in the final road.

Work stage	Exposure pathway	Target for exposure
Demolition	Dust (causal)	Inhalation, ingestion, dermal
Crushing	Dust (causal)	Inhalation, ingestion, dermal
Loading	Dust (causal)	Inhalation, ingestion, dermal
Transport	Dust (causal)	Inhalation, ingestion, dermal
Storage (heap)	Dust (causal), leaching	Inhalation, ingestion, dermal, water, soil
Laying, compaction, shaping	Dust (causal), leaching	Inhalation, ingestion, dermal, water, soil
Use and maintenance	Dust (causal), leaching	Inhalation, ingestion, dermal, water, soil

Figure 5-1 Emission pathways from recycling of recovered bricks, tiles & ceramics in unbound road layers.

Emission pathways during demolition: Here we can assume that the emissions to air are the same as for demolition of concrete buildings.

Emission pathways during storage: Here we assume that the main release is into water and negligible amount to soil. However, if the waste is stored for a long time, the discharge and the potential collection system of discharge water from the heap need to be evaluated and taken into account. The waste heap is here assumed to be wetted to hinder dusting.

Emission pathways from use in the final road: Here we assume that the main emission is to water or soil. However, emissions to soil and groundwater are strongly influenced by the amount of water infiltrating the road and hence the waste material. The emission to air can be neglected, because the sub-base is covered by other layers.

The emission to environment can be estimated from formula (5a). Assumptions in the scenario calculation are as follows: L/S 7 and time horizon 100 years (recycling of bricks in sub-base not realistic, a second recycling period not included).

5.8.3 Recycling of bricks, tiles & ceramics in concrete production

This option is not possible to study further because of limited information in literature about the environmental properties of concrete containing crushed bricks. Based on VTT's previous experience with concrete blocks, it can be assumed that the release will be low. The main release will probably be from leaching of sulphate. However, sulphate is also leachate from other aggregates in concrete.

5.8.4 Disposal/landfilling of bricks, tiles & ceramics waste

Here it is assumed that the calculated release for backfilling is also valid for landfilling. In case of landfilling the bricks are probably mixed with other demolition waste, such as insulation materials.

5.9 Comparison of scenarios

The release of selected substances in two scenarios is illustrated in Figure 5-2. The leached amounts are highly dependent on the achieved L/S ratio in the scenario. In the sub-base scenario, the water infiltration is limited and therefore leading to lower release. It needs also be reminded that the figures are based on data from crushed new bricks. Moreover, for the sub-base scenario, the leached amount of sulphate, arsenic and vanadium are overestimated because the brick waste are always mixed with concrete in the sub-base layer (brick amount is limited to 30 %).

The scenario calculations indicate clearly that in constructions with restrictions for water infiltration the release is lower. Consequently, when bricks are used in high-grade construction works the release will be lower compared to backfilling (*e.g.* use in noise barrier) in a given time frame. However, it has to be stressed that in the assessment of risks to environment also the conditions at the site (*e.g.* distance to important groundwater wells) need to be taken into account. Another aspect also to be pointed out here is that if the time perspective would be very long, the total environmental burden in both scenarios would end up to the same release levels.

Figure 5-2 Release of sulphate, vanadium and arsenic in two scenarios: sub-base (water infiltration low) and backfilling (water infiltration 300 mm/year).

6 Track ballast

6.1 Description and uses of track ballast

Track ballast (or railway ballast) forms the track bed upon which railway sleepers are laid (Figure 6-1). Track ballast is typically made from crushed stone / crushed natural rock of approx. 28 to 50 mm. The track ballast enables water to drain from the track, holds the track in place and distributes the weight of the train from the track structure to the subgrade. Track ballast should therefore be strong, hard-wearing, stable and drainable. The thickness of the layer of the track ballast depends among other factors on the size and spacing of the railway sleepers and the amount of traffic.

Figure 6-1 Example of track structure (Wikipedia).

EN 13450, Aggregates for railway ballast, specifies the properties of aggregates obtained by processing natural or manufactured materials or recycled crushed unbound aggregates for use in construction of railway track. For the purposes of this standard, the aggregate is referred to as railway ballast.

6.2 Generation of track ballast as a waste fraction

Two general scenarios for the generation of track ballast waste can be described. The waste fraction is typically generated during track ballast cleaning. However, during the renovation and modernisation of existing rail tracks (including replacement of the track structure and/or track foundation), a waste fraction will be generated as well. In general, two waste fractions can be distinguished:

- Residual fraction or low size fraction from ballast cleaning; contains typically crushed gravel/ballast, large proportion of fine particles, soil, sand)
- Track ballast, from demolition or excavation

Ballast cleaning machines are typically employed, but practises for ballast cleaning may vary between different countries. Generally, the cleaning machines excavate the track ballast, remove aggregate particles with grain size below about 30 mm and return the remaining coarser particles into the track. New coarse-grained ballast is added to replace the removed used ballast and the waste generated is handled somehow.

In Denmark only virgin materials are used when rail tracks are renovated/modernised and track ballast needs to be replaced (Banedanmark, 2012). Maintenance of the Danish railway network by Banedanmark (2011) resulted in some 174 000 tonnes of used ballast and track ballast in 2011 of which around 99 % were re-used or recovered. The remainder consisting primarily of used ballast and soil was landfilled.

In Finland, typically around 50 % of the mass is removed with ballast cleaning machines and placed on embankments. There is no data available on the environmental properties of the used ballast, only on technical properties (grain size distributions etc.) (Wahlström, 2013).

In Norway track ballast is re-used or landfilled (Haugland & Eggen, 2007). Apart from the geometric and physical requirements, track ballast for re-use must fulfil total content limit values, *e.g.* for PAH¹⁷ in order to be used without restrictions. The level of analysis will depend on prior knowledge of potential pollution of track ballast, *i.e.* there is no need to test for content of PAH in new track ballast material, as it can be regarded as uncontaminated. See more information in Section 6.3.

In Sweden it has been estimated that 75 000-100 000 m³ of track cleaning waste and 150 000-200 000 m³ of shoulder ballast cleaning waste is generated each year by treatment of 150 to 200 km of tracks with ballast cleaning machines (Hedström, 2002). The track cleaning waste generated represents about 20-25 % of the total track ballast treated, while the remaining 75-80 % (675 000-1 200 000 m³) fulfill the technical requirements and are re-used in the ballast cleaning process for the same purpose for which they were conceived.

Depending on the level of activity for track cleaning and track renewal the demand for fresh ballast will vary. Also the amounts of waste generated will depend on the number of rail projects carried out and the practise applied for track ballast cleaning and/or track renovation/modernisation. Since data for the Nordic countries is sparse an example is given for Austria/Germany:

In Austria track rehabilitation machines are employed that allow for integrated ballast recycling. This rail-mounted excavation of ballast means that excavated ballast is crushed and returned to the track together with new material. Some machines combine pre-cleaning/screening, ballast sharpening, screening and washing.

The Austrian rail network is 5 672 route-km and an annual amount of 400 000 tonnes of excavated material is generated. According to Becker & Vierlinger (2007) this results in a requirement for 500 000 to 700 000 tonnes new track ballast per year.

The German rail network is 64 655 route-km and according to Becker & Vierlinger (2007) around 2 200 to 2 500 track-km is renewed each year. This results in a requirement for 4 million tonnes new track ballast per year.

6.2.1 Track ballast cleaning

Track ballast needs to be maintained and this process typically produces a residual fraction or low size fraction which contains crushed gravel/ballast, large proportion of fine particles, soil, sand, humus, plant parts etc.

Maintenance typically covers weed control and cleaning, so that drainage, bearing and structural requirements are met. A standard from the Danish Railway¹⁸ (Banedanmark, 2006) contains requirements for maintenance of ballast layer, criteria for initiation of ballast cleaning and requirements for quality of the crushed stone after ballast cleaning. Similarly, the standard contains handling requirements of oversize fractions and fines and references to requirements concerning quality of supplied ballast in case of ballast supplement.

The standard states that maintenance of the track should be carried out when the proportion of particles below 22.4 mm constitutes minimum 30 % by weight (on average) of the layer of track ballast that is below the level of the sleepers.

During maintenance soil, gravel and crushed ballast are typically removed from the worn/used track ballast. By screening, stones that are below acceptable sizes (the fines) for track ballast are removed. The used ballast is a very heterogeneous material, consisting of 23-89 % of fines below about 23 mm (Banedanmark, 2012). The oversize material is typically returned to the track along with the screened track ballast.

¹⁷ <https://trv.jbv.no/wiki/Overbygning/Prosjektering/Ballast>

¹⁸ BN2-4-1 <http://www.bane.dk/db/filarkiv/5033/BN2-4-1.pdf>

Kral & Brunner (2012) have estimated material flows for track ballast handling. They have assumed that the renewal of the track bed of one kilometre of track results in 3 300 tonnes of track ballast being dismantled. Around 55 % of this is re-used, while the remaining 45 % of used track ballast (about 500 tonnes of fine material and 1 000 tonnes of “technically useless track ballast”) have to be disposed of.

As a result of weed control and many years of train traffic railway tracks are often polluted. Examples of pollutants that require special treatment and possibly disposal of the ballast include¹⁹:

- aliphatics from loss of fuel and lubricating agents;
- polycyclic aromatic hydrocarbons, for example creosote in timber sleepers;
- heavy metals caused by wear of rails, wheels, brakes and catenary;
- residues from herbicides which may include chemical substances such as atrazine, amitrol, bromacil, dalapon, diuron, ethidimuron, glyphosate, MCPA, picloram, and simazine.

Thus worn/used track ballast may be too polluted to be re-used/recycled (directly). A number of track-in techniques exist, that allow for cleaning/screening directly on the tracks. Water washing method for the remediation of polluted ballast gravels is widely used in many countries.

The low size fraction can be used in the production of asphalt where it replaces gravel since the materials can be treated in asphalt plants²⁰. In the asphalt plants’ drier drum pesticides and oils are destroyed.

6.2.2 Modernisation and renovation of tracks

During modernisation and renovation of tracks waste is generated in the form of

- Used ballast (residual fraction or low size fraction from ballast cleaning; contains typically crushed gravel/ballast, large proportion of fine particles, soil, sand)
- Track ballast, e.g. from the subgrade
- Subsoil/natural ground

Modernisation and renovation often involves the replacement of the track ballast, which means that ballast & sub-ballast, blanket (if included in the structure) and subgrade are dug up and replaced. In cases where the underlying subsoil/natural ground does not satisfy requirements to bearing strength, it may be necessary to replace this layer as well.

6.3 Legislation and guidance regarding recycling of track ballast

Here is described the Nordic legislation and guidance specific for recycling of track ballast. If there is no specific legislation, the general waste and environmental legislation will apply. Refer to Chapter 2 for general information on the legislation on C&D waste recycling and to Annex C for European harmonized standards.

Denmark

The Danish Statutory order no 1454/2012 (and Statutory order no 486/2012) is implementing the provisions of Directive 2008/1/EC of the European Parliament and of the council of 15 January 2008 concerning integrated pollution prevention and control. The Statutory order lays down the rules for granting of permits for installations. Re-use of track ballast in asphalt plants is covered by the statutory order; however the scope is limited to track ballast from railroad tracks *outside* the station areas and marshalling areas. This might be regarded as a potential barrier for the re-use of track ballast material in asphalt production, since the re-use in asphalt production of track ballast recovered *within* station and marshalling areas would require a relaxation of the standard terms.

The standard from the Danish Railway (BN2-4-1) (Banedanmark, 2006) contains requirements for the quality of the crushed stone after ballast cleaning. Among others there are requirements on particle size (Table 6-1).

¹⁹ <http://www.railwaygazette.com/news/single-view/view/integral-ballast-recycling-contributes-to-sustainable-development.html>

²⁰ <http://asp.vejtid.dk/Artikler/2002/04%5C3223.pdf>

Finland

No specific legislation or guidance was found.

Norway

The technical specification for construction and design of tracks contains some requirements on track ballast. The specification includes geometrical and physical requirements and refers to "Jernbaneverkets spesifikasjon for ballastpukk"²¹.

Jernbaneverket's environmental handbook (Jernbaneverket, 2011) contains a procedure for track ballast handling. The scope of the procedure is to ensure that track ballast is handled in an environmentally responsible manner. Prior to and after ballast cleaning representative samples of track ballast are to be taken and analysed for metals (As, Pb, Cd, Cu, Cr and Cr(VI), Ni, Zn) and organic pollutants (PAH and oil). Track ballast, with a content of relevant parameters below the legal limit value ("normverdiene i vedlegg 1 til kap. 2 i forurensningsforskriften"), can be disposed of freely within the planning and building act. If limit values are exceeded, material can only be re-used after a risk assessment has been carried out and/or approval by authorities has been obtained.

Sweden

The Swedish specifications for track ballast quality in tracks after ballast cleaning are published in BVF 585.51 (STA, 1998). They include requirements on aggregate size (Table 6-1), excavated surface slope and handling of the waste after ballast cleaning. Material and performance requirements on track ballast at delivery are given in BVF 585.52 (STA, 2004c). Suitable handling of the track ballast waste should be chosen and carried out after investigation of the environmental properties, such as waste class and hazardousness. The waste must not be placed on shoulders, slopes or drainages and not close to ditches. This is due to the frost activity and low draining properties of the waste.

Table 6-1 Danish and Swedish requirement on particle size after ballast cleaning.

Sieve size (mm)	Percentage passing by weight	
	Denmark ^a	Sweden ^b
11.2	≤ 2	≤ 2
22.4	≤ 6	≤ 6
31.5	≤ 25	≤ 16
80.0	= 100	= 100

^a Banedanmark, 2006.

^b STA, 1998.

6.4 Recovery operations

Track ballast is an aggregate and can as such be re-used where there is a need for aggregates/gravel. The residual fraction/low size fraction can be treated/cleaned before further processing. The track ballast can be crushed to the desired grain size to replace aggregates of a specific particle size. The suitability of an aggregate for a particular purpose will of course depend on the physical, mechanical or other properties that are of importance for that particular purpose.

The following general recovery scenarios can be described:

1. waste resulting from track ballast cleaning,
 - re-use as track ballast
 - recycling as asphalt aggregate
 - re-use/recycling as aggregate in unbound applications
 - backfilling
2. waste resulting from modernisation and renovation of rail tracks:
 - re-use as track ballast
 - recycling as asphalt aggregate

²¹ <https://trv.jbv.no/wiki/Overbygning/Prosjektering/Ballast>

- re-use/recycling as aggregate in unbound applications
- backfilling

The re-use/recycling applications for waste resulting from track ballast cleaning and waste resulting from modernisation and renovation of rail tracks are the same, as seen above. Therefore, the term “track ballast” is used for both in this report.

The remaining fractions – i.e. unsuitable in size or physical/mechanical properties or exceeding specific pollution levels – may need to be managed separately, for instance by landfilling.

6.4.1 Selected recovery operations for studies on emissions and impact on transports

Re-use as track ballast

Track ballast can be re-used provided that geometric and physical requirements as well as any limit values for pollution content are met.

Only a proportion of the track ballast removed during ballast cleaning can be re-used on the track directly, provided it meets relevant quality requirements. The remaining material can in principle be re-used, *e.g.* when replacing track structures and track foundations and during the establishment of new tracks. 100 % of virgin material can be replaced. 1 kg virgin gravel can be said to be substituted by 1 kg of track ballast (of suitable size and properties)

Recycling as aggregate in asphalt production

Track ballast can substitute virgin aggregates (gravel) in the production of asphalt. 100 % of virgin material can be replaced. 1 kg virgin gravel can be said to be substituted by 1 kg of track ballast (of suitable size and properties).

There might be national differences with respect to what type of asphalt products recovered track ballast can be used for. In Denmark, for instance, track ballast is not used for the production of asphalt for surface or wearing layers. For this application granite is used.

Re-use as aggregate in unbound applications

Track ballast can substitute virgin aggregates in unbound applications, such as unpaved roads, constructional layers etc. In that case 100 % of virgin material can be replaced. 1 kg virgin material can be said to be substituted by 1 kg of track ballast (of suitable size and properties).

Backfilling (use as filling material)

Track ballast can replace other aggregates/soil/mineral residues in backfilling operations. How much filling material that can be substituted by backfilling with track ballast will depend on the filling materials' properties as compared to track ballast.

6.5 Disposal/landfilling of track ballast waste

Remaining fractions after ballast cleaning – i.e. unsuitable in size or physical/mechanical properties or exceeding specific pollution levels – may need to be managed separately, for instance by landfilling. However, if track ballast is to be landfilled, the material has to undergo basic characterisation testing according to the EU Council Decision 2033/33/EC²² and the criteria for acceptance of waste at landfills will have to be fulfilled.

Information on track ballast management from Denmark and Norway shows that track ballast is landfilled. In Denmark only used track ballast is handled this way. Detailed information for Norway and the other Scandinavian countries is lacking.

²² COUNCIL DECISION of 19 December 2002 establishing criteria and procedures for the acceptance of waste at landfills pursuant to Article 16 of and Annex II to Directive 1999/31/EC (2003/33/EC)

6.6 Resources saved and impact on transports

Use of resources and emissions: Re-use and recycling of track ballast saves natural resources (virgin aggregates). See also Section 6.4.1. The extraction of virgin aggregates can be associated with hydro-geological impacts, *e.g.* when aggregates are extracted from below the water table. Quarrying operations and extraction of aggregates give rise to dust and noise and affect land-use and biodiversity.

Substitution of virgin aggregates with recycled aggregate materials decreases the rate by which the natural aggregate reserves are exploited. This reduces the impacts on land-use and biodiversity associated with extraction processes. Furthermore there are potential energy and pollution savings related to material substitution due to shorter transportation.

Transport: Aggregates are extracted and produced in a number of European countries, including Denmark (51 Mio tonnes), Finland (89 Mio tonnes), Norway (77 Mio tonnes) and Sweden (81 Mio tonnes) (UEPG, 2013b). See also Table 1-1 in Chapter 1.

To assess the impact on transport and possible energy and pollution savings for different re-use/recycling scenarios in detail, the actual geographical location of actual quarrying sites, production and processing sites, landfills and backfilling sites would have to be known. In this context the impact on transport distances is evaluated in relative terms, *e.g.* “Which process causes additional transport?” This is illustrated by Figure 6-2 below, where the length of the blue arrows indicates travel distances relative to each other.

Figure 6-2 Illustration of transport distances for extraction and use of virgin aggregates as compared to track ballast which is re-used on site, elsewhere, recycled/re-used in asphalt production or unbound applications, used in backfilling or landfilled. The length of the blue arrows indicates transport distances.

Assuming that track ballast is produced from crushed rock, it can be argued that only Denmark will have to import virgin aggregate material (*e.g.* from Norway) to ensure a supply of track ballast. It can be assumed that in Finland, Norway and Sweden natural aggregates will be transported locally/within the country to ensure a supply of track ballast. Thus, transport distances can be argued to be longest for natural aggregates from Nordic countries to Denmark.

Re-use of track ballast on site will result in the shortest transportation distances. All other re-use/recycling alternatives for track ballast will result in local/national transport of the material and can be assumed to result in comparable transport distances.

6.7 Pollution content and release from recovered track ballast

In general, track ballast can be polluted by a number of different substances from different sources and activities (Table 6-2). Examples of pesticides/herbicides include substances such as atrazine, amitrol, bromacil, dalapon, diuron, ethidimuron, glyphosate, MCPA, picloram, and simazine.

Table 6-2 Possible pollutants in track ballast and their sources (Ottesen & Haugland, 2007; Burkhardt et al., 2008*).

Source	Substance
Tracks and wheels	Mn, V, Co, Mo, Ni, Cr
Brakes*	Fe, Cu, Mn, Cr, Ni, Pb, Sb, binder ^a
Wheel bearings	Pb, Sb, Sn
Railway axles	Cu, Sn, Pb
Lubricants	Heavy oils, Ca, Al, Ba
Contact wires	Cu
Galvanised equipment	Zn
Signal lights, track switches	Hg
Diesel and oil	Oil, PAH
Sleepers	Creosote
Spraying of weeds	Pesticides/herbicides
Aggregates and gravel	Varying concentrations of all 92 naturally occurring elements. Concentration depends on rock type. No organic environmental pollutants.

^a Composite brakes contain a binder. Ingredients in this binder could not be established by Burkhardt et al. (2008).

The level of pollution may vary for different parts of the tracks, *e.g.* close to stations and areas for maintenance of trains as compared to free stretches. There may also be a difference in pollution content of track ballast depending of the age of trains used. Newer trains that are used nowadays are improved on vital points and this reduces emissions/pollution from train traffic. Furthermore, the period of time since the last ballast cleaning has been conducted may have an influence on the level of pollution to be found.

Railway operations are associated with the emission of inorganic and organic substances into the environment. The release and spread of pollutants from track ballast to the subgrade and to the underlying subsoil will depend on the nature of the pollutants – their properties with respect to solubility, degradation, sorption and mobility etc. This will strongly influence the degree to which pollutants can be anticipated to be found in the track ballast layer or leach to groundwater, soil and surface water.

Type and concentration of pesticides used for weed control will depend on the chosen strategy and what substances are approved by the authorities. There is typically no need for weed control on newly established stretches of rail way tracks. During the lifetime of ballast (approx. 25-35 years) weeds will start to grow especially along the shoulders of the track which will make weed control necessary. Refraining from spraying with pesticides, will shorten the lifetime of ballast and trigger the need for ballast cleaning, which is a costly operation.

The Swedish Transport Administration has gathered and evaluated a number of investigations performed regarding the pollution situation in railway embankments (Backman & Löwegren, 2004). The study included five investigations with about 650 samples in total and parts of the results are presented in Table 6-3. The authors concluded that arsenic and carcinogenic PAHs dominated the pollution situation. Copper and lead were also found in elevated concentrations, as well as pesticides. Less common were chromium, zinc, cadmium, PCB, phenol and petroleum hydrocarbons, while mercury and brominated flame retardants were very rare.

Table 6-3 Summary of analytical results from Swedish investigation of track ballast. Samples origin from the railway embankment along various rail stretches. Data in brackets refer to samples from stations and areas for train maintenance. (after Backman & Löwegren, 2004)

Substance	N	Median (mg/kg TS)	Arithmetic mean (mg/kg TS)	Guideline value (mg/kg TS)			Share of samples at or above guideline value	
				KM ^a	MKM ^b	STA ^c	KM ^a	MKM ^b
Arsenic	166 (129)	6.5 (6.1)	18.2 (16.1)	15	40	15	31 % (20 %)	8 % (9 %)
Chromium	192 (128)	18.0 (14.1)	22.4 (20.7)	120	250	120	0 % (2 %)	0 % (0 %)
Copper	185 (122)	35 (26.6)	101.4 (99.5)	100	200	100	19 % (24 %)	9 % (13 %)
Lead	184 (134)	19 (21.8)	57.0 (87.3)	80	300	80	12 % (16 %)	6 % (7 %)
Zinc	192 (118)	49.9 (44.3)	106.1 (105.5)	350	700	350	3 % (8 %)	3 % (3 %)
Carcinogenic PAHs ^d	151 (62)	0.83 (0.6)	3.68 (5.27)	0.3	>7	0.3	72 % (71 %)	14 % (11 %)
Not carcinogenic PAHs ^e	114 (65)	2 (0.9)	8.58 (9.27)	20	>40	20	7 % (6 %)	3 % (3 %)
Pesticide	N	N^f						
Phenoxy acids	9	0	0	0.004				
Glyphosate ^g	4	2	0.5	15.0				
AMPA	8	3	0.003	0.66				
Diuron	44	38	0.066	0.39				
Demethylated diuron	20	17	0.05	0.16				
Atrazine	9	5	0.02	0.056				

^a KM: Swedish guideline value for sensitive land use (point sources) (SEPA, 2009)^b MKM: Swedish guideline value for less sensitive land use (point sources) (SEPA, 2009)^c STA guideline value for use without restrictions (STA, 2002)^d The PAH-7 group.^e The PAH-16 group excluding the PAH-7 group.^f N: Number of samples with concentration at or above detection limit.^g One analysis with high concentration indicates point source.

Backman & Löwegren evaluated the variations of pollutant concentrations as well and found that large variations occurred along track stretches. The variations were not due to the age of the embankment, but to the ballast material and the frequency of ballast cleaning etc. Track stretches and railway yards (stations and maintenance areas) showed the same pattern for all substances except oil, which was most likely caused by point sources. Sampling at various depths from the track surface showed that metal concentrations were highest at the level of the sub-ballast surface, where the coarse aggregates meet the more fine-grained aggregates, and then decreased by depth. Sampling across the embankment showed that concentrations were highest in the track centre and then decreased by distance.

Pollutants are adsorbed to fine particles and organic material. When removing the fine fractions a large part of the pollutants are removed. In the ballast cleaning process particles finer than 32 mm are removed, but the remaining material (> 32 mm) might be covered with fine particles and elevated PAH concentrations may remain in the cleaned ballast.

The Geological Survey of Norway carried out a project for the Norwegian National Rail in 2007 with the purpose to gather and evaluate data on substances found in track ballast (Haugland & Eggen, 2007). Although a number of datasets have been collected by the authors, the accompanying information on sampling, pre-treatment of samples and which fraction (i.e. particle size) of the track ballast was analysed was very sparse.

Total content: As can be seen from Tables 6-3 and 6-4 the analytical results were compared to Swedish and Norwegian limit values. If limit values are exceeded, material can only be re-used after a risk assessment has been carried out and/or approval by authorities has been obtained.

A number of metals exceed the limit values and carcinogenic PAH, especially benzo(a)pyrene, seems to represent the largest pollution problem for the samples analysed.

Table 6-4 Summary of available analytical results from investigation of track ballast (mg/kg). Numbers in parentheses are calculated on the basis of results from Ofotbanen and results from Lieråsen tunnel are excluded. (Haugland & Eggen, 2007)

Substance	N	Median	Arithmetic mean	Minimum	Maximum	Limit value (revised) ^a	Number of samples exceeding limit value
Arsenic	148 (127)	3.2 (2.8)	5.0 (3.8)	0.4 (0.4)	33 (18)	8	17% (8%)
Lead	152 (131)	13 (12)	52 (50)	2.4 (2.9)	4400 (4400)	60	7% (5%)
Cadmium	152 (131)	0.13 (0.13)	0.40 (0.40)	< 0.055 (< 0.055)	16 (16)	3	1% (2%)
Copper	152 (131)	34 (31)	85 (47)	7.9 (7.9)	880 (340)	100	19% (8%)
Chromium	152 (131)	15 (14)	64 (18)	1.4 (1.4)	2000 (94)	35	20% (8%)
Mercury	108 (87)	0.011 (0.009)	0.033 (0.025)	0.001 (0.001)	0.46 (0.33)	1	- (-)
Nickel	152 (131)	13 (12)	22 (15)	2.4 (2.4)	190 (54)	60	4% (-)
Tin	152 (131)	48 (47)	75 (59)	18 (18)	1200 (240)	100	14% (11%)
Benzo(a)pyrene	150 (129)	0.08 (0.06)	0.21 (0.15)	< 0.01 (< 0.01)	2.3 (2.3)	0.01	85% (83%)
Naphthalene	150 (129)	0.05 (0.005)	0.04 (0.02)	< 0.01 (< 0.01)	2.8 (0.68)	0.8	0.01% (-)
Fluorene	150 (129)	0.01 (0.01)	0.87 (0.02)	< 0.01 (< 0.01)	63 (1.1)	0.8	0.03% (0.01%)
Fluoranthene	150 (129)	0.23 (0.20)	8.24 (0.49)	< 0.01 (< 0.01)	720 (4.6)	1	21% (12%)
Pyrene	150 (129)	0.215 (0.18)	5.61 (0.47)	< 0.01 (< 0.01)	490 (4)	1	21% (14%)
Sum of 16 PAH	150 (131)	1.4 (1.2)	22.93 (2.6)	< 0.2 (< 0.2)	1800 (30)	4	24% (14%)
Hydrocarbon (Total)	152 (131)	20 (20)	100 (40)	< 20 (< 20)	3000 (850)	100 ^a	10% ^b (8%) ^b

^a Norwegian Environment Agency limit values for contaminated soil, developed in 1999 and revised in 2007

^b result for C₁₆₋₃₅ fraction
N = number of data set.

In the Norwegian study the natural content of arsenic and heavy metals in soil along the rail network was compared with the content found in track ballast. Data suggested that the total content of arsenic and heavy metals in track ballast is at the lower end of the total content found in soil. However, the content of copper (based on the median) in track ballast exceeds the level of copper found in soils. Limit values as presented in Table 6-4 are not exceeded (Haugland & Eggen, 2007).

PCB, pesticides, BTEX and asbestos in track ballast: Haugland & Eggen (2007) report that PCB has been detected in 11 out of 46 samples. Levels of PCB (sum of 7 PCB) are generally low. Only three samples exceeded the Norwegian STF limit value of 0.01 mg/kg, and the highest content measured was 0.029 mg/kg.

The content of pesticides was analysed in 19 samples of track ballast and could only be detected in three samples. The highest content of 0.14 mg/kg was detected for dieldrin.

6 Track ballast

A total of 34 samples were analysed for the content of volatile organic compounds (VOCs, BTEX - benzene, toluene, ethylbenzene, and xylenes). BTEX could only be detected in three samples (xylene below 0.5 mg/kg; benzene in two samples – 0.96 mg/kg and 0.1 mg/kg, respectively).

Six samples were analysed for asbestos but nothing was detected.

Creosote: Wooden sleepers impregnated with creosote can give rise to pollution of track ballast. Creosote consists of a large number of different substances, *e.g.* PAH, heterocyclic hydrocarbons, phenolic compounds. Components of the creosote can be released from sleepers by volatilization, exudation (oozing out), leaching and abrasion (Burkhardt et al., 2008).

A project similar to the one carried out by Haugland & Eggen (2007) was done by the Danish National Rail in 2010. Focus in the Danish project was slightly different. Data was gathered with the purpose to determine the level of pollution along existing railway tracks in order to predict the level of pollution that could be expected for a new stretch of railway tracks. Soil samples were taken and analysed for a number of parameters. The pollution level in the soil was used as an indicator for the potential of soil, surface and groundwater impact by pollutants. Soil samples were analysed for total hydrocarbons (and fractions), PAHs and six metals. Out of 488 soil samples analysed a total of 429 samples were classified as clean soil. The remaining 59 samples were classified as class 2 soil (36), class 3 soil (15) and class 4 soil (8), respectively. The parameters that triggered this classification were: hydrocarbons (>C10-C25, >C25-C35 and total hydrocarbons >C5-C35), PAHs in the form of benzo(a)pyrene, lead, cadmium (geogenic, *i.e.* naturally occurring) and zinc.

The graduation in class 1 to 4 soil follows the following principles:

- Class 1: free re-use of soil
- Class 2: slightly polluted soil that can be re-used in construction work once an environmental permit is obtained
- Class 3: polluted soil to be cleaned up or landfilled
- Class 4: very polluted soil to be cleaned up (to some degree) and landfilled subsequently

Leaching tests:

No Swedish data from standardized leaching tests were found.

Haugland & Eggen (2007) present the results from leaching tests performed on four samples of track ballast (Table 6-5). However, the data is not accompanied by sufficient information on sampling, pre-treatment of samples and which fraction (*i.e.* particle size) of the track ballast was tested. Furthermore, information on the type of leaching test used, the L/S ratio used in the test and the analytical method employed is lacking. The results are therefore useless for risk assessment purposes.

Data indicates that Chromium (VI), PAH, oil – and benzo(a)pyrene to some degree – may represent a problem with respect to leaching from track ballast.

Table 6-5 Results from leaching tests (Haugland & Eggen, 2007).

Parameter/ site	Ofofbanen (Bjørnefjell st.)	Ofofbanen (Pettersenfyllinga)	Sandvika	Sandvika
As (mg/kg)	< 0.05	< 0.05	<	<
Pb (mg/kg)	< 0.05	< 0.05	<	<
Cd (mg/kg)	<0.04	<0.04	<	<
Cu (mg/kg)	<0.2	<0.2	0.054	0.045
Cr(VI) (µg/kg*)	0.13	0.57	<	<
Hg (µg/kg)	<0.02	<0.02	<	<
Ni (mg/kg)	<0.04	<0.04	<	<
Zn (mg/kg)	<0.4	<0.4	0.082	0.076
Benzo(a)pyrene (µg/kg)	0.04	<0.01	<	<
PAH (µg/kg)	9	2.3	<	0.7
Oil (µg/kg)	59	68	<50	<50

Methods for pollution content and release: Determination of meaningful measurements of the content and leaching of potentially contaminating substances from coarse granular materials (aggregates) is a particularly difficult task, and for used track ballast it may be further complicated by the fact that it most often consists of relatively inert coarse particles fully or partially covered by a surface layer of contamination. If the materials are crushed and ground prior to chemical analysis, the contamination may be regarded as “diluted” by the mass of relatively inert material (mostly natural stones) making up the aggregate itself. The “content” of contaminants should probably be related to the surface area of the track ballast aggregates rather than the total mass. Ideally, an analytical method that extracts the organic and inorganic contaminants quantitatively from the surface of the particles should be applied, and the results reported as mg/m². It would then be necessary to estimate the specific surface area of track ballast aggregates based on the particle size distribution to assess the total “content” of contaminants in a given mass of track ballast.

Leaching will also be surface related, and probably mainly related to the layer of contaminants on the surface. Most of the existing leaching standards are aimed at describing either release of substances from fine granular materials under (local) equilibrium conditions or release by diffusion from monolithic materials (i.e. particles larger than 40 mm). Wash-off of substances from the surface will normally be included in the latter type of tests, but is often dismissed as a minor initial “disturbance”. Leaching from coarse granular materials - like track ballast - falls in between the two test types. Leaching is definitely surface-related, but primarily associated with the surface coating of contaminants. The best description of the release would most likely be obtained by using the so-called compacted granular leaching test which is part of the technical specification TS-2 developed by CEN/TC 351. This method will account for release of contaminants from both the surface layer and from diffusion from the core body of the particles. However, since the “surface wash-off” is generally regarded as an initial disturbance, it has not been subject to much interpretation as the main part of the source term in impact assessments. When such results are applied in a source term scenario, the release as a function of time (and exposed surface area) will most likely depend significantly on the hydraulic conditions, i.e. the mode of contact with water. Development of such models falls outside the scope of this study. For certain types of residual fractions of track ballast with a high content of fine materials it may be possible to assess the leaching properties using a percolation leaching test under local equilibrium conditions and perform model calculations of emissions from various scenarios as described elsewhere in the report.

For the above mentioned reasons and since detailed data on total content and leaching is lacking, it is not possible to estimate the release and spread of pollutants from track ballast. Only a general description of emission scenarios can be given in the following sections.

6.8 Emission scenarios and calculated release/spread of pollutants

Substances or groups of substances of interest include both inorganics and organics. As described above, pollution of track ballast can be argued to be surface related, and the release and spread of pollutants from track ballast depends on the pollutants’ properties. It may be assumed that pollution from coarse-textured track ballast to a certain extent is transferred to finer material and that the pollution content found in remaining coarse track ballast may be assumed to be lower than in the used track ballast, containing a large proportion of fine material. As a result of this, substances that are not degradable over time may accumulate in the fine fraction.

Metals: The release of metals to the environment by railway operations stems from abrasion processes (e.g. generated by braking processes). Depending on the composition of wheels, rails, brakes metals such as Fe, Mn, Cr, Cu, Ni are released. Data on the size distribution and fate of the emitted particles is lacking. The emitted particles may adsorb to track ballast and other particles and thus be immobilized. Pollutants may leach out via particle-facilitated transport. Particles itself may be toxic due to the particle size (Burkhardt et al., 2008).

If pollutants are leached out from the track ballast layer via particle-facilitated transport, they may adsorb again in the subgrade or subsoil. Soil properties, such as type of soil (particle size) and content of natural organic matter will largely influence the degree to which metals are adsorbed.

Organic pollutants/hydrocarbons: Depending on the nature and type of organic pollutant, pollution can be found as a separate phase, dissolved in water/pore water, adsorbed to particles or it can evaporate. Organic pollutants such as hydrocarbons and PAH's are potentially degradable. Depending on their water solubility organic substances can leach out. Consequently, it may be expected that highly water soluble substances will transfer from track ballast to the subgrade and subsoil by rainfall.

Pesticides: The presence of pesticides will depend on the degradation rate that is substance specific. Furthermore, water solubility and soil conditions will have an influence in the residual concentrations to be found. It may be expected that highly water soluble substances will transfer from track ballast to the subgrade and subsoil by rainfall.

6.8.1 Re-use of track ballast

As described above, it can be assumed that pollution from coarse-textured track ballast to a certain extent is transferred to finer material. Hence it could be argued that subgrade and subsoil as well as railway embankments are sinks for all released substances.

If quality criteria for the re-use of track ballast exist and they are derived from risk assessment, they presumably prevent unacceptable contamination of soil, ground water and surface water as a result of re-use activities. If this is not the case there may be a risk of spreading pollution if track ballast is re-used.

Track-in techniques for ballast cleaning that involve water washing methods can ensure that track ballast is cleaned prior to reusing it on the tracks. However, this is based on the assumption, that those water washing methods effectively remove any pollution (both contamination by metals and organic substances) from the ballast material. Spillage of used water from the washing process may result in the spread of pollutants to the track bed and track ballast.

6.8.2 Backfilling of recovered track ballast

If track ballast is used for backfilling operations, the total content of pollutants adsorbed to track ballast particles can in principle contribute to the release of substances to the surrounding environment. However, detailed data is lacking to estimate the release and spread of pollutants from track ballast. The type of pollution will have an influence on the degree of release and spread to the surrounding environment.

6.8.3 Recycling of recovered track ballast as asphalt aggregate in road bases

The following emission pathways have been studied for the recycling of track ballast as asphalt aggregate in road bases (Figure 6-3).

Figure 6-3 Emission pathways from recycling of recovered track ballast as asphalt aggregate in road bases.

Emission pathways during recovery

Particulate emissions are assumed to be dominating during recovery of track ballast. The degree to which emissions occur will depend on the type of recovery method used. Regular track ballast cleaning may result in dust emissions while the use of (water) washing techniques might give rise to emissions if spillage of the polluted water from the process occurs.

Emission pathways in the asphalt plant

Sieving and crushing of track ballast at the asphalt plant can give rise to particulate emissions to air and surrounding soil and surface water.

Pesticides, oils and other hydrocarbons contained in the track ballast material could be partly destroyed in the hot mix asphalt production. Asphalt plants are not designed for the destruction of organic contaminants and the efficiency will vary widely depending on the plant design. There are no data to support an estimate on the destruction efficiency. Metals will not be destroyed and can be assumed to be present in the asphalt mix leaving the asphalt plant.

Emission pathways during paving

Pesticides, oils and other hydrocarbons that are not destroyed during the asphalt production process will remain in the asphalt mix and may be emitted during paving activities, *e.g.* as emissions to air. This process will be comparable to emission pathways from asphalt in general, see Section 3.8. Recovered track ballast could be a source of pollution in asphalt recycling.

Emission pathways from the final pavement

When used in road bases emission pathways from the final pavement will be dominated by emission to soil and groundwater, since the road base will be covered by the wearing course thus preventing direct emissions to air.

Emissions will be depended on the transport of water through the wearing course and the road base. The release of organic and inorganic substances can be assumed to follow the release of inorganic and organic substances from asphalt in general, see Sections 3.7 and 3.8.

6.8.4 Recycling of recovered track ballast in unbound applications

Figure 6-4 illustrates the overall emission pathways from recycling of recovered track ballast as aggregate in unbound applications, such as unpaved roads, as a bearing layer or non-bearing layer.

Figure 6-4 Emission pathways from recycling of recovered track ballast as aggregate in unbound applications.

6 Track ballast

Emission pathways during recovery

Particulate emissions are assumed to be dominating during recovery of track ballast. Please see above – heading 6.8.3.

Emission pathways in treatment plant

Sieving and crushing of track ballast will not destroy organic or inorganic pollutants that may be present in the track ballast. Sieving and crushing can give rise to particulate emissions to air and surrounding soil and surface water and thus contribute to the spreading of compounds.

Emission pathways during installation

Particulate emissions may be expected during use of the material in unbound applications. If the material is exposed to rain during storage and installation, leaching of compounds adsorbed to the material surface may occur.

Emission pathways from the final application

When used in road bases and when covered by a wearing course emission pathways from the final pavement will be dominated by emission to soil and groundwater.

When used in unpaved roads emission pathways include both emission to air and emission to soil and groundwater. The latter will depend on the transport of water through the layer of recycled material and the release will be governed by the properties of substances present.

6.8.5 Disposal/landfilling of track ballast

Figure 6-5 illustrates the overall emission pathways from landfilling of track ballast.

Emission pathways during excavation

Particulate emissions are assumed to be dominating during recovery of track ballast. Please see above – heading 6.8.3.

Emission pathways during landfilling

Particulate emissions to air and surroundings may be expected during landfilling of track ballast. Emissions will be limited to the period where the material is placed at the landfill and daily landfill cover can be assumed to prevent spreading of particles.

The environmental impact from release of organic and inorganic is reduced by the installation of protective measures such as bottom liners and leachate collection systems.

Figure 6-5 Emission pathways for landfilling of track ballast.

6.9 Comparison of scenarios

Since detailed data on both pollution content and release is lacking for track ballast, it is not possible to estimate the release and spread of pollutants from track ballast. Only a general description of emission scenarios can be given.

Recovery operations that ensure re-use of track ballast directly on-site or nearby can be said to be favourable, as transportation of virgin material and thus emissions as a result of transportation and quarrying of virgin materials can be minimised.

As described in the sections above, it can be assumed that (highly) water soluble pollution from coarse-textured track ballast to a certain extent is transferred to finer material. Therefore recovery operations that

- a) ensure efficient cleaning and removal of any pollutant prior to re-use of material or
 - b) avoid or reduce contact to water and hence emission to soil and groundwater/surface water
- should be preferred if the release and spread of pollutants is to be avoided or minimized.

Moreover, if quality criteria for the re-use of track ballast exist and they are derived from risk assessment, they presumably prevent unacceptable contamination of soil, ground water and surface water as a result of re-use activities. If this is not the case there may be a risk of spreading pollution if track ballast is re-used.

7 Gypsum-based construction materials

7.1 Description and uses of gypsum-based construction materials

Most of the information on gypsum based construction materials is taken from Monier et al. (2011). The reader is referred to the original report for more detailed information.

Two types of gypsum (natural gypsum and synthetic gypsum) are used for a variety of applications. Whilst natural gypsum is extracted from open-cast or underground mines, synthetic gypsum is generated during flue gas cleaning processes at coal-fired power stations (flue gas desulphurisation – FGD gypsum). FGD gypsum is also commonly known as desulphogypsum (DSG).

Primary producers of natural gypsum are Spain (52 %), France (8.1 %), Germany (7.3 %), the UK (6 %), Italy (5.5 %) and Poland (5.2 %) accounting for 24 million tonnes of gypsum produced in 2008 (Monier et al., 2011).

Gypsum is used in a large number of construction types and typical applications include:

- Plasterboards (and modified plasterboards to meet specific construction requirements, such as fire resistance, humidity resistance etc.)
- Decorative plaster (typically plaster powder)
- Building plaster (used for walls and ceilings)
- Plaster blocks (used for partitions and gypsum tiles for ceilings)
- Gypsum based self-levelling floor screeds
- Gypsum fibreboards (used for partitions and the lining of walls, ceilings, roofs, floors)

7.2 Demolition of gypsum-based construction materials

Gypsum based materials are construction materials where “closed loop” recycling is possible, which means that the waste is used to make the same product again and not simply recovering it for some “down-cycling” application as is the case with construction materials such as concrete and bricks used for aggregates.

Monier et al. (2011) state that although gypsum is considered to be 100 % recyclable, only production and construction gypsum waste is currently recycled, whilst demolition waste is not part of recycling activities. Buildings older than 40 years typically contain plaster and only little plasterboard. When deconstructing old buildings plaster often sticks to bricks, which makes the recycling of gypsum difficult.

Gypsum construction waste consists typically of cut off pieces of plaster and boards as well as boards damaged during *e.g.* transport to the building site. However, contamination with other materials (*e.g.* paint, fastenings, screws, nails, wood and insulation materials) can make recycling of gypsum waste difficult.

Selective demolition of gypsum-base material is hard to carry out, which results in a large fraction of mixed waste – containing gypsum – that cannot be recycled properly.

7.3 Legislation and guidance regarding recycling of gypsum based construction materials

Refer to chapter 2 for general information on the Nordic legislation on C&D waste recycling and to annex C for European harmonized standards.

7.4 Recovery operations

Since gypsum is considered to be 100 % recyclable it can be argued that very close to 100 % of natural gypsum or synthetic gypsum can be substituted when using recycled gypsum in the manufacturing of new products.

Figure 7-1 Basic principles of gypsum recycling (EURO:Gypsum²³, 2012).

Gypsum Recycling International A/S <http://www.gypsumrecycling.biz> has developed and patented a recycling technology for plasterboards waste. The company accepts waste materials with up to 3 % contamination (gypsum blocks, virgin gypsum board cut-offs, complete boards or broken parts, gypsum ceilings/floors/walls, nails and screws in the plasterboard allowed, wallpaper, glass tissue and other wall coverings on the boards are allowed). By means of this new technology a gypsum recycling rate of 80 % (40 000 tonnes per year²⁴) can be achieved in Denmark. The majority of Denmark's 200 or so municipal recycling centres are customers of Gypsum Recycling, and today 20-25 % of the plaster used domestically comes from recycling.

The system developed by Gypsum Recycling ensures that plasterboard is stripped of its paper cladding and grinds the plaster to a powder of almost the same quality as virgin plaster. The finished product is 99 % as pure as ordinary plaster and this is good enough for making new plasterboard that is comparable to all the others, and is evidenced by the fact that the world's five biggest manufacturers of plasterboard – Lafarge, BPB, USG, National Gypsum and Knauf – all buy the product²⁵.

²³ <http://www.eurogypsum.org/Uploads/dbsAttachedFiles/livingwithgypsum.pdf>

²⁴ http://www.ecoinnovation.dk/Emneoversigt/Danske_cases_og_resultater/Danske_succeshistorier/Case_Gypsym_genanvendelse_gips.htm

²⁵ http://www.netpublikationer.dk/um/10138/html/printerversion_chapter06.htm

Gypsum Recycling International – The gypsum recycling system (www.gypsumrecycling.biz)

An on-site containerised collection system has been developed for containers located at new building sites, reconstruction and demolition sites, at waste transfer stations, civic amenities and other locations where waste is stored temporarily or sorted.

Specialised collection trucks remove the waste from the containers rather than removing the containers from the collection site. During the removal of gypsum waste by crane and grab, the material is to some degree crushed. The truck can collect waste from several collection sites, before it returns to the processing site.

When approx. 2000 tonnes of waste material have been collected at the processing site, the mobile recycling unit is moved to the processing site. Prior to processing, hand sorting for bigger pieces (metal, wood etc.) is done. The gypsum waste is put into the hopper of the recycling unit. During the process, where the gypsum waste is crushed and sorted, impurities are removed. It is critical to separate the impurities such as nails, screws, various kinds of wall coverings (e.g. wallpaper, glass tissue) from the gypsum powder. The powder is loaded onto containers and transported to a manufacturing site.

The system is capable of handling these impurities. Acceptable waste types include waste from rebuilds, demolitions and new construction, dunnage and underlayers (structural elements) and production scrap. Up to 3 % contamination of other materials can be handled by the system.

However, plastics, foils, insulation material, wood, grout/joint filler, etc. will have to be removed before the gypsum waste is placed in containers at the collection sites.

7.4.1 Recycling of plasterboards

During the first step of plasterboard recycling paper layers are removed as much as possible. The gypsum is crushed into powder which is returned to manufacturers. According to Monier et al. (2011) between 5 and 10 % of gypsum powder resulting from construction plasterboard waste is re-integrated in the system at European level (European average).

The crushed gypsum powder fraction typically contains a residual paper fraction which may hinder the improvement of the introduction rates of recycled powder into production processes.

Research and development aims at improving the manufacturing process by making it possible to recycle gypsum with a higher residual paper fraction. Other R&D activities aim at enhancing the techniques for the removal of paper from plasterboards.

Example of research activities: Gypsum Recycling International participates in an EU funded Life+ project aimed at increasing gypsum recycling. The project will run from 2013 to 2015 and the objective of the project is (GRI, 2013) *“to close the loop effectively and transform the plasterboard demolition waste market to achieve higher recycling rates of plasterboard waste. The major obstacle to close the loop is that buildings are currently demolished and not dismantled in the majority of the Member states of EU. This leads to unsegregated waste going to landfill without having the possibility to recover valuable recyclable materials, among others gypsum plasterboard waste. The manufacturers involved in the project will strive to include 30 % recycled gypsum in the plasterboard, also including construction and production waste.”*

There are large differences in recycling practices in the European member states. Some countries, such as Denmark with a comprehensive gypsum recycling scheme in place, achieve recycling rates that are very high (Table 7-1).

Møller et al. (2012) state that due to market competition, gypsum producers are reluctant to state how much of virgin (natural) gypsum is replaced when waste gypsum powder is used in the production of new plasterboards. The authors therefore assumed that 1 kg natural gypsum is replaced by 1 kg of recycled gypsum. Furthermore, Møller et al. (2012) assume that recycled gypsum exclusively replaces natural gypsum and does not compete with synthetic gypsum from power plants.

Table 7-1 Recycling rates for gypsum-based construction materials in the Nordic countries.

Denmark	<p>A recent Danish report (Møller et al., 2012) showed that approx. 54 000 tonnes of gypsum waste were collected in Denmark in 2009:</p> <ul style="list-style-type: none"> - Approx. 17 000 tonnes were re-use/recycled as gypsum powder (e.g. for plasterboards) - 5 500 tonnes were re-used in cement production, - 6 000 tonnes in compost and - approx. 25 500 tonnes were used as slag heap cover in Germany (see also next section about alternative recovery scenarios).
Finland	<p>In Finland there is currently no market for recovered gypsum due to the generation of FGD gypsum in wet desulphurization processes in coal firing plants. Gypsum wastes from constructions are currently landfilled. Gypsum boards are currently not separated on site but are mixed with other construction wastes (e.g. waste containing wood and organic materials). Due to the ban of organic waste to landfills, the management of gypsum waste is expected to change in near future.</p>
Norway	<p>A Norwegian guidance document on the management of gypsum waste has been issued (Tangsveen et. al., 2012). According to the document approx. 55 326 tonnes of gypsum waste were generated in 2009 and 2010. Statistics Norway (2013) reports that 57 835 tonnes of gypsum waste were generated in 2011. Some 49 348 tonnes (or approx. 85 %)^a were recycled/recovered while the remaining 15 % were landfilled. As the collection system and recycling technique is improving in Norway, the recovery of gypsum waste is expected to increase and the most favourable recovery scenarios will be material recycling.</p>
Sweden	<p>The amount of gypsum-based construction waste that was generated and is potentially recyclable was approx. 100 000-300 000 tonnes in 2010. The amount of gypsum-based construction waste that was recycled was approx. 20 000-25 000 tonnes in 2010. Since 2010, the SEPA regulations and guidelines have been clarified and it is anticipated that recycling has increased over the last two year. However, no recent statistical data is available. (Nygren, 2013)</p>

^a This data may be overestimated (Larsen, 2014).

7.4.2 Alternative recovery operations

Gypsum powder: Recycled gypsum can be used as a raw material in the manufacture of new gypsum-based products, e.g. plasterboard and coving. It can be assumed that 1 kg of virgin material is replaced by 1 kg of recycled material/synthetic material.

Cement production: Gypsum can be recycled as a raw material in the cement production where gypsum is used to modify the setting characteristics of cement, e.g. use of gypsum in Vitcement (Aalborg White), CEM I 52,5 N (CEMENTA AB, 2012). According to the product data sheet approx. 39.5 kg gypsum are used in the production of 1 tonne white cement. Møller et al. (2012) state that in Denmark, the demand for gypsum in cement production is met by the production of synthetic gypsum. However, in Sweden there is a demand for recycled gypsum as replacement of natural gypsum in the production of cement. It can be assumed that 1 kg natural gypsum is replaced by 1 kg of natural/recycled gypsum.

Compost/agriculture: Gypsum has traditionally been used as a soil treatment in agriculture. Gypsum is used to improve soil conditions by e.g. improving soil structure, drainage and water-holding capacity. Gypsum waste is also used in compost as nutrients and structural material. When used in compost/agriculture, gypsum typically replaces sulphurous fertilizer.

Additive in biomass combustion: Biofuels rich in silicon (Si) and/or potassium (K) such as straw and woody fuels (forestry residues, willow and bark) cause problems with slagging and fouling in conventional boilers. A Swedish pilot test showed that slagging and fouling was significantly reduced by addition of 1 % crushed gypsum plasterboards to K rich woody biofuels and 2-3 % for straw biofuels (Piotrowska et al., 2013) and results from full scale test are to be published shortly (Öhman, 2013). In Sweden, forestry residues (GROT) is an important source of K rich biofuels at ca. 1.5 Mton per year with a potential to double within the next 10 years, but only a small proportion of boilers is expected to encounter problems with slagging and fouling on a scale that triggers the addition of gypsum. If recovered gypsum waste would become a 1 % fuel additive at 10 % of K rich biofuels (logging residues) this use would correspond to 1 500–3 000 tonnes of gypsum waste per year, which is a small fraction of the gypsum waste generated in Sweden. Use of gypsum waste as an additive is an emerging technic and has not yet been chosen for life cycle studies on emissions and impact of

transports. It can be supposed though that the scenario used for recovery in soil treatment is fairly relevant.

Backfilling: It should be noted, that backfilling of gypsum is not a suitable solution from an environmentally point of view (protection of groundwater). Hence, this is not included in this report.

7.4.3 Selected recovery operations for studies on emissions and impact on transports

Recycling as gypsum powder in the manufacture of new gypsum-based products, e.g. plasterboards

Gypsum based waste material can be recycled and used in the production of new gypsum-based products.

Recycling as a raw material in the cement production (instead of natural gypsum)

Gypsum based waste material can be recycled and used in the cement production.

Recycling in soil treatment as nutrients and structural material (instead of sulphurous fertilizer)

Gypsum based waste material can be recycled and used in compost as nutrient and structural material.

7.5 Disposal/landfilling of gypsum-based waste

The EU Council Decision 2003/33/EC²⁶ is setting criteria for acceptance of waste at landfills. Article 2.2.3 in Council Decision 2003/33/EC reads as follows:

“Non-hazardous gypsum-based materials should be disposed of only in landfills for non-hazardous waste in cells where no biodegradable waste is accepted. The limit value for TOC [5 %] and DOC [800 mg/kg at L/S = 10 l/kg] given in sections 2.3.2 and 2.3.1 shall apply to wastes landfilled with gypsum-based materials.”
(2003/33/EC Article 2.2.3)

When Council Decision 2003/33/EC was written, it was decided that gypsum-based waste and waste with a content of biologically degradable organic matter which can potentially produce reducing, biologically active leachate, should not be placed in the same landfill cells because such leachates, when coming into contact with gypsum (calcium sulphate) or sulphate-containing leachate from the gypsum, could cause the reduction of sulphate to sulphide. To prevent this from happening, section 2.2.3 in 2003/33/EC specifies that any waste disposed of together with gypsum or gypsum-based waste should fulfil the requirements on TOC (max. 5 %) in section 2.3.2 and DOC (max. 800 mg/kg at L/S = 10 l/kg) in section 2.3.1. Although not directly specified, the requirement applies also to the gypsum-based waste itself.

The risk of formation of sulphide and hydrogen sulphide gas and the risk of groundwater contamination with sulphate are the main reasons why landfilling of gypsum waste is subject to specific requirements as given in EU Council Decision 2003/33/EC. In addition to the above mentioned restrictions, due to the risk of groundwater contamination, gypsum waste cannot be accepted at landfills for inert waste, because protective measures such as bottom liners and leachate collection systems are not required for this category of landfill at the European level.

From an environmental protection perspective (as described above) and from a resource recovery perspective landfilling of gypsum-based material can be regarded of lesser priority than the recycling of gypsum-based waste material. In backfilling there would typically be no protective measures such as bottom liners and leachate collection systems, and backfilling should not be an option. In Germany, gypsum-based waste materials have until recently been used as so-called contour layers as part of covering tailing heaps from potash mining (Kalihalden).

²⁶ COUNCIL DECISION of 19 December 2002 establishing criteria and procedures for the acceptance of waste at landfills pursuant to Article 16 of and Annex II to Directive 1999/31/EC (2003/33/EC)

7.6 Resources saved and impact on transports

Use of resources and emissions: The quarrying of natural gypsum gives rise to dust and noise generation. Dust arises from extractors and conveyor belts. Intermittent noise is produced by blasting activities and the use of crushers and screening equipment. The extraction of natural gypsum has an adverse effect on the environment, and the gypsum industry²⁷ aims at restoring biodiversity at quarrying sites.

Natural gypsum can be substituted by FGD gypsum and recycled C&D gypsum. This material substitution decreases the rate by which the natural gypsum reserves are exploited. This reduces the impacts on land-use and biodiversity associated with extraction processes. Furthermore there are potential energy and pollution savings related to material substitution due to shorter transportation.

Renovation and deconstruction activities as well as processing and sorting of gypsum waste at recycling stations, give rise to dust generation and noise. At building sites and waste processing sites dust generation and spreading can be controlled and contained by dust curtains and the facility itself.

If gypsum waste is landfilled, which is the case for a number of European countries, the waste will contribute to the generation of greenhouse gasses (methane) which are produced when paper from *e.g.* gypsum plasterboards degrades. Avoiding landfilling and recycling gypsum thus reduces the generation of greenhouse gasses associated with landfilling and reduces the rate for exploitation of natural gypsum resources.

Transport: Within the EU, natural gypsum can only be quarried in a few countries. Thus, distances for transport of natural gypsum to manufacturers of gypsum-based products are relatively long, especially for the Nordic countries.

However, to assess the impact on transport and possible energy and pollution savings in detail, the actual geographical location of actual quarrying sites, coal-fired power plants, collection and processing sites for gypsum waste as well as manufacturing sites would have to be known. In this context the impact on transport distances is evaluated in relative terms, *e.g.* “Which process causes additional transport?”

This is illustrated by Figure 7-2 below, where the length of the blue arrows indicates travel distances relative to each other. Transport distances can be argued to be longest for natural gypsum from EU-mining sites to Nordic countries, compared to transport distances of FGD gypsum and gypsum waste. The potential for energy and pollution savings due to shorter transportation will be higher in areas, where:

- Coal-fired power stations are located close to manufacturing sites for gypsum products, cement or soil improver (FGD gypsum)
- Collection of gypsum waste is carried out locally, close to gypsum waste processing facilities and near manufacturing sites for gypsum products, cement or soil improver (gypsum waste)

Monier et al. (2011) evaluated that the substitution of natural gypsum by recycle gypsum results in a net benefit. The net benefits are assumed to be higher if the recycled gypsum replaces the FGD gypsum, since there is a higher energy consumption associated to the manufacturing of plasterboards from synthetic gypsum due to its higher moisture content.

Recently, a Danish project was carried out with the purpose to analyse and quantify the potential environmental and economic costs and benefits of different treatment methods for gypsum waste (Møller et al., 2012). For this purpose a life cycle assessment (LCA) and an economic assessment have been carried out for the four treatment methods that are considered to be most important in Denmark:

- Plasterboard: Use of gypsum waste to produce gypsum powder used for production of new plasterboards.
- Cement: Use of gypsum waste to production of cement.

²⁷ http://www.eurogypsum.org/_Uploads/dbsAttachedFiles/EUROGYPSUMBIODIVERSITYEN.pdf

- Compost: Use of gypsum waste in compost as nutrient and structural material in agricultural soil.
- Potash tailings cover: Use of gypsum waste as covering and profile material for potash tailings heaps in Germany.

The authors concluded that the environmental impacts of the four treatment methods are relatively small and consequently collection and transportation as well as use of residues from the waste gypsum have a significant impact on the results. This means that site specific conditions and transportation distances may be crucial for the assessment.

Figure 7-2 Illustration of transport distances for extraction and use of natural gypsum as compared to FGD gypsum and recycled gypsum; The length of the blue arrows is indicating transport distances.

7.7 Pollution content and release from recovered gypsum-based materials

Ehrnsperger & Misch (2006) collected information on potentially releasable dangerous substances from construction products. For gypsum products the following dangerous substances are mentioned:

- Benzene (adhesives, components)
- Biocides (wood, cardboard containing fungicides)
- Biopersistent fibres (insulation materials)
- Cadmium and its compounds (plastics)
- CMR (carcinogenic, mutagenic and reprotoxic) substances Cat. I/II
- Formaldehyde (wood, components, adhesives)
- Radioactivity (gypsum from phosphoric acid manufacture, called phosphogypsum - PG)²⁸
- Heavy metals (gypsum from phosphoric acid manufacture, called phosphogypsum - PG)
- VOC (adhesives, components)

Gypsum products can be produced from natural gypsum, FGD gypsum, recycled gypsum or combinations of all three gypsum types. In the tables below examples of pollution content and release

²⁸ Three options for management of PG exist: (i) discharging to water bodies, (ii) disposal on land (i.e., stacks, stockpiles), and (iii) re-use for e.g. soil amendments, production of building materials, manufacturing of cement (Parreira et al., 2003; Kassir et al., 2012). Overall, 15% of world's PG has been estimated to be re-used while the remaining 85% is being discharged and/or disposed of (Perez-Lopez et al., 2007; Tayibi et al., 2009). Furthermore, the majority of re-used PG is used in soil amendment application and alike as the re-use of PG in building materials has been banned both in the US and EU because of unfavorable technical properties of the products and presence of hazardous elements (Reijnders, 2007).

7 Gypsum-based construction materials

are presented for FDG gypsum, C&D gypsum waste and recycled gypsum. There is no information available as to what type of gypsum the C&D waste and the recycled gypsum, respectively, are produced from.

Table 7-2 shows the pollution content and release from FGD gypsum. Content of metals and salts in FGD gypsum varies depending on the desulphurisation process used. The results below are given for a sample of FGD gypsum, which originates from a wet process where the gypsum product has been washed. This means that impurities/pollution has been transferred to the gypsum sludge that is produced by this kind of treatment.

Table 7-2 Total content and release from FGD gypsum (results are for one sample only) (DHI data).

Substances	Total content (mg/kg)	Leaching (Column leaching test, upflow, using artificial rainwater)		
		Leached concentrations (mg/l)		
		L/S 0.1	L/S 2.31	L/S 8.97
Chloride	-	690	0.36	0.66
Fluoride	-	8	5.2	0.76
Sulphate	-	2100	1200	1500
Ca	230 000 ^a	660	630	620
Mg	-	170	0.5	< 0.1
Na	-	355	0.8	0.5
K	-	34	0.1	0.1
As	4.2	0.001	< 0.0005	< 0.0005
Ba	-	0.074	0.035	0.030
Cd	< 0.05	0.0072	0.00009	< 0.00005
Cr	< 3	0.4	< 0.03	< 0.03
Cu	< 10	0.0086	0.0012	< 0.0005
Hg	< 0.2	< 0.0001	< 0.0001	< 0.0005
Mo	< 2	0.070	0.002	< 0.001
Ni	< 3	0.0032	< 0.001	< 0.001
Pb	< 2	0.0013	< 0.0005	< 0.0005
Se	0.11	0.22	< 0.003	< 0.003
V	< 2	0.055	0.007	0.002
Zn	< 3	0.1	< 0.01	< 0.01

^a calculated value

Table 7-3 shows an example of the pollution content in and release from gypsum waste and recycle gypsum, respectively. Note that this is data for one sample of gypsum waste and one sample of recycle gypsum. Samples are independent of each other and cannot be used to directly compare, *e.g.* leaching before and after processing of gypsum waste to recycle gypsum.

The parameters covered by the chemical analysis are based on legislation of relevance to recycling activities, *e.g.* limit values for recycling in construction and recycling of waste materials as soil improver. Consequently, the list of parameters is only indicative for substances to be found in gypsum waste and recycle gypsum. The release of sulphate (gypsum – calcium sulphate – has a solubility of approximately 2.5 g/l) is generally considered the major problem in relation to contamination of groundwater and fresh surface water bodies.

Table 7-3 Example of pollution content and release from C&D gypsum waste and recycle gypsum (Note! data for only one sample of gypsum waste and one sample of recycle gypsum). (DHI data)

Substances	Total content (mg/kg)		Leaching (Column leaching tests CEN/TS 14405) Accumulated leached amounts (mg/kg)					
	C&D gypsum waste	Recycled gypsum	C&D gypsum waste			Recycled gypsum		
			L/S 0.1	L/S 2	L/S 10	L/S 0.1	L/S 2	L/S 10
Chloride			32	92	120	45	200	220
Fluoride			1.0	14	55	0.98	17	61
Sulphate			220	3000	14000	240	3600	15000
Si			1.3	11	32	1.7	18	53
Ca			88	1400	6400	87	1500	6400
Mg			20	86	92	18	110	130
Na			18	58	69	28	140	160
K			43	190	260	26	150	200
Al			0.38	1.5	11	0.063	0.25	4.9
As	<2.0	<2.0	0.0024	0.018	<0.024	0.0026	0.021	0.037
Ba			0.021	0.18	0.76	0.010	0.14	0.63
Cd	<0.05	<0.05	0.000026	<0.00012	0.0059	0.000069	<0.00017	0.0038
Co			0.0069	0.028	0.035	0.0056	0.035	0.043
Cr	7.1	3.5	0.0039	0.0084	0.072	0.0037	0.017	0.062
Cu	4.7	3.9	0.0024	0.0056	0.062	0.0028	0.010	0.041
Fe			0.26	1.6	4.5	0.60	7.3	10
Hg	0.12	0.22	<0.0000055	<0.00010	0.00068	0.000066	0.00030	0.00088
Mn			0.48	2.9	5.6	0.37	3.3	6.4
Mo			0.0086	0.033	0.069	0.0067	0.042	0.080
Ni	4.8	2.3	0.016	0.061	0.13	0.017	0.098	0.15
Pb	<3.0	6.1	0.00022	<0.0012	0.027	0.00048	0.0017	0.026
Sb			0.0011	0.0056	0.026	0.0020	0.012	0.028
Se			0.050	0.31	0.56	0.0068	0.048	0.081
V			0.0042	0.015	0.085	0.012	0.083	0.14
Zn	10	36	0.032	0.12	0.20	0.077	0.31	0.35
DOC			390	1300	1800	68	400	540
TOC	9100	9000						
DEHP ^a	1.1	11						
LAS ^b	<50	<50						
NPE (Sum of nonylphenols)	<0.60	<0.60						
Sum of 9 PAH ^c	<0.020	0.1						
PCB, sum ^d	<0.0050	0.027						
Dioxins ^e	0.046 ng/kg dm/ 1.02 ng/kg dm	5.08 ng/kg dm/ 5.34 ng/kg dm						

^a phthalate plasticizer^b Linear alkylbenzene sulfonates^c Sum of acenaphthene, fluorine, phenanthrene, fluoranthene, pyrene, benzo(b+j+k)fluoranthene, benzo(a)pyrene, indeno(1,2,3-cd)pyrene, benzo(g,h,i)perylene^d Sum of PCB no. 28, PCB no. 52, PCB no. 101, PCB no. 118, PCB no. 138, PCB no. 153, PCB no. 180^e Dioxins as WHO(2005)-PCDD/F TEQ (lower bound) / WHO(2005)-PCDD/F TEQ (upper bound)

To set the total content measured into context, Table 7-4 shows

- Limit values for recycling of residual products and soil in building and construction work and on recycling of sorted, unpolluted C&D waste (Statutory order no 1662/2010 – does not currently apply to C&D waste or gypsum waste) – and

- Limit values for the use of waste for agricultural purposes (Statutory order no 1650/2006)

The analysed samples fulfil the limit values as shown below. The Danish limit value for PCB in C&D waste of 0.1 mg PCB/kg (sum of 7 PCBs and multiplied by a correction factor of 5) is exceeded for one of the samples.

Table 7-4 Danish limit values for recycling of residual products and soil and for the use of waste for agricultural purposes.

Substance	Recycling of residual products and soil in construction works ^a			Use of waste for agricultural purposes ^b Total content (mg/kg)
	Total content (mg/kg)	Leaching L/S 2 (mg/l) Category 1 and 2	Category 3	
Chloride		150	3000	
Sulphate		250	4000	
Na		100	1500	
Ba		0.3	4	
As	20	0.008	0.05	
Pb	40	0.01	0.1	120
Cd	0.5	0.002	0.04	0.8
Cr	500	0.01	0.5	100
Cr(VI)	20			
Cu	500	0.045	2	1000
Hg	1	0.0001	0.001	0.8
Mn		0.15	1	
Ni	30	0.01	0.07	30
Se		0.01	0.03	
Zn	500	0.1	1.5	4000
LAS				1300
Sum of 9 PAH				3
NPE (Sum of nonylphenols)				10
DEHP (di-2-ethylhexyl phthalate)				50

^a Statutory Order no. 1662/2010 on recycling of residual products and soil in building and construction work and on recycling of sorted, unpolluted C&D waste

^b Statutory Order no. 1650/2006 on use of waste for agricultural purposes (sludge ordinance)

Results from chemical analyses of gypsum waste to be used in compost are presented in Table 7-5.

Table 7-5 Physical/chemical parameters for sorted/cleaned gypsum waste for use in compost (Møller et al., 2012).

Parameter	Average	Standard deviation	Relative standard deviation %	Limit value ^a
Dry matter (%)	79.0	5.37	6.8	
Loss on ignition (%)	2.91	2.64	90	
Cd (mg/kgDM)	0.047	0.012	25	0.8
Hg (mg/kgDM)	0.116	0.063	54	0.8
Pb (mg/kgDM)	2.83	1.367	48	120
Ni (mg/kgDM)	7.55	2.833	38	30
Cr (mg/kgDM)	2.86	0.558	20	100
Zn (mg/kgDM)	21.8	9.543	44	4000
Cu (mg/kgDM)	2.88	0.997	35	1000
Total N (mg/kgDM)	94.9	47.5	50	
Total P (mg/kgDM)	152	24.9	50	
PAH (mg/kgDM)	< 0.2			
NPE (mg/kgDM)	< 0.6			
DEHP (mg/kgDM)	3.3		2.57	
LAS (mg/kgDM)	< 50			

^a Statutory Order no. 1650/2006 on use of waste for agricultural purposes (sludge ordinance).

7.8 Emission scenarios and calculated release/spread of pollutants

Data on pollution content and the release of pollutants from gypsum-based materials is limited. Therefore an overall description of the emission pathways are given for each selected recovery operation.

7.8.1 Recycling of recovered gypsum-based materials in new gypsum-based products

The overall emission pathways are illustrated in Figure 7-3.

Figure 7-3 Emission pathways from recycling of recovered gypsum-based materials in new gypsum-based materials.

Emission pathways during reclamation/demolition

During renovation, demolition and construction particulate emissions can be expected. Gypsum-based construction products are primarily used indoors, thus emissions are assumed to remain confined to indoor air at renovation, demolition and construction sites.

Emission pathways during transport

Particulate/dust emissions can be expected when materials are prepared for transport, to some degree during the transport of material and eventually when the waste material is unloaded.

Emission pathways during recycling/material recovery

Recycling of gypsum-based waste material involves the crushing and screening of the material and will generate particulate/dust emissions. Gypsum-based waste material is typically stored inside to prevent the material from getting wet. This means, that dust emissions can be contained within the storage facility.

Emission pathways during production of new products

During the production of gypsum based products, *e.g.* plasterboards, the gypsum powder is mixed with water to produce a slurry which sets by hydration. Excess water is removed in subsequent dryers.

During mixing of gypsum powder with water additives are blended in as well. Additives are used to *e.g.* ensure that gypsum and cardboard adhere to each other, that the slurry is light and to control the setting of the gypsum slurry²⁹.

²⁹ <http://www.gyproc.dk/om+gyproc/produktion>

Any substance present in the gypsum powder – anthropogenic or naturally occurring – will remain in the gypsum product, unless temperatures in drying processes are sufficient to degrade specific compounds. Provided that there are unwanted substances or pollutants present in the gypsum powder there is a risk of substances accumulating in the production chain as gypsum-based waste materials are recycled again and again.

7.8.2 Recycling of recovered gypsum-based materials as a raw material in cement production

Figure 7-4 illustrates the overall emission pathways from recycling of gypsum waste in cement.

Emission pathways during reclamation/demolition

During renovation, demolition and construction particulate emissions can be expected. Gypsum-based construction products are primarily used indoors, thus emissions are assumed to remain confined to indoor air at renovation, demolition and construction sites.

Emission pathways during transport

Particulate/dust emissions can be expected when materials are prepared for transport, to some degree during the transport of material and eventually when the waste material is unloaded.

Emission pathways during recycling/material recovery

Recycling of gypsum-based waste material involves the crushing and screening of the material and will generate particulate/dust emissions. Gypsum-based waste material is typically stored inside to prevent the material from getting wet. This means, that dust emissions can be contained within the storage facility.

Emission pathways during production of new products

Cement is produced from a mixture of limestone (calcium carbonate) and small quantities of clay or shale. The raw material mix is fed into a rotary kiln where the mix is heated to 1400-1500 °C. This process is known as calcination. The resulting material is called clinker. The clinker is then ground with a small amount of gypsum. The gypsum is added to control the setting properties of the cement when water is added. Consequently, any substance present in the gypsum powder – anthropogenic or naturally occurring – will remain in the cement product.

Figure 7-4 Emission pathways from recycling of recovered gypsum-based materials in cement.

7.8.3 Recycling of recovered gypsum-based materials in compost as nutrients and structural material

Figure 7-5 illustrates the overall emission pathways from recycling of gypsum waste in compost.

Emission pathways during reclamation/demolition

During renovation, demolition and construction particulate emissions can be expected. Gypsum-based construction products are primarily used indoors, thus emissions are assumed to remain confined to indoor air at renovation, demolition and construction sites.

Emission pathways during transport

Particulate/dust emissions can be expected when materials are prepared for transport, to some degree during the transport of material and eventually when the waste material is unloaded.

Some composting facilities process gypsum based waste material on site, which means that the waste material can be transported directly from a collection point to the facility.

Emission pathways during recycling/material recovery

Recycling of gypsum-based waste material involves the crushing and screening of the material and will generate particulate/dust emissions. Gypsum-based waste material is typically stored inside to prevent the material from getting wet. This means, that dust emissions can be contained within the storage facility.

Emission pathways during application as structural material/nutrient

During the final phase of the composting process gypsum is added to the compost. The compost product is spread on agricultural land and gypsum dissolves via contact with rain water. The degree to which gypsum dissolves will depend in the hydrological conditions (rainfall, infiltration, surface run-off etc.). This will affect the degree to which sulphate is released and can leach to soil and groundwater.

Sulphate will be taken up by the vegetation. However, any sulphate exceeding the plants need for nutrients may leach to soil and groundwater. Furthermore, outside the growing season there is a risk of sulphate leaching to soil and groundwater (Møller et al., 2012).

Any other substance present in gypsum (*e.g.* additives, metals), and not removed during processing of the gypsum waste, will end up in the compost product. Consequently, there is a risk that these substances are released when compost products containing gypsum are used on agricultural land. To what degree substances will be released and may leach to soil and groundwater will depend on the substance properties as well as their interaction with compost material and soil.

Figure 7-5 Emission pathways from recycling of recovered gypsum-based materials in compost.

7.8.4 Disposal/landfilling of gypsum waste

Figure 7-6 illustrates the overall emission pathways from landfilling of gypsum waste.

Emission pathways during renovation, demolition, construction

During renovation, demolition and construction particulate emissions can be expected. Gypsum-based construction products are primarily used indoors, thus emissions are assumed to remain confined to indoor air at renovation, demolition and construction sites.

Emission pathways during landfilling

Particulate emissions to air and surroundings may be expected during landfilling of gypsum waste. Emissions will be limited to the period where the material is placed at the landfill and daily landfill cover can be assumed to prevent spreading of particles.

The environmental impact from release of substances is reduced by the installation of protective measures such as bottom liners and leachate collection systems and by the requirements described in Section 7.5.

Figure 7-6 Emission pathways from landfilling of gypsum-based materials.

7.9 Comparison of scenarios

Due to limited data on release of pollutants from gypsum-based materials it is not possible to estimate the release and spread of pollutants. Therefore, only a general description of emission scenarios can be given.

Gypsum-based materials can contain a number of potentially dangerous substances; substances that originate from the materials use (e.g. paints) or from production (e.g. additives). Unless pollutants are effectively removed during the production of new products (e.g. production of new plasterboards or cement) or during alternative recovery operations (e.g. structural material/nutrient), there is a risk that pollutants are incorporated in the new product or released to the environment, respectively. If pollutants are not removed during production of new products, there may be a risk of substances accumulating in the production chain as gypsum-based waste materials are recycled again and again. However, to what extent this in fact may be a problem would require relevant data and could not be evaluated in this project.

The purpose of the Danish project carried out by Møller et al. (2012) was to analyse and quantify the potential environmental and economic costs and benefits of different treatment methods for gypsum

waste. The authors concluded that the recovery of gypsum waste to produce gypsum powder for the production of new plasterboards and the use of gypsum waste in the production of cement are favourable as compared to the use of gypsum waste in compost (or as slag heap cover), as the two first recovery processes resulted in net environmental benefits.

The authors concluded moreover that the environmental impacts of collection and transportation as well as use of residues from the waste gypsum have a significant impact on the results. This means that site specific conditions and transportation distances may be crucial for the assessment. The sensitivity analysis carried out by Møller et al. (2012) showed, that alternatives for substitution of natural gypsum (*e.g.* gypsum from Spain, Germany or from power plants) is very important.

8 Impact of the 70 % target on the handling of wood waste

This section differs from the assessment of other materials in this study, since it does not include any leaching estimation but instead includes life cycle assessment (LCA) calculations. This different approach is due to the fact that other environmental aspects are of concern regarding recycling of wooden waste.

8.1 National data

8.1.1 Data from Eurostat

Data on wood waste amounts have been obtained from Eurostat's database (Eurostat, 2013). Every second year, all European member states shall report to EU about waste generation and treatment according to the Waste Statistics Regulation (EU, 2002).

The waste generation is reported in 50 different waste categories (named EWC-Stat) and in 19 different sectors. Wood waste is one of the waste categories reported. Wood waste is divided into hazardous wood waste (mainly impregnated wood) and non-hazardous wood waste. Construction is defined as NACE F³⁰ according to the European economic nomenclature. NACE F is divided into Construction of buildings, Civil engineering, and Specialised construction activities (including demolition).

The waste treatment reporting is divided into the following five categories:

- incineration with heat recovery
- incineration without heat recovery
- recovery operations (excluding energy recovery)
- landfilling
- other disposal.

In the waste treatment reporting there is no connection between generation and treatment, for example the statistics do not show how the wood waste from construction is treated, only how the wood waste from all sectors in general is treated.

The data reported for the year 2010 is presented in Table 8-1.

Table 8-1 Generation of non-hazardous wood waste from construction, reported in 2010 (Eurostat³¹, 2013).

Country	Wood waste (tonnes)
Denmark	20 641
Finland	891 000
Norway	169 201
Sweden	125 000

There is no Eurostat data on the treatment of wood waste from construction, nor from any other sector. Treatment data is only available for all wood waste from all economic sectors. The data on the treatment of non-hazardous wood waste from all economic sectors for 2010 is presented in Table 8-2.

³⁰ NACE = Nomenclature générale des Activités économique dans les Communautés Européennes

³¹ <http://epp.eurostat.ec.europa.eu/portal/page/portal/waste/data/database> accessed 2013-09-30

Table 8-2 Treatment of non-hazardous wood waste from all economic sectors in 2010 (tonnes). (Eurostat³², 2013).

	Total waste treatment	Incineration / energy recovery (R1)	Recovery other than energy recovery	Incineration / disposal (D10)	Disposal
Denmark	129 196	10 631	116 762	0	1 803
Finland	10 445 832	7 630 000	2 795 000	15 140	5 692
Norway	426 514	4 057	323 080	61 155	38 222
Sweden	1 413 833	1 320 459	93 317	0	57

Some comments on these figures on generation and treatment are presented in the following.

8.1.2 Denmark

The Eurostat database shows a rather small amount of wood waste generated compared to the other Nordic countries and also compared to data from earlier years (Denmark reported 226 754 tonnes of wood waste from construction in 2008 and 8 185 tonnes from waste collection). For 2010 some 21 097 tonnes are reported as generation of wood waste from construction. For the same year, about 289 687 tonnes are reported as generation of wood waste from other waste collection.

Data from the Danish EPA's waste database (Danish EPA, 2013), which is based on data from the ISAG system (data from waste received at registered treatment plants) indicates that the amount of wood waste from the construction sector was 63 415 tonnes in 2009. Denmark is currently updating the waste registration and newer data is not obtainable for the moment. The following treatment of wood waste in 2010 is reported in Eurostat for Denmark:

- Total treatment of wood waste: ca. 149 567 tonnes
- Incineration/energy recovery 18 715 tonnes
- Recovery other than energy recovery 125 953 tonnes
- Disposal 4 899 tonnes

The treatment of the wood waste is not presented, but probably nearly all wood waste from construction is treated by energy recovery.

8.1.3 Finland

The figures from Finland are based on waste factors (based on construction statistics). The same figures are also presented at Statistics Finland's website (Statistics Finland, 2013). The amount seems to be high compared to the other Nordic countries.

There are no reports about the treatment of wood wastes from construction in particular, only on the treatment of wood waste from all sectors. The total treatment of wood wastes from all sectors amounts to 10.5 million tonnes, of which 21 % is recycled (wood residues from saw mills), 76 % is energy recovered, 1.4 % is incinerated without energy recovery and 2.4 % (about 250 000 tonnes) is landfilled. It is likely that some of the landfilled wood waste originates from constructions.

8.1.4 Norway

The figures from Norway are based on a statistical survey of the municipalities which collect annual data on waste from construction. In 2011 the amount had increased to 228 345 tonnes (Statistics Norway, 2013). According to Statistics Norway 226 917 tonnes of wood waste from construction is incinerated whereas the treatment of the remaining 1 428 tonnes is not specified.

The major part of the total wood waste from all sectors is either energy recovered or recycled, but small amounts are also either landfilled or incinerated without energy recovery.

³² <http://epp.eurostat.ec.europa.eu/portal/page/portal/waste/data/database>, accessed 2013-09-30

8.1.5 Sweden

The figures from Sweden are based on a combination of waste factors (based on construction statistics) and a survey of the bigger construction companies. In principle all wood waste is incinerated with energy recovery. No wood waste is landfilled, according to the data from Swedish EPA (SEPA, 2012b), and only minor amounts are material recycled (most likely in the form of bark waste for landfill cover).

8.1.6 Non-reported wood waste

It should be mentioned that the figures above show the amounts that are handled as wood waste. There may also be wood waste that is discarded into mixed waste, and thus is reported as mixed waste. There is no indication of how much of the wood waste is handled as mixed waste. In the Swedish survey, a waste factor approach estimated the total potential amount of wood waste to 266 000 tonnes ($\pm 50\%$), of which 125 000 tonnes were handled as a wood waste fraction, and the rest put into mixed waste.

The mixed waste can be of different character:

- Combustible waste: Mixed combustible waste that is brought to energy recovery.
- Waste to sorting: Mixed waste that is brought to a sorting plant where combustible waste (including wood) is sorted out for energy recovery and the residue for landfilling. The landfill residue in Sweden may contain up to 10 volume per cent of combustible material.
- Landfill residue. Mixed waste with more or less combustible material that is disposed at a landfill.

The amounts of *mixed waste* from construction in 2010 are presented in Table 8-3.

Table 8-3 Amount of non-hazardous mixed waste (10.2 Mixed and undifferentiated materials) from construction, reported in 2010 (Eurostat, 2013).

Country	Mixed waste (tonnes)
Denmark	46 983
Finland	5
Norway	300 934
Sweden	101 900

Thus, the potential amounts of wood waste can be expected to be larger than the reported amounts above.

8.1.7 Discussion

The Eurostat database does not give sufficient structure and level of detail to really follow up the 70 % target, but national data gives better information for some countries. In Sweden, Norway and Denmark the wood waste from the construction sector is mainly incinerated with energy recovery, but in Finland, the wood waste from construction may go to incineration without energy recovery or to landfill. The Eurostat data does not show if wood waste from construction is re-used or material recycled. Better information about the flows from the cradle to the grave is necessary to follow up the target.

In Sweden, there is a study in progress where the possibility to use data from all treatment plants is analysed. It should also be possible to trace wood waste in the new Danish waste registration system. Also the Norwegian system, where municipalities collect data on waste from the construction sector, gives opportunities to follow the flows of waste from the generation to the treatment.

8.2 Screening LCA

8.2.1 The origin of the waste matters

The environmental benefits of recycling will vary depending on the origin of the waste. In statistics the waste source is not always known, as mentioned above. In this context, when the goal is to evaluate environmental aspects, it is interesting to know if the waste originates from an old discarded product or if it is actually a production residue. These two alternatives may be defined as follows³³:

- **Pre-consumer** materials are generated by manufacturers and processors, and may consist of scrap, trimmings and other production residues that were never used on the consumer market.
- **Post-consumer** material is an end product that has completed its life cycle as a consumer item and would otherwise have been disposed of as a solid waste. Products made from “post-consumer” materials implies that the material originates from the society (from the techno sphere resource pool) and not directly from a natural resource.

With respect to the environmental burden in any system analytic tool, pre-consumer materials have to take some responsibility for the environmental impact from the process that it originates from – even though it might be very small depending on the principle of the allocation procedure used.

Inapproachable of the allocation procedure used, the well-established international standard for LCA (ISO 14044) sets that an inherent property may never be allocated away. Examples of such inherent properties are inherent energy content or the fact that the material is made of bio-based raw materials.

The term “post-consumer materials” means that the historical environmental burden has already happened and is allocated to the first or previous product system using the raw material as such. In a so-called attributional³⁴ LCA – the methodological approach regarded as the most robust way of performing an LCA – the historical impact is treated as a sunken cost³⁵ and the only “problem” left is to define the system boundary when the first product life cycle ends and the new one starts. A common way to do this is to follow “the lowest value of interest³⁶”. So, when the scrapped product is sold as a waste material its forthcoming environmental impact will be allocated to the new product system. This definition may vary between countries and in some cases also regionally. If a producer of a recycled product gets paid to “take care” of the wood waste, it is only the environmental impact from this point that will be allocated to the recycled product system. Note that even in this case, when an allocation is performed in accordance with the ISO 14044 main stepwise allocation procedure, the energy efficiency for e.g. district heating may be as low as 1 (but not less). This is a fact since the upstream energy use is allocated to the original product system, but the inherent energy content is not possible to allocate “away” and the environmental burden will follow the waste used as fuel.

The principle of “the lowest value of interest” is included in the new guideline for environmental construction products (EN 15804) which is related to CPR. However, it is overruled by the fact that energy recovery has to meet at least an energy efficiency of 60 % (as defined in the waste directive). Consequently, all energy recovery processes with efficiency better than 60 %, are defined as energy processes, and the waste used will then have an environmental upstream impact and resource consumption at least equal with the inherent mass. All district heating plants in northern Europe are by this definition energy processes and the delivered energy will have an environmental impact from the combustion etc. This means that the energy delivered from such plants will have an environmental burden and the waste is not responsible for the environmental impact from the combustion process. Therefore, this kind of analytic figures based on basic natural science assumption shall not be

³³ http://www.feeneyarchitectural.com/feeneyarch/LEED/pop_LEED.html

³⁴ Attributional LCA = a study where each product has a given environmental burden based on allocation rules. The system is characterized of the 100% rule meaning that if this LCA approach is used the environmental burden related to all product consumed will match what is actually emitted in the real world. This is also why this LCA approach also sometimes is called book keeping LCA and is a methodology that is frequently used for EPD (environmental product declaration). An alternative or supplement is as a consequential LCA study where the difference between two alternatives are analyzed and following the difference between these alternatives are reported. This LCA approach requires scenario setting that strongly affects the outcome of the study.

³⁵ Meaning that all historical impact will be regarded as zero and not allocated to the product under study.

³⁶ The delineation between two product systems is considered to be the point where the waste has its “lowest market value”. This means that the generator of the waste has to carry the full environmental impact until the point in the product’s life cycle where the waste is transported to a scrap yard or gate of a waste processing plant (collection site).

compared to those simplified approaches referred to as so-called primary energy factors that also include elements of on value choices (i.e. subjective).

8.2.2 Basic assumptions for setting the scenarios

Re-use

There are no official statistics in the Nordic countries on the re-use of wood products from the construction sector. It is assumed that the re-used amount is insignificant, even though we notice that products with antique values actually have a place on the market like old doors, iron ovens and windows and may be re-used³⁷. Moreover, the realism in an increased re-use of wood products is low but theoretically possible. In the waste context of interest here, re-use may then be regarded as a waste prevention strategy. However, since we assume that this will only cover a minimal part of the wood waste flow we do not consider this alternative further. In the long run also these re-used products will become wood waste and have to be treated properly from an environmental perspective. Re-use is therefore not an interesting alternative for this study.

Pre-consumer recycling

The basis for the present analysis is construction waste, thus, the wood waste that appear in the manufacturing of *e.g.* sawn timber is not regarded as construction waste (compare with the Finnish statistics above). It could be disputed if the production residue from sawmills should be regarded as waste at all, since most of the wood residues from sawmills are actually sold as raw materials.

The (only) wood waste that falls into this group is scrap wood from construction sites and different construction activities during the life cycle of construction works. This scrap wood has the potential to be sorted at the construction site and divided into different fractions suitable for further processing. Furthermore, it is dry and normally free from mould and rot.

To achieve an environmental gain, it will help if the fact that the material as such is already dried is used to save energy in the further processing. Currently, such dried wood scrap is *e.g.* used by the wood pellets industry. However, since that opportunity is part of energy recovery it falls outside the target recycling products that we seek for here. Instead, different wood composite products could be interesting. Scrap wood could be used for bio-based wood/plastic composites commercially available on the market today (*e.g.* deck and terrace material). However, these composites typically utilise pre-consumer plastic waste, thus the environmental gain is not as beneficial as if post-consumer waste had been used. The commercial development of non-renewable binders is therefore an alternative for wood composite products to achieve a more favourable environmental performance. The use of bio-based binders is potentially possible but not an economical realistic alternative today. These kinds of products will therefore not be evaluated further.

Other boards like plywood, OSB and LVL are not applicable since they require solid wood as resource. MDF and HDF boards include wet processing steps (such as the [Masonite-method](#)). More interesting is if a dry process was used instead and should then imply to any larger environmental gains. It was not found any running manufacturing equipment for production of insulation material based on a dry process. Therefore, due to lack of environmental or process data³⁸, the environmental performance could not be evaluated.

The only commercially available product made of recycled wood scrap identified here (besides the one mentioned above) is particle board. The fact that the scrap wood itself normally will have a lower moisture content than 20 % (water/dry mass) makes it a resource efficient alternative compared to the use of virgin wood that first has to be dried and the scrapped product is thus a perfect raw material for particle boards. Particle boards might also substitute other boards made of gypsum, silica, cement etc.

In theory, it should be possible to defibrate, i.e. to mechanical breakdown, the wood chips together with pressurized steam into wood fibrous components opposed to the method mention above. This kind of waste wood fibres produced from the defibrator method (also known as the Asplund method)

³⁷ [http://www.ircow.eu/media/downloads/D%205.1%20\(v2\)_%20DP_%20Report%20on%20CS1%20and%20CS2.pdf](http://www.ircow.eu/media/downloads/D%205.1%20(v2)_%20DP_%20Report%20on%20CS1%20and%20CS2.pdf)

³⁸ <http://www.siempekkamp.com/index.php?id=800>

8 Impact of the 70 % target on wood waste handling

should be possible to use as raw material for cellulose based insulation materials. This approach is a suggested innovation and does not exist and therefore has to be evaluated here from an environmental point of view rather than a technical (that might be interesting based on the environmental evaluation).

Post-consumer recycling

Post-consumer recycling means that the wood is aged and the inherent properties are changed. Perhaps the most problematic is if the wood is biological contaminated. The most promising products listed above are particle boards and insulation materials. In the latter case the product is often treated with fire retardants like boron (borax and boric acid). There are other fire retardants, but from a wood preservative perspective it should be known that these boron additives are efficient against mould and fungi as well. Boron is not used as wood preservative outdoors or in ground contact since it leaches too easy, but it is used as wood preservative indoors or in applications where the products are not exposed to weathering. For this case study, we will assume that it is possible to sort and store the discarded wood for the use as raw material in a particle board, without adding any chemical treatment to resist biological attacks. When used in insulation material it is common to add fungicide and fire retardants. In the case of recycling wood that is mechanically broken down in the size of saw dust or wood chips, 5 % by weight of calcium hydroxide ($\text{Ca}(\text{OH})_2$ slaked lime) has historically been added to resist biological attacks. In literature it is mentioned that in Germany that cement is used for the same purpose.

Cement bound products are interesting as the combined cement-wood mixture resists biological attack. The products available are boards and wood wool manufactured for plaster base, building elements (with wood reinforcement), sheet for removable false ceilings and for wall cladding. The products consist of wood wool or particles, cement and water. The products consist of 70 to 90 % by weight of cement and therefore have difficulties to compete with more wooden based products, thus these products are not accounted for here.

In conclusion, particle board is one of the most realistic alternatives for recycling of wood scrap and discarded wood material. One manufacturing technique is that the recycled wood is used in the the particle board centre while the surface is made of virgin wood. This type of board is illustrated in Figure 8-1 and will guarantee an attractive wooden colour of the surface. Different machinery that may use recycled wood as raw material was found, such as Fransson's Recycling machines (www.Franssons.com). Another machinery producer had experienced that it may cause problem when recycled wood is used, and that sorting is important. A test production of particle boards made of recycled wood was set up based on experience from Norway but had to close after half a year due to too much contamination that caused problem for the milling machinery³⁹.

Figure 8-1 Particle board made of waste wood.

³⁹ Personal communication 2013-09-02, Bo.E Sjöberg, BO-E. SJÖBERG i Stockholm AB

8.2.3 Selected products and functional unit

Based on the current knowledge, two products are selected as target for the case study to be manufactured with recycled wood as raw material. These products are particle boards and a wooden based insulation material. These two product groups are investigated and compared to a market dominant product that they might substitute.

The comparisons are made for 1 m² of board or insulation material with the same functional performance. In both cases this means that the thickness of the board materials may vary, and also the insulation slab thickness⁴⁰, in order to achieve a functional performance. By experience and regarding the usable intended use of these evaluated products, the products' service lives are assumed to be equal and no maintenance is needed for either product group. This product-to-product route will make it easy to compare individual product alternatives in a life cycle perspective, without any assumptions on service life predictions.

Moreover, the analyse in this case study is dived in two scopes as follows;

- product perspective
- societal perspective

The product perspective is found on an attributional life cycle perspective, meaning that a univocal result is achieved, since minimal methodology settings are needed according to the so-called core product category rule (PCR) EN 15804. Another benefit with attributional life cycle perspective is that the environmental impact reflects consequences in the real world, and thus the impact may be compared with national statistics like climate reporting etc.

The analysis then takes a societal scope into account. First, the individual saving is evaluated for a product which is substituted with a bio-based product. The attributional LCA approach is applied in this evaluation. The societal level is included by taking the annual savings into account. Then, the calculation requires knowledge on the market volume of the substituted product (and that the bio-based alternative with recycled wood can fulfil the functional requirements). Since the attributional system perspective is used, only *direct effects* are included, i.e. only the product perspective is accounted for. A second alternative evaluation is then set up where also *indirect effects* related to the foreground system is analysed. The foreground system defines the part of the expanded product system that in the society is actually affected by the changed manufacturing alternatives analysed. In this case, the foreground system includes the scrap wood that is currently used as energy carrier for district heating and then will be taken away. This energy system has to make use of a new alternative energy source – **a margin fuel** – that is the next energy source when the energy market is expanded and the old sources are insufficient.

The potential margin fuel that most likely will be explored in the future is forestry residues, also called primary forest fuel and including branches and tree tops (“GROT”), stem wood and stumps. Forestry residues are already used today, but the extraction can increase and will probably be used further in the future. Large investments in new district heating plants in Sweden have that in common that the plants are designed to use bio-based fuel like forestry residues⁴¹. This waste is a pre-consumer waste. The “Heat plan Denmark” also take into account a significantly increased amount of biomass. Moreover, in the current market situation, the additional biomass extracted from the forestry will be used as a fuel instead of, as in many parts of the forestry today, just left at the harvesting site. It is assumed here that the biomass from the forestry will be the main new marginal biomass based fuel. The forestry residual wood will not, according to this scenario and in the context of LCA, be a waste but a commodity with a market value. In theory, these kind of secondary effects do not necessary stop at the foreground system, but also affect other linked product systems that share a common (margin) market. This kind of secondary effects beyond the foreground system is not accounted for here and is a common limitation in this kind of so-called system expansions.

⁴⁰ An equal thermal comfort is taken into account based on the current λ value. The λ value describes the thermal conductivity that is equal with the rate at which heat is transmitted through a material, measured in Watts per square metre of surface area for a temperature gradient of one Kelvin per metre thickness, simplified to W/mK. The lower λ value, the better the thermal efficiency of the material.

⁴¹ Våra bränslen. Söderenergi AB, 2010.

As complement to the basis scenario given above, a sensitivity scenario is introduced where waste is assumed to be the margin fuel. Oil or any other fossil fuel is not regarded as a realistic alternative and is only used to describe ‘What if’, and may represent a historical perspective for district heating. In the short time perspective it is likely that waste actually will be the marginal fuel if there is no biomass. In the calculations, an environmental performance of 20 g CO₂/MJ⁴² is set for an average waste fraction used in the district heating. This figure then represents a fuel that consists of about ¼ of fossil resources. In the current “Heat plan Denmark”, waste can actually be regarded as a marginal waste. The same kind of scenario is found for Sweden. However, a number of political actions is likely to take force when the waste in the long run most likely will increase and in a Swedish scenario the waste amount will therefore likely peak around 2030. Danish policy goals⁴³ support this development, *e.g.* the (material) recycling of household waste should increase from the current 22 % to 50 % in 2022, and organic waste from the service sector should go from 17 % today to 60 % in 2018. In the context of wood and wood waste from the construction sector, the wood cycle in this kind of application will be from 30 to 50 years or even longer if parts of the building or other structural parts are included.

8.2.4 Consequences in the product perspective from changing to a recycled bio-based products

Two product groups are included, namely boards and insulation, as potential future products that may be produced from recycled wood. Note that other products may exist and the study do not claim that the selected alternatives handled here is the only alternatives or the best suitable alternatives. The alternatives included here shall be regarded as examples.

Insulation products environmental performance – attributional LCA

Mineral wool is the dominant insulation material on the Swedish market followed by cellular plastic (EPS, XPS) and cellulose fibre (made from recycled paper or virgin paper pulp). Mineral wool is divided in stone wool and glass wool. Stone wool is here used as reference product for the environmental calculations performed.

Wood from recycled construction products may be sorted into manufacturing wastage and demolition waste. Manufacturing wood wastage from the construction site may be handled so that contamination and weather exposure is avoided. If so, a pure wood fraction may be sorted out that can be used at the site to produce *e.g.* cutting shaving that may be recycled and installed in the very same construction works. Such cutting machinery exists and may be used for a local production of cutter shavings⁴⁴. The question is, however, what volumes that may be generated of this virgin wood pure waste. A more prefabricated construction sector should reduce this amount of manufacturing wastage. This cutting shaving and saw dust is by tradition mixed with calcium hydroxide or cement that also can be added on site. Saw dust is produced by cutting, and a product with relative high density is crushed wood from wood waste may be produced by a hammer crusher, or less sensitive mill technique. No measured data on the thermal quality for these fractions was found, but we have assumed a relative high λ value (assumed values for cutter shaving and sawdust/wood chips is set to 0.044 and 0.08 W/mK respectively), why these alternatives are not so resource efficient.

In old houses a porous wood board was commonly used for insulation (known as *e.g.* Tretex). Modern versions of this product exist and are not limited to thin boards (<300 mm) but also thicker rigid boards by gluing a number of board (typically by using PCAc)⁴⁵. The use of this kind of rigid board is assumed to be a minor market share compared to other insulation products like flexible bats. More interesting is an insulation production of a rigid or more preferable flexible wooden bat. Such products exist and may use a wet process⁴⁶ or a dry process⁴⁷. The identified machinery manufacturer for the

⁴² Current figures for household waste in Sweden are about 30 to 40 % based on: Bestämning av fossilt kol o avfall som förbränns i Sverige. Avfall Sverige, rapport U2012:02.

⁴³ http://www.mim.dk/NR/rdonlyres/F58E8C8B-3EB9-47DF-8DF3-4BF9207C9DFE/0/Ressourcestrategi_UK_web.pdf

⁴⁴ <http://www.sjoberg-jonkoping.com/stallstro.htm>

⁴⁵ http://www.kalcer.si/iz_pavatex/Prospekt_Pavatex.pdf

⁴⁶ http://swedish.hunton.no/assets/webbilder/brosjyrer/hunton_flex/Flex_produkter_low_feb_2010_Nyadress.pdf

⁴⁷ http://www.siempekkamp.com/fileadmin/media/Englisch/MaschinenundAnlagen/prospekte/Siempekkamp_wood-fiber_insulation_board-eng.pdf

dry process was contacted, but did not answer about *e.g.* energy need etc. for running the process or if it is installed anywhere. It could be assumed that the dry process is more energy efficient and more tolerant in the mechanical processing; crushing, sorting and a final milling of the perhaps contaminated wood. In the calculations below, data for a wet process is used in lack of data from the dry process. It should be noticed that a dry process requires a resin since the lignin cannot be utilised in the same way as in a wet process.

In order to make a fair comparison between different insulation alternatives a functional unit that takes an equal thermal quality into account is used. This implies that the different material will have the same thermal resistance ($R=0.2$). This equal resistance is calculated based on the material specific so-called lambda value, λ . In practice this means that that 200 mm stone wool shall be compared to 400 mm of saw dust. An average density for stone wool is set to 30 kg/m³ and 320 kg/m³ for saw dust, which means that it requires a huge amount of saw dust to reach the same thermal quality. Cutter shaving, on the contrary, has a low density (about 65 kg/m³) and better λ value. This means that saw dust will not give any environmental gains compared to if cutter shavings were used instead of stone wool (Table 8-4).

Table 8-4 Environmental performance reported as cradle-to-gate LCA data and an equal resource use per m² to achieve an equal functionality. This equal functionality is based on an equal thermal resistance, R , of 0.2 m²·K/W.

Product alternatives	kg/m ²	kg CO ₂ e/m ²	Relative insulation quality
Stone wool	6	7 159	1.00
Cutter shavings	14	709	1.13
Insulation board (Pavatex) ^a	28	11 508	1.03
Saw dust	128	6 211	2.05
Flexible Wood Fibre Insulation boards (Feelingwood) ^a	8	2 900	0.97

^a The environmental impact is based on the manufacturing process using virgin resources, why the impact is over estimated compared to if recycled wood was used instead.

Data for stone wool is based on an EPD from Rockwool in Norway. LCA data reported from other sources is in the same range.

The LCA data for a porous wood insulation board (Pavex) originates from one producer in Germany. It includes different additives and is based on a wet process. Moreover, the data is based on primary resources and therefore the environmental impact should be lower if based on recycled wood. This fact is also valid for the flexible wood fibre board (Feelingwood). The data from Flexible wood is also based on a combination of different sources based on a specific recipe valid for Feelingwood. The data for the wood wool is data from Karlit and their (former) production of porous wood boards (the plant is closed down). The environmental impact for a polyolefin (PE, PP etc.) that is used for binders and ammonium polyphosphate⁴⁸ is added and the latter works as a fire-retardant. This latter data is based on generic data from Ecoinvent (a LCA database). The LCA data for cutter shaving and saw dust includes the mechanical treatment of wood and the calcium hydroxide (note that if data for cement was used, the impact would be reduced a bit).

Based on the result in Table 8-4 it can be concluded that – if possible in practice – the shaving cutting as insulation material is the most environmental preferable. Obstacles concerning the economy for manufacturing of this product, small volumes and cheap material direct from the construction or demolition site make this product a more theoretical alternative than a practical one. Moreover, the product is a loose fill product why the intended use is limited. The low density flexible wood insulation bats are therefore more promising alternatives to compete with stone wool. The low environmental impact is generated by a lower density that also gives a better λ value. The porous wood insulation board will be a competitive alternative in applications such as wind breaking insulation (in the external part of the wall construction inside the facade material).

⁴⁸ Ammonium polyphosphate is also used as a food additive and emulsifier (E number: E545).

Cladding boards environmental performance – attributional LCA

Gypsum is the dominant wall cladding board on the market. The inherent property of gypsum gives it added value for sound insulation (heavy mass) and for fire protection. The weak aspect is that it risks suffering from mould attacks and potential fastening of different elements. Gypsum is easy to cut and easy to give a final plane finish over the board laps, which gives it added values in the construction process. As an educated guess, 2/3 of the interior walls are made out of single gypsum boards and 1/3 with double layer (or even triple in some occasions). Double layer increase the fire resistance, sound insulation and fastening possibilities. In the walls with double layers it is common to use OSB (oriented stain board), plywood or particle board to increase the potential fastening possibilities. Since these boards require more work to get plain over board laps etc. it is regarded as more cost effective to add an extra layer of gypsum board, instead of (most likely) just a layer of particle board. In this latter case we may assume that if only increased fastening is aimed at two layer of gypsum board will be equal with one layer of particle board (with an equal thickness).

In order to have an equal comparison we will assume that one standard gypsum board with a nominal thickness of 12.5 to 13 mm is equal to a 12 mm particle board. An average density of 650 kg/m³ is set for the particle board and 720 kg/m³ for the gypsum board, respectively. The functional unit will be an equal cladding board 12 mm gypsum board in standard application without requirement on heavily fastening. This functional unit may be regarded as in favour to gypsum board, but in remind that 2/3 are single boards applications, it seems justified for most applications.

The resulting environmental impact for gypsum board and particle board is listed in Table 8-5. No environmental data was found in the literature concerning manufacturing of recycled wood particle board. As basis for the calculation of a traditional particle board, LCA data from the end of the nineties is used and reported above as “STD particle board”. In those days, it was still common to use oil for thermal energy supply. This fact is changed nowadays. Moreover, the environmental profile for the STD particle board is made from wood from sawmills, and following EN 15804, it means that this post-consumer waste has an environmental backpack, which will not be the case as if construction waste was used instead. Therefore, these figures have been recalculated to reflect a manufacturing where wood is used for thermal energy and wood waste is used as raw material. This will definitely improve the environmental performance of the recycled wood particle board, which is reported in Table 8-5.

Table 8-5 Environmental performance reported as cradle-to-gate LCA data and an equal resource use per m² cladding board.

Product alternatives	kg/m²	g CO₂/m²
Gypsum board	9.0	1 990
Recycled wood particle board	7.8	1 201
STD particle board	7.8	2 129

The LCA data for gypsum board is from a producer that uses 99 % gypsum waste as raw material (as we understand the allocation approach in this LCA, all gypsum waste used is regarded as free of environmental loads). Therefore, the gypsum board is regarded as a top of the line version, but is used here since it is likely that other producer will follow this trend. The recycled particle board is the most profitable alternative concerning environmental impact contribution to climate change. As mentioned above, but worth repeating, different wood composite products will not imply an improved environmental performance that may compete with gypsum board, as long as pre-consumer plastic waste is used as raw material. Recycled particle board is a product that exist on some markets and has a relative low environmental performance why this is a strong candidate to potentially replace gypsum board for some extent. It should however be kept in mind that this requires a domestic manufacturing and not transport from abroad, which will increase the environmental impact of the recycled particle board.

8.2.5 Result on the societal level – attributional LCA

What is the potential national savings in Sweden if recycled wood from the construction sector is partly used to manufacture wooden based insulation bats and cladding boards and then replaces stone wool and gypsum boards?

The Swedish insulation market has a turnover of about 3.5 to 4 million m³ (Mm³) per year⁴⁹. 60 to 65 % of this market is mineral wool, 25 % is cellular plastic and the remaining part is dominated by cellulose fibre⁴⁹. These figures, together with the assumption that 2/3 of the mineral wool market share is substituted with wooden insulation bats made from recycled wood from the construction industry, will result in a yearly production of 61 kton. The remaining 1/3 comprises construction solutions where insulation boards cannot meet the required properties. From environmental performance and thermal quality figures reported in Table 8-4, the savings from shifting from mineral wool (represented here by figures on stone wool) will lead to a yearly saving of 52 kton CO₂e. It shall be noticed that the savings are much larger if cutter shavings were substituting mineral wool for some portion. However, technical issues concerning the possibility to actually use construction manufacturing waste as raw material, together with questions marks concerning the economy for this manufacturing process, have to be investigated further before this alternative could be seen as a realistic potential substitute. The potential savings related to this option are, however, very interesting to explore more in depth in the future.

The Swedish gypsum board market is dominated by Knauf Danogips and Gyproc that together have a market share of about 70 % (in 2012) of a yearly total turnover of about 38 million m² (Mm²). As mentioned above, 2/3 of the market is assumed to be single layer gypsum boards. In the remaining 1/3 it assumed that another 1/3 is double layer gypsum boards that might be substituted with particle boards. The remaining part is applications where gypsum board has properties that do not make particle boards as a potential alternative or other materials such as OSB and plywood is used. Altogether this means that 31 Mm² recycled particle board can be manufactured substituting gypsum board. Based on the figures in Table 8-5, a yearly potential saving of 25 CO₂e kton is possible to reach.

Figure 8-2 Relative impact related to global warming, when producing 1 m² board or insulation bats, comparing wooden product made of C&D waste and normalized to the potential substituting products. The difference, and therefore the potential saving when substituting, is equal to the difference between the blue and the brown bar.

In Figure 8-2 the most assumed realistic substitutes given in Tables 8-4 and 8-5 are summarized. The difference between the current product (blue bars) and the wooden product made of C&D waste (brown bars) is reported per square meter.

⁴⁹ <http://www.viivilla.se/Energi/Spartips/Isolera-ratt---och-spar-pengar-42820>

8.2.6 Marginal approach for product substitution – consequential LCA

Secondary effects are analysed in a consequential LCA. The case of future recycling of wood waste from the construction sector, which previously was used as fuel in district heating plants, will affect the district heating plants fuel mix. The evaluation performed here is divided in two parts; first, potential consequences of biotic carbon and temporary carbon sink effects are accounted for, second, the environmental margin consequence is calculated when different margin fuels are assumed. This effect is analysed with a so-called system expansion as illustrated in Figure 8-3. Note that the calculations are independent of the moisture content in different energy wares, since the combustion plant is supposed to be equipped with flue gas condensation. This is a common technique today, both in municipal solid waste fired plants and in bio-based fired plants.

Figure 8-3 The principle of “system expansion” takes indirect effects into account by adding the consequences from the new fuel and the replaced existing product resulting in a marginal effect. Note that the “remaining” or more correct “equal comparable” complex function after system expansion is an equal amount of fuel. The calculated resulting negative figure is called “avoided emissions”.

Biotic carbon and temporary carbon sink

This scenario describes the case when the wood waste from the construction sector is used in new construction products and the carbon fixed in these products will result in a delayed emission, compared to if the waste was used for district heating. On the other hand, additional harvested forestry wood waste, which before harvesting was biodegraded in a few decades, will now be emitted directly after harvesting in the district heating plant combustion process. An exponential approach is set up to model this biodegrading of biomass on the ground, where almost all biomass is supposed to be emitted within 50 years (Figure 8-4).

Figure 8-4 Accumulated emission scenario for a) construction wood waste that will be used as raw material for a new wood product and stored in construction works for 50 years, and b) forestry residues that now will be used as replacement fuel at the district heating plant.

In LCA, the time horizon boundary cut-off is normally set to 100 years and this is also the case for setting the integrated effect when calculating the impact assessment factors for climate change. If one accepts this boundary condition, the effect of a sink or delayed emission can be estimated based on the same underlying equation as given by IPCC⁵⁰. The approach used here follows the British standard for climate footprint calculation named PAS 2050. Equation 8a below is the same that is suggested by ILCD handbook (JRC, 2010) as method for the so-called Product Environmental Footprint (PEF), which is commissioned by DG Environment. This weighting factor, *WF*, is calculated as (PAS 2050:2011):

$$WF = (\sum p \cdot (100 - i)) / 100 \quad (8a)$$

where

WF is the weighting factor

i is each year in which emissions occur

p is the proportion of the total emissions occurring in any year *i*.

The wooden product in the scenario calculations is assumed to be stored in construction works for 50 years and then emitted. When the emission scenarios (Figure 8-4) are combined with equation 8a to model delayed emissions, the impact of 1 kg CO₂e from wood waste stored in a building for 50 years will be equal to a positive contribution to climate change of 0.5 kg CO₂e/kg. The, so to say, “lost storage” of forestry residues from harvesting and thinning is a negative consequence that will be allocated to the environmental burden to the new recycled product made of wood waste from the construction sector. This sink effect accounts for the “lost storage” (see Figure 8-3), where more than 80 % of the wood has transformed to carbon dioxide by natural processes and according to equation 8a has generated a sink effect of 0.16 kg CO₂e/kg. This will result in a net positive contribution of 0.34 kg CO₂e/kg recycled wood, in favour to the recycled C&D waste product. With other words; the sink effect from the storage wood in the construction works has greater positive effects than the negative aspects from taken forestry residues as a new fuel to the district heating plant. If the wood is stored for 100 years (or longer) the net benefit will increase to 0.84 kg CO₂e/kg recycled wood. However, as mentioned above, this kind of calculation is not general agreed upon and is not part of the mandatory impact categories in EN 15804. Nevertheless, if such sink effect is accounted for, it will gain the material recycling alternative.

It should be noticed that the way to calculate the effect from a carbon sink may be disputed and is not given as a default indicator in EN 15804. Therefore, the figures given here in this matter shall only be regarded as an indicator of the sink effect. The evaluation of the sink effect is based on the assumption that the emission credit of a sink is limited to 100 years, and will then be in line with the time horizon as for the GWP 100 years, used to assess the value of emitted greenhouse gases.

Consequence on system expansion and different marginal fuels

A future recycling of wood waste from the construction sector will have consequences on the fuel market. This wood waste was previously used as fuel in district heating plants and the changed use will therefore affect the district heating plants fuel mix. As argued above, in the base scenario, the margin fuel in an expanding fuel market relevant for district heating plants is assumed to be forestry residues. This additional potential fuel is not economical to harvest today, but we assume that it will be in the future. For a sensitivity analysis it is assumed that a generic waste fraction that generates 20 g CO₂/MJ is the margin fuel. The environmental impact from combustion of the same amount of recycled wood is assumed to be the same as if forestry residues were combusted. This assumption seems to be fair concerning contribution to climate change. Moreover, for the case study we assume that the same handling and transportation is carried out (equal to 150 km road transport from source to storage and then to the district heating plant). Furthermore, emission of biotic carbon dioxide is in this kind of calculations set to zero for all resources originating from a sustainable silviculture (the forestry management). Therefore, the potential difference in fuel quality will not be visual when comparing different bio-based fuels. It is now possible to calculate the impact from the recycling, see Figure 8-5.

⁵⁰ See: http://www.ipcc.ch/publications_and_data/ar4/wg1/en/ch2s2-10-2.html and the note concerning estimating the CO₂ response function that is based on the revised version of the Bern Carbon cycle model (Bern2.5CC; Joos et al. 2001)

8 Impact of the 70 % target on wood waste handling

Figure 8-5 Environmental savings from material recycling when producing 1 m² board (left) or insulation bats (right) made of C&D wood waste and substituting forestry waste (green bars) and generic waste (red bars). The contribution to global warming is normalized to the largest value for respective material scenario. Both the source data (transparent blue and brown bars) and the resulting potential savings (green and red bars) from the system expansion are included in the same figure.

The result when performing a consequential LCA is typically dependent on which marginal fuel that is selected. In this case, when the marginal fuel is a biomass product, the material recycling route will be gained (see green bars in Figure 8-5). In this case, the attributional and consequential LCA system perspective gives the same result. If a generic waste fraction is assumed as marginal fuel, the energy recovery will be more profitable than producing particle board (see left part of Figure 8-5 and red bar that generate a larger avoided emission indicating that energy recovery in this case is the favourable alternative). However, in the case of material recycling for a wooden insulation product, the material recycling route will be the most profitable alternative, inapproachable if the marginal fuel is forestry waste or a generic waste.

If fossil fuels like oil or coal was the margin fuel, this would support the energy recovery route in both cases. In the very short run, this assessment indicates that energy recovery is the overall preferable route, as long as a fossil fuel is the marginal fuel. In the case that the marginal fuel is a waste fraction that contains about ¼ of material with a fossil origin, the most preferable recycling rout depends on what substitute that is analysed. It should be noticed that current waste includes up to 40 % fossil materials, which if used would change the result in Figure 8-5 so that material recycling and the wood bats in this case is not the preferable route. Based on the two wooden based products investigated here, a tipping point between energy recovery and material recycling seems to appear when a waste fuel with a fossil carbon content around 20 % is the substitute that replace the C&D wood waste. However, in the long run when the district heating probably will be almost fossil free and biomass is the marginal fuel, the material recycling route has the potential to be the better alternative. When dealing with the future, it is also technically possible to increase the use of bio-based fuels in the manufacturing of stone wool and therefore replace the coal used. Such development will generate an improved profile for stone wool, but is not accounted for here.

This study included environmental consequences in respect to global warming. Other aspects are not considered and should require a more extensive work. In respect to toxicity it is for instance interesting to evaluate if it is accepted that non-hazardous wood waste is suited for material recovery, even if it can contain paint with zinc, lead etc.

9 Discussion and conclusions

The main point for this project was “Which are the environmental consequences of the European target for recovery of C&D waste?”. Here in chapter 9, this question is discussed for each of the waste types studied (Section 9.1) and suggestions are given to improve the recovery target (Section 9.2). Furthermore, uncertainties of the current waste and recycling statistics and lack of data for material properties raised difficulties for the project work and therefore, in Section 9.3, suggestions are given to improve the current waste statistics. Finally, Section 9.4 includes ideas for generally promoting the use of recycled products in the future.

9.1 Which are the environmental consequences of the various scenarios studied?

General

The study has had a scenario-based approach, which was chosen because of the uncertainties of the current waste and recycling statistics. Various recovery scenarios have been compared in terms of resource management and spreading of pollutants. However, lack of data for material properties raised difficulties for the comparisons. This was due to waste reporting in mixed fractions and the fact that few or no leaching tests had been performed. For example, on the one hand recovery of asphalt and track ballast waste saves natural resources and potentially reduces the impact on the environment from transport of the natural resources. On the other hand, limited data on pollution content and release exists to evaluate the potential release and spread of pollutants from recovered material. It is concluded that in order to report progress regarding the target, an improvement of the statistics on the generation and recycling of waste as well as pollution content and release is needed.

Regarding consequences for transports, it is reminded that on site re-use or recycling reduces transport work. Furthermore, the impact will vary among countries or even regions since it depends on what supplies there are and which resources are replaced as well as on the distance to recycling facilities. In the Nordic countries, Denmark is different to Sweden, Norway and Finland regarding the resources of high quality crushed rock, which is largely imported to Denmark.

The consequences for spreading of pollutants will also vary since emissions (leaching, particles etc.) depend on how and where the waste is re-used or recycled. It is emphasised that the results of this study do not represent a risk assessment, but merely describe the potential release/emissions. To get reliable information on the environmental consequences there is a need for risk based assessment. Simplification may lead to wrong conclusions.

Although the recycling statistics are uncertain, it can be concluded that the present recovery rate of asphalt and track ballast waste is well above 70 % in the Nordic countries. These materials represent large material flows, but at the moment they are generally not included in the Nordic waste statistics. Considering their high re-use or recycling rate, including or excluding them will highly influence the opportunities of attaining the WFD target of 70 % recovery.

Asphalt

Reclaimed Asphalt (RA) represents one of the biggest flows of materials related to the WFD target in the Nordic countries, with about 3.4 Mtonnes per year. The recovery of RA is a well-established operation with a variety of recovery routes and techniques but it is generally not included in the waste statistics today.

LCA studies have proven that the re-use into new asphalt mixtures and bound courses is favorable compared to recycling into unbound layers and backfilling due to the re-use of the bitumen binders in the RA. In Denmark, where there is a short supply of high quality aggregates, recycling of high quality aggregates in bound layers is also important. The benefits of recycling are sensitive to transport distance. In situ and on site recycling techniques are available and widely used which results in

significant savings of transport work. Many applications still require temporary storage and pretreatment of the RA at recycling sites prior to recycling into new pavements. Keeping the distance between demolition, processing and recycling sites short is important to preserve the benefits of recycling. Asphalt paved roads are widely dispersed, which means that generation and recycling occurs widespread. To keep down transport, it is important that recycling sites are available in high numbers at short distance. Administrative barriers for processing sites could counteract high grade recycling and a simplified regulatory framework (permit system) for RA recycling is held to be beneficial from an environmental point of view.

The historic use of tar in asphalt pavements is an important source of PAH contamination in the existing stock of asphalt pavements. The extent of contamination varies between the Nordic countries, where it is high in Sweden and probably considerably lower in other Nordic countries. Data is limited, and no updated statistics are available regarding the extent of contamination in RA.

In several Nordic countries, guideline values (in Sweden and Norway) or limit values (in the Finnish region of Åland) are given for RA that can be recovered without restrictions (free use). Those values, 70-100 mg PAH-16 per kg asphalt, are more than 100 times higher than the content in asphalt based on virgin raw material, where a PAH content of <10 mg per kg bitumen (NVF, 2000) corresponds to a PAH content of <0.6 mg per kg asphalt.

PAH emission from contaminated RA is dependent on the chosen route of re-use or recycling. In Table 9-1, the calculated emission from the current stock and various potential recycling routes in Sweden are compared with the emission from point and diffuse sources.

Table 9-1 Estimated yearly emission and deposition of PAH in Sweden (kg). Data for tar contaminated asphalt from Tables 3-20 and 3-21. Data on emission from point sources and diffuse sources from SMED (2012).

	Source	PAH-4	PAH-12	PAH-16
Water	Point source Industry	10-50		
	Municipal sewage treatment	70		
	Urban stormwater			500
Air	Point source Industry	200		
	Diffuse source	13 000		
Deposition	On land		10 000	
	On lakes		1 000	
Emission from tar contaminated asphalt	Stock in bound base layer including recycling into bound base	14-20 ^b	125-180 ^b	140-200
	Recycling into surface layers ^a	6-16 ^b	55-145 ^b	60-160

^a Potential yearly emission when recycling into road surface layers assumes that all contaminated RA generated is recycled into surface layers. Emissions will accumulate due to the fact that the emission from road surfaces is much higher than the emission from bound base layers.

^b To get comparable numbers, PAH-16 data has been recalculated to PAH-4 and PAH-12 data based on tar asphalt composition in Table 3-10.

It can be concluded from Table 9-1 that the emission from the existing stock of tar contaminated asphalt in bound base layers is smaller than the emissions from other sources to water, but not insignificant. If tar contaminated RA with a PAH content corresponding to the limits for “free use” is recycled into surface layers, the emission related to contaminated pavements would increase by 50-100 % every year. Contaminated pavements would then become a leading source of PAH emission to water within a period of five years, but still be small compared to diffuse emissions to air and deposition.

The results of this study show that emission from the current stock of contaminated asphalt pavements is limited, and recycling of contaminated RA back into bound layers does not increase the emission. Current benchmarks for the free use of contaminated asphalt are high compared to uncontaminated

asphalt. Thus, they may need to be lowered to prevent increasing emission to water as a result of tar contaminated RA entering recycling routes where the emission factors are higher.

It should be emphasised though that the results of this study do not represent a risk assessment, but merely describe the potential release/emission and are associated with large uncertainties. To improve the assessment and give recommendations on adjusted and lowered limit values for free use of tar contaminated asphalt, the following improvements on data sources and emission models are recommended:

- Improved data on the generation routes and recycling routes for RA.
- Improved data on the PAH content of RA from different generation routes.
- Improved data, models and estimations on the PAH emission through fumes.
- Improved data, models and estimations on the PAH emission through surface run-off.

Concrete

Four recovery scenarios were evaluated including the use as: sub-base material in road construction, aggregates in new concrete production, final cover material at landfills and backfilling material at the demolition site. Out of these four recovery scenarios, it is likely that the total transport decreases if recycled concrete aggregates are increasingly used as sub-base material, aggregates in new concrete production and as backfilling material at the demolition site.

Primary aggregates are saved when recycled aggregates are used in all scenarios evaluated. In the road sub-base scenario, up to 1800 kg of primary aggregates can be saved per square meter of road. Regarding the use in new concrete production, the following estimation was made for coarse aggregates: If 30 % of the coarse aggregate fraction for the Norwegian concrete class B30 would consist of RCA, around 370 000 tonnes of primary aggregates would be saved annually.

When considering storage of concrete waste, processing and production of RCA and a user phase of 100 years, the chemical release to soil and groundwater was evaluated to be the most important factor to assess for processed concrete waste in the said recovery scenarios. Furthermore, a comparison showed a lower release of Cr, Zn and SO_4^{2-} in the road sub-base scenario than in the final cover at landfill scenario (used as drainage layer). It is also reasonable to conclude that the leaching from the road sub-base scenario will be lower than the leaching from backfilling at the demolition site. When RCA is used in new concrete (bound use), the chemical release is expected to be comparable to that of regular concrete provided that the engineering properties are roughly the same.

The purpose of the leaching assessments conducted in this study was to compare different scenarios. Although one scenario is in favour of another, the maximum tolerable release was not assessed. It can be concluded that limit values at the potential pollution source are needed, at least for scenarios that are environmentally favourable, in order to decide if the release is within the acceptable range.

Bricks, tiles and ceramics and mixtures of these and concrete

This waste flow is small compared to concrete waste. There are also problems with the definition of waste streams, e.g. bricks which are not re-used are often mixed with concrete wastes and it is likely that the waste is reported just as concrete waste in the statistics. It is difficult to find reliable information from the literature about the amounts and the quality of the specific waste streams containing bricks.

The data for emissions to water/soil presented here is only an example. Not much has been published about leaching properties, probably because this waste stream is not tested as such, especially if disposed in low grade applications or if the waste streams are named as concrete waste even if it contains bricks.

Track ballast

Recovery operations that ensure re-use of track ballast directly on-site or nearby can be said to be favourable, as transportation of virgin material and thus emissions as a result of transportation can be minimised and avoided as well as quarrying of virgin materials.

Furthermore, recovery operations that

- a) ensure efficient cleaning and removal of any pollutant prior to re-use of material or
 - b) avoid or reduce contact to water and hence emission of pollutants to soil and groundwater/surface water
- should be preferred if the release and spread of pollutants is to be avoided or minimized.

If quality criteria for the re-use of track ballast exist (as for example in Norway) and they are derived from risk assessment, they presumably prevent unacceptable contamination of soil, ground water and surface water as a result of re-use activities. If this is not the case, there may be a risk of spreading pollution if track ballast is re-used.

Gypsum

The purpose of the Danish project carried out by Møller et al. (2012) was to analyse and quantify the potential environmental and economic costs and benefits of different treatment methods for gypsum waste. The authors concluded that the recovery of gypsum waste to produce gypsum powder for the production of new plasterboards and the use of gypsum waste in the production of cement are favourable as compared to the use of gypsum waste in compost (or as slag heap cover), as the two first recovery processes resulted in net environmental benefits. However, since site specific conditions and transportation distances are very important for the assessment, the efficiency on a local level may vary and it may be necessary to carry out similar assessments for the individual Nordic countries.

Wood

Significant environmental improvements are potentially possible if construction and demolition wood waste (C&D wood waste) is material recycled. An initial screening of such potential recycling was performed in this study and this novel assessment indicated that the environmental gain is possible taking existing technically feasible products like particle board and different wooden insulation products into account. However, the work did not cover economic aspects, *e.g.* if this is commercially possible under current market situations. An educated guess is that the manufacturing of the recycled wood products is only possible if the wood waste is free of charge for the building material industry and if the wood is sorted in a quality suited for the recycling purpose. Furthermore, the calculations made presume that the transport distance will be about the same as today, which means that the production of the recycled wood product has to take place domestically or in a neighbouring country.

The assessment performed used two systems that are applied in LCA: a) the product approach, also known as attributional LCA, and b) consequential LCA covering a complex system. Attributional LCA is very robust and only includes direct consequences, while consequential LCA also includes indirect effects. More information about different system perspectives when performing an attributional or a consequential LCA may be found in Erlandsson et al. (2013).

An evaluation that also accounts for indirect environmental effects when the C&D wood waste is removed from the current energy market was also performed. This evaluation assumed that the marginal fuel affected by the recycling of C&D wood waste will be forestry residual wood. The consequences of this, so-called system expansion, includes the carbon storage and the sink effect and their effects on climate change. Such carbon storage is accounted for in the national climate reporting, but there is not full consensus on how such evaluations should be transformed and handled from a product perspective. The calculations made, follow the principle given by IPCC and are streamlined by PAS 2050 (that is also used in the Product Environmental Footprint, PEF, as suggested by DG Environment). This indirect effect results in an additional gain, illustrating that material recycling of wood as part of a construction work gives larger environmental gains, compared to the removal of forestry recedes (GROT), which is assumed to be the marginal fuel.

In conclusion, according to an attributional LCA, which has a product perspective, the use of C&D wood waste for manufacturing of particle board and insulation bats and then substituting gypsum board and mineral wool, will result in environmental improvements. According to a consequential LCA, a marginal fuel has to be defined. This marginal fuel is the fuel that will replace the current C&D wood waste. The most environmental profitable alternative will depend on what fuel is assumed to be the marginal fuel. The selection of a likely marginal fuel will be different if using a short time perspective compared to a longer time perspective (a few decades). The two extreme alternatives are that the marginal fuel is either any fossil fuel or a bio-based fuel. The first alternative will support that wood should be used for energy recovery and the second alternative that there is a potential environmental gain in wood material recycling.

The market acceptance and other technical as well as non-technical barriers related to the substituting alternatives given here is not part of the study and therefore not accounted for. These issues should be investigated further in future studies.

9.2 Suggestions for the next generation of the 70 % target

The recovery target is an important step towards improved recycling. Based on the project result the following suggestions for amendments of the target are given:

- The wastes covered and the calculation method for reporting progress: **Recovery targets should be set individually for different waste types reflecting their potential benefits when they are recycled.**
The studied waste types represent large material flows with good prospects for re-use or recycling in new constructions or along other recovery routes. However, we find that lumping all waste generation and waste recycling flows into one single target does not ensure a sustainable waste recovery. The current target is very sensitive to interpretations of what is considered as waste recycling and uncertainties in the current statistics. Furthermore, the current target is weight based and therefore mineral wastes will have the largest impact. One may argue that % by weight is the worst way to put the 70 % goal, as it gives focus to mineral materials while the environmental gain is perhaps on other materials. This does not mean that we do not support a sustainable use of natural mineral resources, but it has also to be efficient in a life cycle perspective and economically motivated.
- The recovery routes covered: **The recovery targets should distinguish between different recovery routes.** Our study shows that the choice of recovery operation could have big impact on the environmental gain but also on the potential release of contaminants.
 - Re-use or recycling into new constructions is often favourable due to environmental benefits (*e.g.* minimised transports or savings of virgin resources) even if the recovered materials are contaminated compared to virgin raw materials. The constructions often provide means of protection against release and exposure to contaminants. Future management of waste during the demolition or maintenance of constructions could also ensure that future recovery can be sustained. Guidelines on management and environmental criteria for recycling into new construction are important in order to promote favourable recycling routes but also to prevent unacceptable release of contaminants as a result of unfavourable recycling routes. These situations may vary from country to country depending on the supply of raw material and the frequency of contamination.
 - Backfilling is currently regarded as equal to other recovery operations when assessing the 70% target, but there is a risk for “downcycling”, which means that the waste is not used in the most optimal way. Besides, renewed future recovery is more unlikely for backfilled materials. Environmental risks will depend on the land use scenario of the backfilled area which may change without prior assessment. Backfilling should therefore be regulated by more stringent environmental quality criteria than other activities. The use of quality criteria derived from risk assessment, may prevent unacceptable contamination of soil, groundwater and surface water as a result of backfilling activities.

We recommend that the general target of 70 % should be transformed into material specific recovery targets, since this would favour efficient recycling of C&D waste in general and not only for high weight materials. These targets could be set both on a European and a national level. In addition, we suggest that the Nordic countries put up national material and scenario specific recovery targets so that the most effective re-use and recycling routes are promoted and those posing the biggest environmental risk are avoided. This could be done in combination with management guidelines and environmental quality criteria. Recycling routes posing high risk for environmental contamination would presumably be regulated by more stringent quality criteria compared to those with minimal or no risk for environmental contamination.

9.3 How can the current data be improved?

The current statistics do not allow a quantification of the environmental effects or the effects on recovery rate.

Regarding wood waste, the Eurostat waste data does not provide a structure and level of sufficient detail to follow up the 70 % target, but national data sources contain better information for some of the countries. The wood waste (that is reported as wood waste) from the construction sector is mainly incinerated with energy recovery in Sweden, Norway and Denmark, but in Finland, the wood waste from construction may go to incineration without energy recovery or to landfill. The present Eurostat data does not show if wood waste from construction is re-used or material recycled. Better information about the flows from the cradle to the grave is necessary to follow up the target. It should also be investigated if it is a problem that statistical data does not separate between pure wood and “treated” (painted) wood.

Lack of reliable information related to re-use or recycling of asphalt, concrete, bricks, track ballast and gypsum are similar as for wood waste. Many of these represent premium materials with high grade re-use or recycling options. Still, data on these, if available, are often reported merged with low grade mixed mineral wastes.

As relevant initiatives and projects are carried out, more information and data on C&D waste may become available. In Sweden, there is a study in progress where the possibility to use data on material flows from all waste treatment plants is analysed. It should also be possible to trace wood waste in the new Danish waste registration system. Also the Norwegian system, where municipalities collect data on waste from the construction sector, gives opportunities to follow the flows of waste from the generation to the treatment. Another example is “Building and Construction Waste” innovation platform under the strategic initiatives of the Copenhagen Clean Tech Cluster⁵¹, which was established in 2013 in order to examine the potential of new methods and technologies within waste management from the construction and demolition industry.

Large flows of asphalt, track ballast and concrete are not reported as waste handling, and the practice of in-situ re-use techniques, for example asphalt and track ballast, further complicates the collection of statistical data. These material flows are significant and will have an impact on the target fulfilment. If the flows are to be included in the follow up on the 70 % target, data has to be collected from industry organizations, clients or contractors. For asphalt and track ballast, market is dominated by major governmental clients and a limited number of contractors, why data may be possible to collect. In some cases there is already a certain collection of statistical data by industry organizations. For asphalt, several RA generation and recycling routes are of importance in relation to assessment of environmental benefits, contamination content as well as emission of contaminants. Detailed information on these routes is required for a comprehensive assessment.

The setting of environmental quality criteria for re-use and recycling of C&D waste can contribute to safe recovery of C&D waste. On the one hand this requires sufficient data, to be able to derive quality criteria, but it will on the other hand contribute to the generation of relevant data once requirements for

⁵¹ <http://www.cphcleantech.com/home/strategic-initiatives/innovation-platforms/building-and-construction-waste>

testing and analysis are implemented. If data is scarce in relation to material flows, it is even more so in relation to content and emissions of contaminants. Available data is also uncertain due to the use of diverse and poorly documented (non-standardized) sampling, sample treatment, emission measurements and content analysis methods. Asphalt, track ballast and mixed concrete/brick waste represent large flows with significant amounts of contaminants. A standardized data collection regarding the content and leaching of contaminants from these materials would enable better assessment of potential environmental risks than currently possible. We propose that a joint Nordic project is set up to specify a standardized procedure and to collect such data.

Quality requirements for recycled CDW when used as construction material are covered by harmonized European construction product standards. Discussion is needed on how to address recycling aspects in CE-marking, both regarding savings of natural materials and release of specific substances to soil and water. For waste related construction products the information of release should be reported as actual measured values (and not as classes indicating values below a threshold), thus providing another source of relevant data. Numerical data would enable further use as input data in environmental product declarations (EPD) and risk assessment.

9.4 How to promote the use of recycled products in the future?

Most (if not all) of the mineral C&D waste can replace natural aggregates. According to UEPG a better acceptability of recycled materials enhances their economic value and thus their profit margin to natural materials (UEPG, 2006). However, the recycled materials need help to compete with natural aggregates. Some ideas are presented here that would create drivers for the application of recycled products:

- Planning/designing: Put efforts in the selection of virgin raw materials. Recycling pathways should be assessed at the design stage, dangerous substances substituted and safe management routes employed. This will enhance the future recycling possibilities of the designed construction.
- Certifications and specifications: Establish quality certification and specifications for recycled products encouraging safe management routes. Recycled materials may have difficulties to provide the same quality as, and meet the criteria for, virgin raw materials and new products due to contamination. In some cases there is a misconception of the quality of recycled products compared to new materials, which will make it difficult to market the recycled products. They might still be advantageous from an environmental point of view in certain re-use and recycling scenarios. That is the case *e.g.* for RA and track ballast. Certifications and specifications would make them competitive in quality.
- Administrative and legal requirements: Facilitate recovery of waste on routes that are environmentally beneficial by providing simplified legal and administrative requirements with a predictable outcome for these routes.
- Construction: Use recycled aggregates on a local market close to the production site. This would reduce transport/logistics costs (and emission from transport), which in turn helps recycled aggregates to compete in price with natural aggregates, since production costs of recycled aggregates tend to be higher.
- Demolition: Set up requirements on deconstruction (instead of demolition), sorting at source (*e.g.* taxation of mixed waste stream) and reporting of different waste streams in the demolition process. Agree on clear definitions of the waste streams (bricks and concrete waste), maybe through the waste reporting. Require that pollutants are monitored, reported and if possible removed during demolition. All this would ensure the quality of recycled materials and reduce their market price.
- Establish landfill bans or increase landfill taxes. This would create an incentive to re-use/recover and recycle more. The availability of raw/virgin materials at low costs together with low landfilling taxes for C&D waste in many case discourage recycling.
- Formalize and increase the contacts/communications with stakeholders involved in the value chain. The focus must be on the total benefit including resource saving, less generation of waste, transportation costs, green points in environmental rating systems, according to national policy.

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(European harmonized standards are compiled in Annex C)

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Annexes

A Extracts from WFD

Article 4

Waste hierarchy

1. The following waste hierarchy shall apply as a priority order in waste prevention and management legislation and policy:

- (a) prevention;
- (b) preparing for re-use;
- (c) recycling;
- (d) other recovery, e.g. energy recovery; and
- (e) disposal.

2. When applying the waste hierarchy referred to in paragraph 1, Member States shall take measures to encourage the options that deliver the best overall environmental outcome. This may require specific waste streams departing from the hierarchy where this is justified by life-cycle thinking on the overall impacts of the generation and management of such waste.

Member States shall ensure that the development of waste legislation and policy is a fully transparent process, observing existing national rules about the consultation and involvement of citizens and stakeholders.

Member States shall take into account the general environmental protection principles of precaution and sustainability, technical feasibility and economic viability, protection of resources as well as the overall environmental, human health, economic

Article 5

By-products

1. A substance or object, resulting from a production process, the primary aim of which is not the production of that item, may be regarded as not being waste referred to in point (1) of Article 3 but as being a by-product only if the following conditions are met:

- (a) further use of the substance or object is certain;
- (b) the substance or object can be used directly without any further processing other than normal industrial practice;
- (c) the substance or object is produced as an integral part of a production process; and (d) further use is lawful, i.e. the substance or object fulfils all relevant product, environmental and health protection requirements for the specific use and will not lead to overall adverse environmental or human health impacts.

2. On the basis of the conditions laid down in paragraph 1, measures may be adopted to determine the criteria to be met for specific substances or objects to be regarded as a by-product and not as waste referred to in point (1) of Article 3. Those measures, designed to amend non-essential elements of this Directive by supplementing it, shall be adopted in accordance with the regulatory procedure with scrutiny referred to in Article 39(2).

Article 6

End-of-waste status

1. Certain specified waste shall cease to be waste within the meaning of point (1) of Article 3 when it has undergone a recovery, including recycling, operation and complies with specific criteria to be developed in accordance with the following conditions:

- (a) the substance or object is commonly used for specific purposes;
- (b) a market or demand exists for such a substance or object;
- (c) the substance or object fulfils the technical requirements for the specific purposes and meets the existing legislation and standards applicable to products; and (d) the use of the substance or object will not lead to overall adverse environmental or human health impacts.

The criteria shall include limit values for pollutants where necessary and shall take into account any possible adverse environmental effects of the substance or object.

2. The measures designed to amend non-essential elements of this Directive by supplementing it relating to the adoption of the criteria set out in paragraph 1 and specifying the type of waste to which such criteria shall apply shall be adopted in accordance with the regulatory procedure with scrutiny referred to in Article 39(2). End-of-waste specific criteria should be considered, among others, at least for aggregates, paper, glass, metal, tyres and textiles.

3. Waste which ceases to be waste in accordance with paragraphs 1 and 2, shall also cease to be waste for the purpose of the recovery and recycling targets set out in Directives 94/62/EC, 2000/53/EC, 2002/96/EC and 2006/66/EC and other relevant Community legislation when the recycling or recovery requirements of that legislation are satisfied.

4. Where criteria have not been set at Community level under the procedure set out in paragraphs 1 and 2, Member States may decide case by case whether certain waste has ceased to be waste taking into account the applicable case law. They shall notify the Commission of such decisions in accordance with Directive 98/34/EC of the European Parliament and of the Council of 22 June 1998 laying down a procedure for the provision of information in the field of technical standards and regulations and of rules on Information Society services (1) where so required by that Directive.

Article 11

Re-use and recycling

1. Member States shall take measures, as appropriate, to promote the re-use of products and preparing for re-use activities, notably by encouraging the establishment and support of re-use and repair networks, the use of economic instruments, procurement criteria, quantitative objectives or other measures.

Member States shall take measures to promote high quality recycling and, to this end, shall set up separate collections of waste where technically, environmentally and economically practicable and appropriate to meet the necessary quality standards for the relevant recycling sectors.

Subject to Article 10(2), by 2015 separate collection shall be set up for at least the following: paper, metal, plastic and glass.

2. In order to comply with the objectives of this Directive, and move towards a European recycling society with a high level of resource efficiency, Member States shall take the necessary measures designed to achieve the following targets:

- (a) by 2020, the preparing for re-use and the recycling of waste materials such as at least paper, metal, plastic and glass from

households and possibly from other origins as far as these waste streams are similar to waste from households, shall be increased to a minimum of overall 50 % by weight;

- (b) by 2020, the preparing for re-use, recycling and other material recovery, including backfilling operations using waste to substitute other materials, of non-hazardous construction and demolition waste excluding naturally occurring material defined in category 17 05 04 in the list of waste shall be increased to a minimum of 70 % by weight.

3. The Commission shall establish detailed rules on the application and calculation methods for verifying compliance with the targets set out in paragraph 2 of this Article, considering Regulation (EC) No 2150/2002 of the European Parliament and of the Council of 25 November 2002 on waste statistics (1). These can include transition periods for Member States which, in 2008, recycled less than 5 % of either categories of waste referred to in paragraph 2. Those measures, designed to amend non-essential elements of this Directive by supplementing it, shall be adopted in accordance with the regulatory procedure with scrutiny referred to in Article 39(2) of this Directive.

4. By 31 December 2014 at the latest, the Commission shall examine the measures and the targets referred to in paragraph 2 with a view to, if necessary, reinforcing the targets and considering the setting of targets for other waste streams. The report of the Commission, accompanied by a proposal if appropriate, shall be sent to the European Parliament and the Council. In its report, the Commission shall take into account the relevant environmental, economic and social impacts of setting the targets.

5. Every three years, in accordance with Article 37, Member States shall report to the Commission on their record with regard to meeting the targets. If targets are not met, this report shall include the reasons for failure and the actions the Member State intends to take to meet those targets.

B Extract from [Decision 2011/753/EU](#)

COMMISSION DECISION of 18 November 2011 establishing rules and calculation methods for verifying compliance with the targets set in Article 11(2) of Directive 2008/98/EC of the European Parliament and of the Council.

Article 1

Definitions

In addition to the definitions laid down in Article 3 of Directive 2008/98/EC, the following definitions shall apply for the purposes of this Decision:

- (1) 'household waste' means waste generated by households;
- (2) 'similar waste' means waste in nature and composition comparable to household waste, excluding production waste and waste from agriculture and forestry;
- (3) 'municipal waste' means household waste and similar waste;
- (4) 'construction and demolition waste' means waste corresponding to the waste codes in Chapter 17 of the Annex to Commission Decision 2000/532/EC (3), excluding hazardous waste and naturally occurring material as defined in Category 17 05 04;
- (5) 'material recovery' means any recovery operation, excluding energy recovery and the reprocessing into materials which are to be used as fuel;
- (6) 'backfilling' means a recovery operation where suitable waste is used for reclamation purposes in excavated areas or for engineering purposes in landscaping and where the waste is a substitute for non-waste materials.

Article 2

General requirements

For the purposes of verifying compliance with the targets set in Article 11(2) of Directive 2008/98/EC, the following rules shall apply:

- (1) Member States shall verify compliance with the targets set in Article 11(2) of Directive 2008/98/EC by calculating the weight of the waste streams which are generated and the waste streams which are prepared for reuse, recycled or have undergone other material recovery in 1 calendar year.
- (2) The weight of the waste prepared for reuse, recycled or materially recovered shall be determined by calculating the input waste used in the preparation for reuse or the final recycling or other final material recovery processes. A preparatory operation prior to the submission of the waste to a recovery or disposal operation is not a final recycling or other final material recovery operation. Where waste is collected separately or the output of a sorting plant is sent to recycling or other material recovery processes without significant losses, that waste may be considered the weight of the waste which is prepared for reuse, recycled or has undergone other material recovery.
- (3) The amount of waste prepared for reuse shall be included in the amount of recycled waste and shall not be reported separately.
- (4) Where waste is sent for preparation for reuse, recycling or other material recovery in another Member State, it may only be counted toward the targets of the Member State in which it has been collected.

(5) Where waste is exported out of the Union for preparation for reuse, recycling or other material recovery, it shall be counted as prepared for reuse, recycled or having undergone other material recovery only where there is sound evidence showing compliance of the shipment with the provisions of Regulation (EC) No 1013/2006 of the European Parliament and of the Council (1), and in particular Article 49(2) thereof.

(6) Where the target calculation is applied to the aerobic or anaerobic treatment of biodegradable waste, the input to the aerobic or anaerobic treatment may be counted as recycled where that treatment generates compost or digestate which, following any further necessary reprocessing, is used as a recycled product, material or substance for land treatment resulting in benefit to agriculture or ecological improvement.

Article 4

Construction and demolition waste

1. For the calculation of the target set in Article 11(2)(b) of Directive 2008/98/EC with regard to construction and demolition waste, Member States shall apply the calculation method set out in Annex III to this Decision.
2. Member States' implementation reports on construction and demolition waste shall comply with the specific requirements in Annex III.
3. The amount of waste used for backfilling operations shall be reported separately from the amount of waste prepared for reuse or recycled or used for other material recovery operations. The reprocessing of waste into materials that are to be used for backfilling operations is also to be reported as backfilling.

Article 5

Reporting by Member States

1. Member States shall report their progress to the Commission with regard to meeting the targets set in Article 11(2) of Directive 2008/98/EC by means of the implementation report referred to in Article 37 thereof.
2. Member States shall provide data in the implementation reports on the state of preparation for reuse, recycling and material recovery of the respective waste streams for either each year of the 3-year reporting period or for the years of the reporting periods laid down in Annex I, Section 5 to Regulation (EC) No 2150/2002.
3. In the implementation report covering the year 2020, Member States shall demonstrate compliance with the targets set in Article 11(2) of Directive 2008/98/EC for the amounts of the respective waste streams generated and recycled or recovered in the year 2020.
4. Member States shall transmit the data and metadata required by this Decision to the Commission in electronic form, by means of the interchange standard set up by Eurostat

Annex III to [Decision 2011/753/EU](#)

Methods for the calculation of the target for construction and demolition waste referred to in article 4(1) of this decision

Calculation method

Recovery rate of construction and demolition waste; in % =

$$\frac{\text{Materially recovered amount of construction and demolition waste}}{\text{Total amount of generated construction and demolition waste}}$$

Specific requirements for Member State implementation reports

- 1) Reporting on the materially recovered amounts of construction and demolition waste (numerator of the formula) shall include only the following codes of the Annex to Decision 2000/532/EC:
List of Waste, Chapter 17 – Construction and demolition waste:
17 01 01, 17 01 02, 17 01 03, 17 01 07, 17 02 01, 17 02 02, 17 02 03, 17 03 02, 17 04 01, 17 04 02, 17 04 03, 17 04 04, 17 04 05, 17 04 06, 17 04 07, 17 04 11, 17 05 08, 17 06 04, 17 08 02, 17 09 04

List of Waste, subchapter 19 12 – Waste from mechanical treatment of waste (for example sorting, crushing, compacting or pelletising), if it is generated from the treatment of construction and demolition waste:
19 12 01, 19 12 02, 19 12 03, 19 12 04, 19 12 05, 19 12 07, 19 12 09
Member States shall explain, in a report to be submitted together with the data, how double-counting of waste is avoided.
- 2) Construction and demolition waste generation shall be reported according to Regulation (EC) No 2150/2002 (denominator of the formula) containing:
 - a) waste generated by Section F of the NACE Rev. 2 code as mentioned in Annex I, Section 8, item No 17 to that Regulation consisting of the following waste codes as defined in Annex I, Section 2 to that Regulation:
 - 06.1. – Metallic waste, ferrous
 - 06.2. – Metallic waste, non-ferrous
 - 06.3. – Metallic waste, mixed
 - 07.1. – Glass waste
 - 07.4. – Plastics
 - 07.5. – Wood
 - b) the total of the waste category (over all economic activities):
 - Mineral construction and demolition waste as defined in Annex III to that Regulation.
- 3) Member States may alternatively report on the recycling and material recovery of construction and demolition waste based on their own reporting system. In this case they shall submit, together with the data, a report explaining which materials are covered, and how the data relates to the data on construction and demolition waste to be reported pursuant to Regulation (EC) No 2150/2002. If the data based on the reporting system of the Member State are more precise than the data provided according to that Regulation the compliance with the target shall be assessed based on the data from the Member State's reporting system.

Table B-1 Waste types covered by the WFD target for recovery of C&D waste. The waste types included in the present study are highlighted in red.

Code	Waste type
17	17 CONSTRUCTION AND DEMOLITION WASTES
17 01	17 01 Concrete, bricks, tiles and ceramics
17 01 01	concrete
17 01 02	bricks
17 01 03	tiles and ceramics
17 01 07	mixtures of concrete, bricks, tiles and ceramics not containing dangerous substances
17 02	Wood, glass and plastic
17 02 01	wood
17 02 02	glass
17 02 03	plastic
1703	Bituminous mixtures, coal tar and tarred products
17 03 02	bituminous mixtures not containing coal tar
17 04	Metals (including their alloys)
17 04 01	copper, bronze, brass
17 04 02	aluminium
17 04 03	lead
17 04 04	zinc
17 04 05	iron and steel
17 04 06	tin
17 04 07	mixed metals
17 04 11	cables not containing oil, coal tar and other dangerous substances
17 05	Soil (including excavated soil from contaminated sites) – stones and dredging spoil
17 05 08	track ballast not containing dangerous substances
17 06	Insulation materials and asbestos-containing construction materials
17 06 04	insulation materials not containing asbestos or dangerous substances
17 08	Gypsum-based construction material
17 08 02	gypsum-based construction materials not contaminated with dangerous substances
17 09	Other construction and demolition wastes
17 09 04	mixed construction and demolition wastes not containing mercury, PCB or other dangerous substances
19 12	Wastes from the mechanical treatment of waste if it is generated from the treatment of C&D waste
19 12 01	paper and cardboard
19 12 02	ferrous metal
19 12 03	non-ferrous metal
19 12 04	plastic and rubber
19 12 05	glass
19 12 07	wood not containing dangerous substances
19 12 09	minerals (for example sand, stones)

C European harmonized standards

Aggregates

- EN 12620 Aggregates for concrete
- EN 13043 Aggregates for bituminous mixtures and surface treatments for roads, airfields and other trafficked areas
- EN 13055 Lightweight aggregates
 - o Part 1: Lightweight aggregates for concrete, mortar and grout
 - o Part 2: Lightweight aggregates for bituminous mixtures and surface treatments and for unbound and bound applications
- EN 13139 Aggregates for mortar
- EN 13242 Aggregates for unbound and hydraulically bound materials for use in civil engineering work and road construction.
- EN 13450 Aggregates for railway ballast

Unbound mixtures

- EN 13285 Unbound mixtures – Specification

Blast furnace slag

- EN 15167 Ground granulated blast furnace slag for use in concrete, mortar and grout

Asphalt

- EN 12697 Bituminous mixtures – Test methods for hot mix asphalt
 - Part 1: Soluble binder content.
 - Part 2: Particle size distribution.
 - Part 3: Bitumen recovery: Rotary evaporator.
 - Part 4: Bitumen recovery: Fractionating column.
 - Part 5: Determination of the maximum density.
 - Part 6: Determination of bulk density of bituminous specimen by hydro-static method.
 - Part 7: Determination of bulk density of bituminous specimens by gamma rays.
 - Part 8: Determination of the air voids content of bituminous materials.
 - Part 9: Determination of the reference density, gyrator compactor.
 - Part 10: Compactability.
 - Part 11: Determination of the compatibility between aggregate and bitumen.
 - Part 12: Determination of the water sensitivity of bituminous specimen.
 - Part 13: Temperature measurement.
 - Part 14: Water content.
 - Part 15: Determination of the segregation sensitivity of bituminous mixtures.
 - Part 16: Abrasion by studded tyres.
 - Part 17: Partial loss of porous asphalt specimen.
 - Part 18: Binder drainage from porous asphalt.
 - Part 19: Permeability of specimen.
 - Part 20: Indentation using cube or marshall specimen.
 - Part 21: Indentation using plate specimens.
 - Part 22: Wheel tracking.
 - Part 23: Determination of the indirect tensile strength of bituminous specimens.
 - Part 24: Resistance to fatigue.
 - Part 25: Dynamic creep test.
 - Part 26: Stiffness.
 - Part 27: Sampling.
 - Part 28: Preparation of samples for determining binder content, water content and grading.
 - Part 29: Determination of the dimensions of bituminous specimen.
 - Part 30: Specimen preparation, impact compactor.
 - Part 31: Specimen preparation, gyratory compactor.
 - Part 32: Laboratory compaction of bituminous mixtures by a vibratory compactor.
 - Part 33: Specimen preparation, slab compactor.
 - Part 34: Marshall test.
 - Part 35: Laboratory mixing.

Part 36: Method for the determination of the thickness of a bituminous pavement.
 Part 37: Hot sand test for the adhesivity of binder on precoated chippings for HRA.
 Part 42: Amount of foreign matter in reclaimed asphalt.
 ...up to part 49

- EN 12970 Mastic asphalt for waterproofing. Definitions, requirements and test methods
- EN 13108 Bituminous mixtures – Material specifications – Part 1 up to part 8

Road marking materials

- EN 1423 Road marking materials. Drop on materials. Glass beads, antiskid aggregates and mixtures of the two
- EN 1424 Road marking materials – Premix glass beads

Concrete

- EN 206-1 Concrete – Part 1: specification, performance, production and conformity
- EN 450 Fly ash for concrete
- EN 12273 Slurry surfacing – requirements
- EN 12274 Slurry surfacing – test methods part 1-6
- EN 12350 Testing fresh concrete
- EN 12390 Testing hardened concrete
- EN 14488 Testing sprayed concrete

Ceramic tiles

- EN ISO 10545-1 Ceramic tiles – Part 1: Sampling and basis for acceptance

Gypsum

Standards set up by CEN/TC 241 include:

- EN 520 Gypsum plasterboards – Definitions, requirements and test methods
- EN 12859 Gypsum blocks – Definitions, requirements and test methods
- EN 13279-1 Gypsum binders and gypsum plasters – Part 1: definitions and requirements
- EN 13279-2 Gypsum binders and gypsum plasters – Part 2: Test methods
- EN 13454-1 Calcium sulphate floor screed – part 1: Definition and requirements
- EN 13454-2 Calcium sulphate floor screed – part 2: Test methods
- EN 13815 Fibrous gypsum plaster casts – Definitions, requirements and test methods
- EN 13915 Prefabricated gypsum plasterboard panels with a cellular paperboard core - Definitions, requirements and test methods
- EN 13950 Gypsum plasterboard thermal/acoustic insulation composite panels – Definitions, requirements and test methods
- EN 14190 Gypsum plasterboard products from reprocessing – Definitions, requirements and test methods
- EN 14209 Preformed plasterboard cornices – Definitions, requirements and test methods
- EN 14246 Gypsum elements for suspended ceilings – Definitions, requirements and test methods
- EN 15283-1 Gypsum boards with fibrous reinforcement – Definitions, requirements and test methods – Part 1: Gypsum boards with mat reinforcement
- EN 15283-2 Gypsum boards with fibrous reinforcement – Definitions, requirements and test methods – Part 2: Gypsum fibre boards

Waste

- EN 12457 Characterisation of waste – Leaching – Compliance test for leaching of granular waste materials and sludges
 - Part 1: One stage batch test at a liquid to solid ratio of 2 l/kg for materials with high solid content and with particle size below 4 mm (without or with size reduction)
 - Part 3: Two stage batch test at a liquid to solid ratio of 2 l/kg and 8 l/kg for materials with high solid content and with particle size below 4 mm (without or with size reduction)
- EN 13656 Characterization of waste. Microwave assisted digestion with hydrofluoric (HF), nitric (HNO₃), and hydrochloric (HCl) acid mixture for subsequent determination of elements

- EN 13657 Characterization of waste. Digestion for subsequent determination of aqua regia soluble portion of elements
- EN 14039 Characterization of waste. Determination of hydrocarbon content in the range of C10 to C40 by gas chromatography
- CEN/TS 14405 Characterization of waste – Leaching behaviour tests – Up-flow percolation test (under specified conditions)
- EN 15308 Characterization of waste. Determination of selected polychlorinated biphenyls (PCB) in solid waste by using capillary gas chromatography with electron capture or mass spectrometric detection

Environmental product declarations

- EN 15804 Sustainability of construction works – Environmental product declarations – Core rules for the product category of construction products.

Test methods

- EN 932-1: Tests for general properties of aggregates – Part 1: Methods for sampling
- EN 933-11 Tests for geometrical properties of aggregates – Part 11 Classification test for the constituents of coarse recycled aggregate
- EN 1097-1 Tests for mechanical and physical properties of aggregates
 - o Part 1: Determination of the resistance to wear (micro-Deval)
 - o Part 2: Methods for the determination of resistance to fragmentation (Los Angeles)

Danish standards

- DS 423 Testing of concrete Part 1-45
- DS 2426 EN 206-1:2011 Concrete – Materials – Rules for application of EN 206-1 in Denmark
- DS/CEN/TR 15678:2008 Concrete – Release of regulated dangerous substances into soil, groundwater and surface water – Test method for new or unapproved constituents of concrete and for production concretes

Finnish standard

SFS 5884 (2001). Production control of reclaimed concrete for earth construction. Suomen standardointiliitto. Finnish Standards Association (SFS). (in Finnish)

ISO standard

ISO 14044 (2006) Environmental management – Life cycle assessment – Requirements and guidelines.

D Examples on additives in bituminous binders

(NIOSH, 2000.)

Type	Examples
1. Antioxidants	Calcium salts Carbon Lead compounds
2. Antistripping agents	Amines Lime
3. Combinations	Blends of plastics (9) and rubbers (10)
4. Extenders	Lignin Sulfur
5. Fibers	Manufactured: Cellulose Fiberglass Mineral Polyester Polypropylene Natural: Asbestos Rock wool
6. Fillers	Carbon black Mineral filler: Crusher fines Fly ash Lime Portland cement
7. Hydrocarbons	Hard and natural asphalts Recycling and rejuvenating oils
8. Oxidants	Manganese salts
9. Plastics	Ethylene acrylate copolymers Ethylene propylene Ethyl-vinyl-acetate Polyethylene/polypropylene Polyolefins Polyvinyl chloride
10. Rubbers:	Natural rubber
Block copolymers	Styrene-butadiene
Natural latex	Polychloroprene latex
Reclaimed	Styrene-butadiene-styrene, styrene-isoprene-styrene
Synthetic latex	Crumb-rubber-modifier
11. Waste materials	Glass Recycled tires Roofing shingles
12. Miscellaneous	Coal liquefaction products Components of shale oil Deicing calcium chloride granules Petroleum distillates: Diesel and other gas oils Gasoline Kerosene Naphthas Stoddard solvent Shale oil residues Silicones

E Using leaching test results to assess the release of substances from granular C&D waste

by Anke Oberender and Ole Hjelm, DHI

Source terms based on leaching tests for granular materials

When construction and demolition wastes (C&D wastes) are recovered (i.e. re-used, recycled, backfilled) or landfilled, they may be placed above ground, underground or even submerged in water. As a result of contact to water substances present in the C&D waste (organic or inorganic substances) may be released to the surrounding environment. The environmental impact of any released substances on soil, groundwater and/or surface water can be assessed near field or far field, for example depending on the legal requirements (e.g. POC directly at the edge of the construction, 30 m away or 100 m from it).

Substances are released into the water and transported to and through the soil, groundwater or surface water. When passing through those environmental compartments the leachate and the substances in it are subject to attenuation (the combined effect of dilution, dispersion, retention/sorption and degradation/reaction). The nature of substances, the degree to which they are attenuated, and the sensitivity of the surrounding environment are some of the factors determining the potential environmental impact and its effect.

Results from leaching tests can be used to evaluate the environmental impact on soil, ground- or surface water of a given material applied in a given manner. However, the test results alone do not provide the answer. They must always be interpreted within the framework of a specific application/construction scenario. The link between the source (release from the C&D waste), transport and spreading of substances and their impact on soil, groundwater or surface water depends on how water (in this case precipitation) comes into contact with (percolates through) the granular C&D waste, the mode of water contact being crucial for the assessment of the release. In this report two basic leaching scenarios can be distinguished – C&D waste is used above ground or underground and precipitation may infiltrate into and through the material.

The leaching data that was collected comprises primarily results from batch leaching tests and column leaching tests (also called percolation test), respectively. Both types of leaching tests are carried out at defined L/S ratios or ranges of L/S ratios under near equilibrium or near local equilibrium conditions. L/S describes the liquid to solid ratio in the test. In the context of leaching tests, L is the volume of liquid (leachant, L) which is in contact with a certain amount of solid (dry mass, S). The unit of L/S is l/kg (which is equal to ml/g and m³/tonne). The liquids collected from the leaching tests are called eluates.

Batch leaching tests are conducted at fixed L/S ratio (e.g L/S 2 or L/S 10) and give therefore only one result. A batch leaching test performed at L/S 10 will provide an average value for release for the entire L/S range 0 to 10 l/kg. In a column test, on the other hand, several eluates for L/S ratios ranging from 0.1 l/kg to 10 l/kg are collected for analysis. Results from a column leaching test provide a relatively detailed description of the leaching, as average values for several eluate fraction are provided, e.g. L/S = 0.0-0.1 l/kg, 0.1-0.2 l/kg, 0.2-0.5 l/kg etc.

Application of leaching test results to construction scenarios

The expression of leaching test results as a function of L/S allows one to compare the results of different (equilibrium) leaching tests to each other and with some caution to relate laboratory leaching results to the expected leaching situation for a recovery or landfilling scenario of granular C&D waste. For a specific recovery (application) scenario the L/S scale can be translated to a time scale if the total mass of waste, the height of the material, the average bulk density and the net infiltration are known. The relationship between time and L/S for a given scenario may be determined from the following equation:

$$T = (L/S) \times d \times H / I \quad (\text{Ea})$$

where:

- T = the time since the first leachate was produced (year)
- L = the total volume of leachate produced at time T (l)
- S = the total mass of waste recovered/deposited (dry bulk weight) (t)
- d = the average dry bulk density of the recovered/deposited waste (t/m^3)
- H = the average height (m)
- l = the net rate of infiltration (mm/year)

The amount of waste recovered, for example waste used as an aggregate in unbound layers of a road, can be regarded as the mass S . The amount of precipitation infiltrating into the waste can be regarded as the volume L .

In order to compare different recovery scenarios with each other and to compare with the landfilling of C&D waste, the amount of waste recovered or landfilled is set to be equal in all cases - for each scenario an amount of 1 m^3 (i.e. height, width and length are $1 \text{ m} \times 1 \text{ m} \times 1 \text{ m}$) of material is said to be in contact with water. This is referred to as a functional unit.

The service life is assumed to be 100 years. The bulk density for a specific granular C&D waste type is fixed, but will vary for different types of C&D waste. The net infiltration varies for different recovery scenarios. The net infiltration for bound applications (e.g. C&D waste is used in a road base and is covered by a relatively impermeable layer, e.g. asphalt) will often be lower than for C&D waste used in backfilling activities. Consequently, only net infiltration will vary for the chosen scenarios for a given C&D waste.

Calculations showing how long time, T , it would take to reach certain L/S values for various scenarios are shown in Table E-1 below. A dry bulk density of 1.8 t/m^3 has been assumed for this example. The annual rate of infiltration represents different scenarios.

Table E-1 Calculations showing how long time, T , it will take to reach a certain L/S value for various scenarios.

Height <i>m</i>	Width <i>m</i>	Length <i>m</i>	Dry bulk density <i>t/m³</i>	Infiltration <i>mm/year</i>	Year <i>year</i>	<i>L/S</i> <i>l/kg or m³/t</i>
1	1	1	1,8	50	100	2.8
1	1	1	1,8	100	100	5.6
1	1	1	1,8	150	100	8.3
1	1	1	1,8	200	100	11.1
1	1	1	1,8	300	100	16.7

Table E-1 shows that a scenario with a low rate of infiltration and a service life of 100 years will correspond to an L/S of 2.8 l/kg. For a scenario where the net infiltration is relatively high, e.g. 300 mm/year, an L/S of 16.7 l/kg may be reached after 100 years.

Based on the derived L/S value one can estimate the eluate concentration or leached amount for the corresponding L/S from a leaching test. In cases where the derived L/S ratio for a given scenario goes beyond the L/S range covered by a leaching test, one will have to estimate the release. For example a typical column leaching test will stop at L/S 10 l/kg. A very conservative approach would be to extrapolate results assuming, that leaching beyond L/S 10 l/kg will continue at the same rate as up to L/S 10 l/kg. Another approach is based on the CSTR- model (continuously stirred tank reactor). This approach is based on the assumption, that the leaching of the contaminants under consideration can be described as an exponentially decreasing function of L/S or time. The concentration C of a contaminant in the leachate (or eluate, from a laboratory leaching test) may be estimated as follows:

$$C = C_0 \times e^{-(L/S) \times \kappa} \quad (\text{Eb})$$

where:

- C = concentration of a contaminant (mg/l)
- C_0 = initial peak concentration of the contaminant in the leachate (mg/l)
- L/S = is the accumulated liquid to solid ratio corresponding to the concentration C (l/kg)

κ = kinetic constant describing the rate of decrease of the concentration as a function of L/S for a given material and a given contaminant (kg/l)
The kinetic constant κ may be estimated from laboratory leaching data.
Based on the data available in this project, different approaches have been used.

It should be noted that the application of results from batch and column leaching test cannot take longer term changes of the materials into account. In order to enable at least a qualitative assessment of the effect of *e.g.* changes in pH (for example as a result of carbonation of alkaline materials), it is recommended also to carry out a pH dependence leaching test.