LICENTIATE THESIS

Environmental Perspectives on Urban Material Stocks used in Construction

– Granular Materials

Simon Magnusson

Soil Mechanics

Luleå University of Technology

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– Granular materials

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Licentiate Thesis

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ABSTRACT

The peoples demand of functions and services in cities is the driver for energy and material flows. Most people in the world are now living in urban areas. In order to achieve a sustainable development of cities, both resource use and environmental impact have to be reduced. For construction activities, an important aspect is to increase the reuse of construction materials. From a resource perspective, the urban demand for construction of buildings, infrastructure and other facilities results in materials accumulated in constructions but also in other applications and in landfills. The materials can be described as the urban material stock where some materials are used and others are not used, i.e. wasted. There are many cases where material stocks are used for construction purposes. For example, used concrete and bricks, excavated soil and rock from construction projects and other wasted materials such as rubber from tires can be crushed, shredded and sorted to granules and used in many different construction applications. Different perspectives can be applied when assessing the environmental impacts of using stocked material in construction. The overall aim of this thesis is to study the environmental impacts of using granular soil, rock and rubber in construction. For soil and rock, the aim is to study the environmental impact of material management in urban areas. For granular rubber, the aim is to study the environmental impact of artificial turf from a life cycle perspective and from different infill materials of recycled and new rubber and plastics.

The literature of excavated soil and rock was reviewed in order to identify and quantify the material flows and greenhouse gas (GHG) emissions from the management of soil and rock materials. For artificial turf and the different infill materials, a life cycle approach was used to quantify the energy use and GHG emissions. A chemical analysis of potential chemical leaching from the different infill materials to water was conducted in order to compare potential local emissions to water.

Based on the results, it was concluded that the knowledge about the urban flows of excavated soil and rock is lacking in terms of patterns, quantities, qualities and its environmental performance. A resource perspective is missing in the literature. However, the recycling of soil and rock can reduce resource use and GHG emissions. It was suggested that models are developed that take into account future material demand and availability to soils and rock. From such information it would be able to assess sustainable management practices and the possibilities of sharing materials between urban construction projects in order to reduce resource use and environmental impact.

It was concluded that for the life cycle of artificial turf, the production of construction materials contributes largely to energy use and GHG emissions. Differences in terms of energy use and GHG emissions for the production of infill materials are large. The production of new material required more energy and resulted in more GHG emissions than using recycled rubber. The potential release of substances from infill materials to water were shown to be possible for all infill materials analyzed. Previous assessments of local environmental impacts of using infills generally concludes that the impacts are small. These assessments are primarily focused on infill of recycled tires. It is therefore concluded that environmental assessments of local impact should include all infill types.

Environmental assessments of using stocked materials in construction should take into consideration the material applications´ significance for the environmental impacts at a higher system level. Broader system boundaries in environmental assessments will reduce the risk for sub-optimizations when taking decisions on how materials should be used in construction.

Key words: Construction; Stock; Environmental Systems Analysis; Excavated soil and rock; Rubber

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Thanks to my former colleague Susanna Toller and to my colleague Björn Frostell who both got me interested in environmental systems analysis in the first place.

Also, thanks to my friends and caring family who have always been there for me.

LIST OF PAPERS

Paper I

Sustainable management of excavated soil and rock in urban areas – A literature review

Simon Magnusson, Kristina Lundberg, Bo Svedberg & Sven Knutsson

Journal of Cleaner Production 93 (2015), pp. 18-25

Paper II

Analysis of energy use, emissions of greenhouse gases and prioritized substances for artificial turf and its construction materials

Simon Magnusson, Josef Mácsik [Submitted]

LIST OF ABBREVIATIONS

CO_2, Carbon Dioxide

C&D, Construction and demolition

DOC, Dissolved Organic Carbon

EPDM, Ethylene Propylene Diene Monomer

GC-MS, Gas Chromatography Mass Spectrometry

GHG, Greenhouse gas

LCA, Life Cycle Assessment

R-EPDM, Recycled Ethylene Propylene Diene Monomer

PAH, Polycyclic aromatic hydrocarbon

SBR, Styrene Butadiene Rubber

TPE, Thermoplastic Elastomer

1. INTRODUCTION

1.1. Background and problem definition

1.1.1. Function demands and urban material stocks

People's demand for functions is a driver for the use of resources. In urban areas, were about half of earth's population now live, the material and energy flows are intensive. Increased demands presume a steady supply of food, drinking water and shelter etc. Access to natural resources is crucial, such as to fertile soil for farming, clean fresh water, energy and construction materials. The flows of material and energy in urban areas has been described by Wohlman (1965) as the urban metabolism.

People's demands are many. Over time, the demands changes. Functions get outdated and need to be changed or replaced. Buildings and infrastructure are constructions that provide important functions. Construction activities requires large amounts of materials and generates large amounts of waste. Natural resources are extracted and used in construction, at the same time as wasted materials is accumulated in landfills (Johansson et al., 2013). Construction waste contain of different materials such as concrete, ceramics, metal, wood, plastics, soil, stone, glass, insulation materials and others (Coelho and Brito, 2013; Hiete et al., 2011).

The use of natural resources in the construction sector is large and quarry materials has shown to account for 50 % of materials use (Ravetz, 2008). Sand, gravel and crushed rock are extracted from quarries and used as construction materials in roads, buildings and other civil engineering projects. Common applications are in road subbase, asphalt and concrete (Arulrajah et al., 2013). EU statistics show that the annual use of non-metallic minerals extracted from quarries is about 4.3 ton/ capita. At the same time, generated construction waste is about 1.1 ton/ capita (EUROSTAT 2012; 2016). The resource use in construction is extensive and need to develop in a more sustainable direction. Resource use has been pointed out as major issue for a sustainable development in United Nations Sustainable development goals (UN, 2016).

Materials are generated during digging, blasting and demolition activities. Some of the materials can be recycled as construction materials. Recycling of stocked materials can therefore decrease resource use. Many of the wasted materials need to be treated before use. By sorting, crushing or shredding to fine granules, it is possible to use them in different construction applications (Arulrajah et al., 2013). The geotechnical material properties are mainly determined by the behavior of the bulk of granules rather than the behavior of every single one. The potential applications in construction are therefore many.

1.1.2. Environmental systems analysis of construction

Environmental systems analysis (ESA) can be used to study the environmental impact of an object or a process such as products and manufacturing (Finnveden and Moberg, 2005). ESA can be used as a basis for decision making by clarifying the environmental impacts. Roth and Eklund (2003) describes how ESA can be used for environmental assessment of materials in construction. They identify levels of environmental assessments that are important for different environmental perspectives. Further, they conclude that several levels must be considered in order to assess environmental impacts in a broad perspective.

Roth and Eklund (2003) identify four system levels for the environmental impact of a road:

1) the material level,

2) the road environment level,

3) the narrow life cycle level, and

4) the industrial system level.

The system levels include different aspects and time perspectives. In figure 1, a general illustration of the environmental levels for assessing construction projects is given. The white upper arrow illustrates the life cycle of a construction, from extraction to disposal. The environmental level's system boundaries are illustrated as grey boxes. Depending on the environmental impact of interest, different environmental levels can be considered in the ESA.

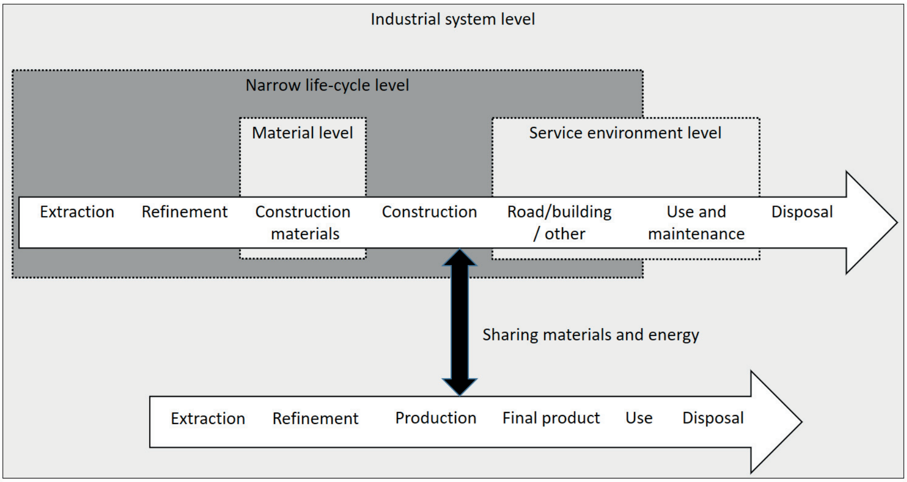

Figure 1. *Levels in ESA of construction. Modified from figure by Roth and Eklund (2003).*

The material level includes intrinsic and extrinsic material properties. Examples of intrinsic properties is mass and chemical composition which both are independent of the environment. The extrinsic material properties vary depending on the surrounding environment. For example, weight is related to gravity and emission of substances is related to the presence of other matter, for example water and air. At the material level, questions can be raised about chemical content and potential emissions to water and air (Roth and Eklund, 2003).
At the road environment level, questions are raised about the environmental impacts at the time the road is constructed and is used. This level includes environmental impacts when the road is used, from the construction itself and from traffic (Roth and Eklund, 2003). Examples of impacts are local emissions to water from the roads' construction materials and from vehicle exhaust. This environmental level is applicable to other constructions or products where there is a need to assess the environmental impact when the item is used. In figure 1, the road environmental level has been renamed to the service environmental level in order to consider other constructions and products than roads.

At the narrow life cycle level, the system boundaries are expanded to include the environmental impact of the production of the road. This level includes the environmental impact of extraction of resources, manufacturing of input materials and construction of the road (Roth and Eklund, 2003).

The industrial system level concerns the environmental impact of the whole life cycle of the road, from resource extraction to final disposal. The level also includes parallel processes and has been illustrated with a white arrow (lower arrow) in figure 1. The parallel process share material and energy with the road system. For example, waste generated in other industrial processes can be recycled as construction materials in roads (Roth and Eklund, 2003). At the industrial system level, the system boundaries are therefore expanded to include the life cycle of the industrial processes. The industrial system level has high complexity since material and energy flows between industrial processes are intertwined.

1.1.3. Environmental Systems Analysis of two granular material types

In this thesis, questions are raised about how the use of recycled granular materials in construction perform at different environmental levels. Two granular materials have been chosen for further study:

1) granules of excavated soil and rock and
2) granules of rubber from tires and other products.

Excavated soil and rock are materials generated at construction sites when digging and blasting in the ground. Excavated soil and rock can be used in construction of roads, railways, buildings and other facilities. It can replace quarry products such as soil, sand, gravel and stone in many applications such as in road sub-base, concrete, asphalt, and used for general fill purposes. In this thesis, all types of management are included together with its environmental impact. The management of soil and rock in construction projects require handling and transportation of large and heavy volumes. For urban areas where construction activities are frequent, the management can be comprehensive (Lundberg et al., 2012). However, the possibilities to recycle and share soil and rock materials between projects may also be large (CL: AIRE, 2013). This thesis raises questions about the environmental impacts of the material management and the potential of recycling. The environmental impacts of interest are therefore at an industrial level and involves a large number of construction projects.

The second material type is rubber from tires and other products which can be used for different construction purposes. For example, it can be used as light weight filling materials in road embankments and as shock absorbing infills in sport surfaces of artificial turf. Most knowledge about environmental impact of artificial turf concerns the use of tire rubber infill. In this thesis, questions are raised regarding

- The environmental impact of artificial turf during its life cycle and with different infill materials
- The differences in environmental impact at the material level for recycled rubber and other infill materials

In table 1, the environmental levels studied in this thesis are bolded. Suggestions on other questions at each environmental level are also given in order to clarify the differences between the levels.

Table 1. *Environmental assessment levels for granular soil, rock and rubber in construction. The scopes of the thesis are bolded.*

Environmental Level	Granular soil and rock as construction material in construction applications	Granular rubber as construction material in artificial turf
Industrial	**Soil and rock management** The life cycle of roads and buildings	Rubber management **The life cycle of an artificial turf field with different infill materials**
Narrow life cycle	Production of a road or building	Production of an artificial turf field
Project environment	Local emissions from the construction in use	Local emissions from the construction in use
Material	Content and leaching of soil and rock materials	**Leaching potential of different infill materials**

1.2. Overall aim

The overall aim is to study the environmental impacts of using granular soil, rock and rubber in construction. For granular soil and rock, the aim is to study the environmental impact of urban material management at an industrial level. For granular rubber, the aim is to study the environmental impact at a material and industrial level when rubber is used in artificial turf.

Research questions studied

1) What is the environmental impact of soil and rock management in urban areas? **(paper I)**
2) What is the life cycle environmental impact of an artificial turf field? **(paper II)**
3) What is the leaching potential of different infill materials for artificial turf? **(paper II)**

1.3. Objectives

From the research questions that were developed and presented in the above section, objectives were formulated. In order to expand the knowledge about the environmental impacts from granular soil, rock and rubber, the thesis objectives were to:

1) Develop a model for the flows of soil and rock materials in urban areas.
2) Quantify material flows and GHG emissions from the management of urban soil and rock.
3) Analyze energy use and climate impact from the life cycle of an artificial turf field with different infill materials.
4) Analyze potential emissions of substances from different infill materials.

2. METHODOLOGY

2.1. Developing a model for urban soil and rock material flows

2.1.1. Urban soil and rock management and environmental aspects

The management of soil and rock can cause different types of environmental problems, such as depletion of non-renewable resources, landfilling, contamination, noise, dusting and heavy transports. In a Swedish context, the environmental problems and other aspects of sustainability were in focus in a multi stakeholder project named *Hållbar Materialförsörjning i Stockholms län* – HMFS (Lundberg et al., 2012). The project aimed to gather knowledge from key actors and to identify possible ways to increase the resource effectivity in the system. Problems identified in the system were:

- Large amounts of excavated soil and rock that could be recycled were sent to landfills due to the lack of space and time for recycling practices. At times when recycling was possible, savings in terms of quarry materials, landfill capacity and transportation were made.
- Excavated soil and rock were transported to landfills in other regions due to the decreased number of landfills in the Stockholm county. The demand for excavated soil and rock as cover material on closing landfills were in decline.
- There was no concrete data that could provide basis for decisions by private and public actors in order improve the situation, both in terms of environmental and economic costs.

These issues were further studied in the project Optimass (www.optimass.se) which aimed to improve the conditions for soil and rock management in urban areas. The need of data about environmental impacts of urban soil and rock management was a prioritized issue in the project. In order to answer some questions about environmental impacts such as climate impact and resource efficiency, an urban soil and rock flow model was developed. The model is described in this thesis and in Paper I.

Other projects about environmental and resource aspects of soil and rock management have been launched in Europe. In the Finnish project Absoils and the Slovenian based project SARMa, treatment methods for on-site reuse of soil and rock was tested and assessed (Kreft-burman et al., 2013) and the environmental consequences of recycling construction materials have been assessed (Blengini and Garbarino, 2011). The results from these projects confirm that recycling practices can reduce transportation, landfilling and environmental impact of soil and rock management.

2.1.2. A model for soil and rock material flows

It was decided that the flow model should start from the demand for buildings, roads and other constructions. These demands are important since they are drivers for the generation of excavated soil and rock and the use of quarry material. In addition, they set the framework for how much excavated soil and rock that potentially can be recycled. The demand for applications where excavated soil and rock can replace quarry material, becomes a maximum limit for recycling. The relation between the demand and availability to resources could be described as the degree of self-sufficiency of the system. A high availability to excavated soil and rock could decrease the use of quarry materials, and thereby increase the resource self-sufficiency of the system. It was decided to illustrate self-sufficiency in the soil and rock model. An overview of research involving other resource supply to cities was made in order to find examples of such illustrations. An illustration was found in a study by Welfle et al. (2013), where the degree of self-sufficiency of bioenergy in a city was assessed. In their model, the energy demand was considered as the driver for the energy flows. The energy supply was separated into an internal and an external supply in order to identify the degree of self- sufficiency. The concepts of demand, supply and internal and external supply were integrated in the flow model of soil and rock. Internal and external material suppliers were separated in order to identify the degree of self-sufficiency in the system.

The different flows of soil and rock from construction activities were separated in the model. Construction activities generates excavated soil and rock which can be 1) used on-site 2) used in other projects 3) pretreated before use in other projects 4) stored for later use, 5) used as landfill cover or disposed at landfill (Eras et al., 2013; Lafebre et al., 1998).

A study by McEvoy et al. (2004) quantified the use of construction minerals in the North West of England. They constructed a flow model and quantified the use of construction minerals for buildings, infrastructure and industry etc., together with the fate of wasted materials. With inspiration from their work, a similar structure was used in the flow model of soil and rock to illustrate the fate but also the import, export and the passing through of soil and rock materials in the urban system.

It was decided to also illustrate the material stock in the system. The stock is a potential material resource. Construction activities accumulate materials into the building stock of houses, infrastructure and other facilities. Materials are also accumulated in stocks such as landfills. The concept of urban stocks was borrowed from the research field of urban mining and was integrated in the model. Urban mining is the research field of how to take back valuable metals that have been stocked in urban areas and is no longer in use, i.e. hibernating Johansson et al. (2013). In urban mining, the concepts of active and inactive stocks are used to describe what resources are available and not. These concepts were further used in the soil and rock model.

2.2. Literature review of soil and rock quantities and GHG emissions

From the literature, material quantities for each flow, data on GHG emissions from transportation, recycling activities, quarry production and earthworks were collected. Data were searched for in studies at different construction levels, from case studies of multi projects, to studies of cities, regions and countries. Environmental assessments from different research fields were searched through for data on GHG emissions. The soil and rock model was used to target data of interest among other information. The collected data on material quantities were used to quantify each material flow in the model.

2.3. Life cycle assessment approach for energy use and GHG emissions of artificial turf

For the analysis of environmental impacts of artificial turf, a Life Cycle Assessment (LCA) approach was used. The LCA approach was developed based on the ILCD LCA guidelines (EUR 24708, 2010). Environmental aspects such as energy use and GHG emissions have hardly been described in impact assessments of artificial turf. However, some studies are available. Uhlman et al. (2010) and Holmström (2013), compared artificial turf to natural grass with an LCA approach. Natural grass was concluded to be favorable to artificial turf. However, the opposite results could be seen if the environmental impact was allocated to the number of playing hours that could be provided (Cheng et al., 2014). Other studies have compared the environmental impact of the production of different infill materials (Clauzade et al., 2010; Fiksel et al., 2011; Skenhall et al., 2012). Still, a life cycle perspective of artificial turf which includes the construction, maintenance and final removal of artificial turf is missing. The life cycle of artificial turf is in this thesis analyzed in order to identify material use and processes that significantly have an impact on the environmental performance in terms of energy use and GHG emissions.

An illustration of the system boundaries is given in figure 2.

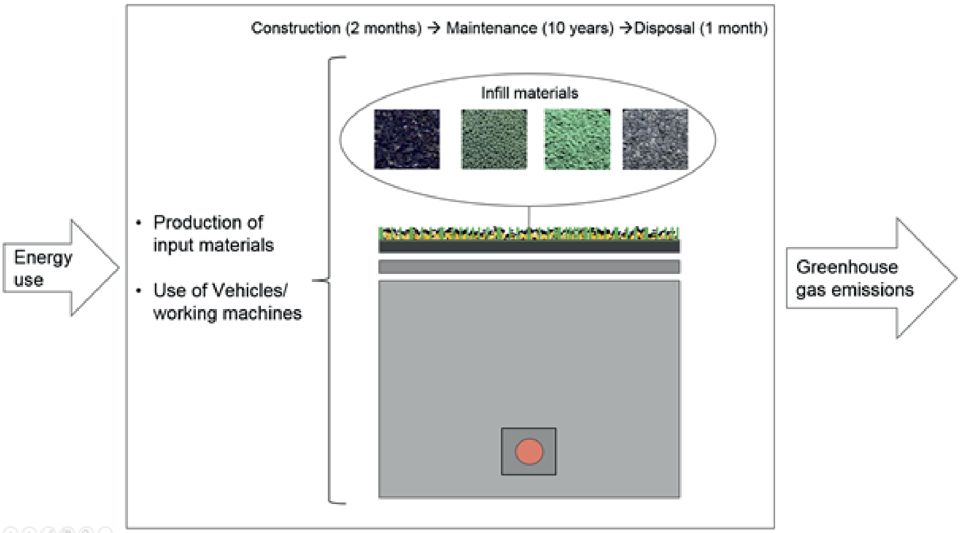

Figure 2. *System boundaries for analysis of energy use and GHG emissions throughout the life cycle of an artificial turf field with different infill materials (Paper II, Chapter 2.1).*

An artificial turf field consist of several layers of materials. The main materials used are crushed rock, plastic pipes for drainage, a shock absorbing rubber layer, a plastic turf mat, sand, rubber or plastic infill and paint for lines. The infill types can be made of different materials. The study included four granular materials; granules of used rubber from car tires (SBR), granules of new thermoplastic (TPE), of new rubber (EPDM) and from used cable rubber and automotive rubber carpets (R-EPDM). The study was based on a fictive case of an artificial turf field in Sweden, constructed and maintained over a ten-year time period and finally removed. The fictive turf football field was a full sized field measuring 7881 m^2.

Detailed inventory data can be found in Paper II. The system boundaries were limited to include production of input materials, use of vehicles and construction machines in order to provide a service life of ten years before final disposal. The field was assumed to be brushed, harrowed, plowed and salted to allow playing during the whole year. Surrounding facilities were excluded such as lighting, paved areas, fencing and grandstands.

Material and energy flows were inventoried. Information about activities and dimensions for construction together with maintenance and removal activities are described in Paper II, chapter 2.3. Manufacturing of machinery such as excavators and tractors used in construction, maintenance, and final removal were not included.

The studied artificial turf system was selected to reflect construction and maintenance praxis used in Sweden and in other Scandinavian countries.

2.4. Analysis of leaching potential to water for different artificial turf infill materials

The chemical content of a material determines what could potentially be emitted to the environment. The leaching potential to water is of interest in order to assess the infill materials´ local environmental impact and depends on how substances are bound to the material and the substance's physical-chemical properties. The infill materials consist of polymers and different additives providing certain properties to the rubber or plastic material such as softness and UV protection. Previous leaching studies have primarily studied SBR infill and concluded that zinc can be emitted to water (Bocca et al., 2009; Bristol and McDermott, 2008; Gomes et al., 2012; Hoefstra 2007; Lim and Walker, 2009; Nilsson et al. 2008; Plesser and Lund, 2004; Ruffino et al., 2013). Zinc leaching has also been detected in leachates from TPE (Ruffino et al., 2013) and EPDM (Nilsson et al., 2008; Plesser and Lund, 2004).

A few studies have analyzed leaching of Polycyclic aromatic hydrocarbon (PAH). These studies have confirmed that PAH: s can be detected in leachates from SBR (Gomes et al., 2012; Plesser and Lund, 2004; Ruffino et al., 2013;) and in leachates from TPE (Ruffino et al., 2013). Data from leaching analysis, together with other laboratory studies have been entered in assessments of local environmental risk to ground water, surface water and soil contamination and assessments of health risk due to dermal contact, ingestion and inhalation. These studies show that the health risks are minimal (Birkholz et al., 2003; Ginsberg et al., 2011; Kim et al., 2012; Lim and Walker, 2009; Menichini et al., 2011; Moretto, 2007; Nilsson et al., 2008; Norwegian Institute of Public Health and the Radium Hospital, 2006; Pavilonis et al., 2014; Ruffino et al., 2013; Vidair et al., 2007). Environmental risk assessments conclude that the risks are small in general (Birkholz et al., 2003; Moretto, 2007; Nilsson et al., 2008; Vidair et al., 2007). However, a review of the risk assessments that was conducted in Paper II, chapter 3.1, shows that most studies concern SBR infill while assessments of other infills are few or completely missing.

It was decided to analyze the infill materials´ leaching potential to water with respect to primarily metals and organic substances. Leachate water from infill were produced by a single-stage shaking test at L / S = 10 in accordance to prEN 12457-2. Analysis of metals, chloride, fluoride, sulfate, phenols and dissolved organic carbon (DOC) followed standardized methods and are presented in detail in Paper II, chapter 2.5. Besides the possible detection of metals, PAH: s and some other substances such as phthalates and phenols, the leachates of organic substances to water have hardly not been studied. A screening of organic substances in the leachates was therefore conducted. Screening means that the substances´ characteristics are identified from the leachate and are matched against a substance data library to identify specific substances. In this method, leachate water was analyzed for semi-volatile organic compounds by using a Gas Chromatography Mass Spectrometry (GC-MS) column instrument. A chemical analysis specifically for substances of environmental and health concern such as PAH:s, PCB7, Chlorobenzes, Chlorophenols, Chlorinated hydrocarbon, and Phthalates were also conducted. The screening method is described further in Paper II, chapter 2.5.

3. RESULTS

3.1. Material flows and GHG emissions from soil and rock materials

3.1.1. A model for soil and rock materials

A model for soil and rock flows was developed. The model is presented in Figure 3.

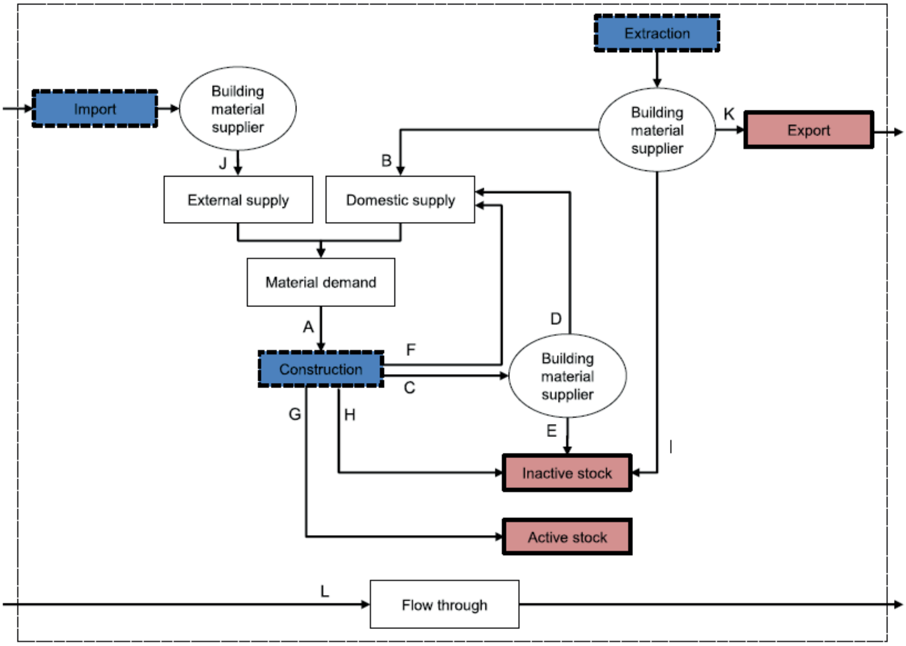

Figure 3. *Conceptual model for soil and rock material flows. Dashed/blue boxes illustrate sources to construction material. Bold/red boxes illustrate whether construction materials are in use, such as stocked in a building, or not in use, such as disposed at landfills (Paper I, chapter 2.2).*

The model illustrates construction materials demand and generation of materials in construction. Sources to construction materials, illustrated with dashed / blue boxes can be either domestic or external. These resources can come from new extracted resources or excess materials from other construction activities. Materials in use, or in stock are illustrated by the bold/ red boxes. These materials can be in use, for example as construction materials in a building. It has been accumulated in the building stock and the materials are part of the active material stock. If the materials do not fulfill a function in the system, for example if they are accumulated at landfills, then the materials can be described as part of the inactive material stock.

The material flows illustrated in figure 3 are referred to as flow *A* – L and are described in detail in Paper I, chapter 2.2. Flow A refers to the demand and use of materials in applications such as road sub-base. Flow B is extracted material from domestic sources such as sand, gravel and rock from quarries. Flow C is the excess soil and rock generated in construction which is sent away as waste to a building material supplier. The building material supplier processes the material before it is used in construction (flow D) or sends the material to landfill (flow E). Flow F is excavated soil and rock generated at the construction site which is either needed on-site for use in construction or needed in other construction projects. The use of flow F materials is often preceded by some kind of treatment, for example crushing and sorting. Flow G is excavated soil and rock that is used in construction while Flow H refers to excess excavated soil and rock which is sent to landfills or similar sites where it is inactively stocked. Flow I is fractions of extracted quarry materials that are not used, such as rock flour. Flow J, K and L respectively refers to the import, export and flow through of excavated soil and rock.

3.1.2. Quantities of urban soil and rock flows

A compilation of known quantities of excavated soil and rock flow is presented in figure 4. The quantities represent annual flows in tons per capita. For each flow, the highest values found in the literature are presented in the figure. A detailed description of the quantities that are presented in this section is given in Paper I, chapter 3.3 and chapter 5. For some flows, very few data could be found in the literature. The production of construction materials primarily extracted from quarries has been estimated in several studies (e.g. McEvoy et al., 2004; Hiete et al., 2011; Huang and Hsu, 2003; Rosado et al., 2014). The quantities of quarry material used per capita was up to 11 tons but large variations were found. The use was about 7 times lower in one case (Huang and Hsu, 2003). Depending on how much material is recycled the use of quarry material will vary. The differences can also partly be explained by fluctuations in economy, which impacts on the construction intensity (Huang and Hsu, 2003).

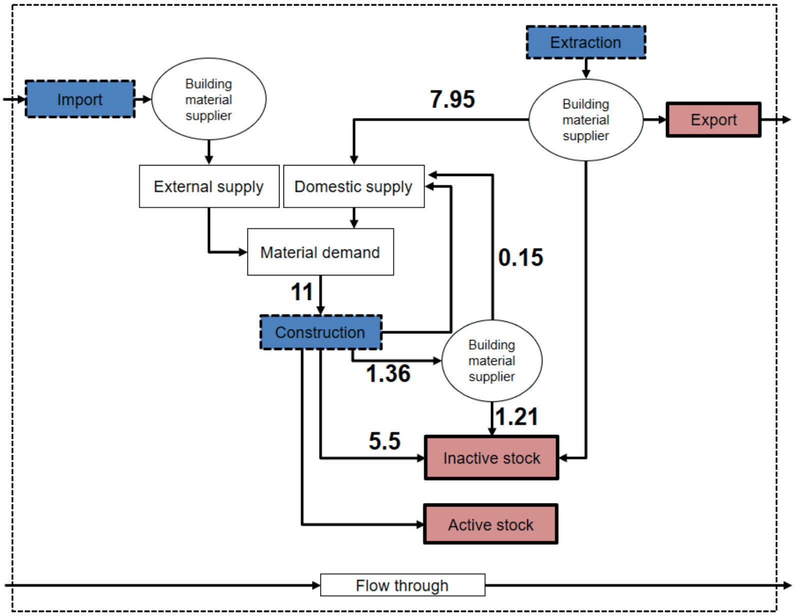

Figure 4. *Maximum values of known annual quantities of construction material demand and excavated soil and rock flows, in tons per capita.*

The use of construction minerals such as sand, gravel and rock from quarries (Flow B) were up to 7. 95 tons/ capita (Hiete et al., 2011). The only data found on quantities of excavated soil and rock that enters and leaves recycling facilities (Flow C, D and E), were from Hiete et al. (2011). Soil and rock sent to recycling facilities (Flow C) were estimated to 1.36 ton/ capita. The supply of recycled soil and rock from recycling facilities (Flow D) to construction were estimated to 0.15 ton/ capita. Landfilling of material from recycling facilities (Flow E) was estimated to 1.21 ton/ capita. (Hiete et al., 2011)

The direct landfilling of excavated soil and rock (Flow H) was up to 5.5 ton/ capita (Forsman et al., 2013). For the other flows of excavated soil and rock shared between construction projects (Flow F), accumulated in the active stock (Flow G), directly landfilled (I) and imported, exported and passing the system (Flow J, K and L), no data was found. For most flows of soil and rock, none or only single estimates of the quantities could be found.

3.1.3. GHG emissions from the management of excavated soil and rock

The environmental impact in terms of GHG emissions for using excavated soil and rock in construction has partly been evaluated. Two studies assessed the GHG emissions from using conventional quarry materials and included extraction, processing and handling of quarry materials. The GHG emissions were estimated to 7.8 kg and 10.3 kg of CO_2 per ton material (Simion et al., 2013; Zuo et al., 2013). Some studies assessed the difference in terms of GHG emissions between using quarry materials compared to recycled soil and rock materials. In one study, the CO_2 savings due to reduced transportation when reusing soil and rock on site were up to 12 kg CO2 / ton (Chittoori et al., 2012). In another study, the CO_2 savings were 2.3 kg CO_2 /ton if the material was shared between construction projects by using a local recycling facility (CL:AIRE, 2013). From a life cycle perspective, another study estimated the CO_2 savings to 14 kg/ ton material when the material was sent to a facility for recycling and reused in other construction projects (Blengini and Garbarino 2010).

3.2. Energy use and GHG emissions from the life cycle of artificial turf

The environmental impact from an artificial turf field was studied from a life cycle perspective. In figure 5 and 6, the energy use and GHG emissions for construction, maintenance and removal of the artificial turf field are presented.

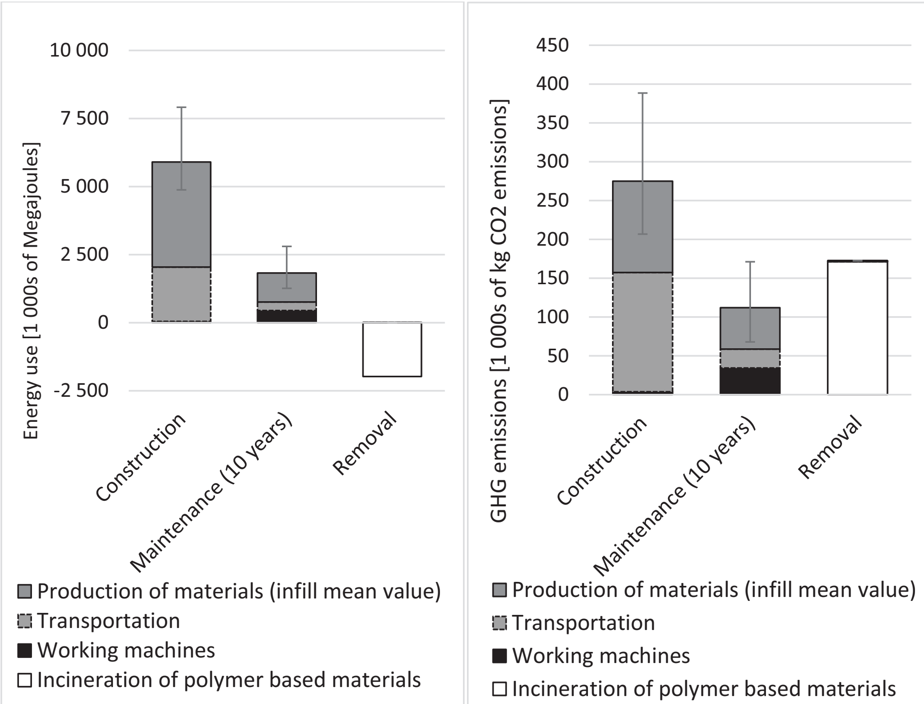

Figure 5 and Figure 6. *Energy use and GHG emissions from construction, maintenance and removal of 7 881 m^2 artificial turf. Mean values for energy use and GHG emissions of the infill were used (Paper II).*

The total energy use was 5.8 GJ and the GHG emissions corresponded to 560 ton CO_2. The energy use and GHG emissions of the infill material in figure 5 and 6 are the mean values for all four infill types. The maximum and minimum values for production of materials are related to the infill alternative used, where the use of SBR and R-EPDM resulted in lowest energy use and GHG emissions and where the use of TPE resulted in the highest. The total energy use and GHG emissions varied by a factor of 1.9 and 1.6 respectively, depending on the infill type used (Paper II, chapter 3.2). The most significant contributions to GHG emissions were primarily the construction, followed by removal and maintenance. In the construction phase, the excavation and use of quarry materials for subbase and the use of infill materials were large contributors to energy use and GHG emissions. Overall, it can be seen that the production of materials makes up the major part of energy use while GHG emissions are due to both material production, transportation and incineration. Energy recovery from incineration of the infill, turf and pad materials contributed to a reduction of the total energy use.

3.3. Potential leaching to water from artificial turf infill materials

The only metal that was detected was zinc which was found in the leachates of SBR and R-EPDM at concentrations of 94 µg/l and 5000 µg/l respectively. These results are further described in Paper II, chapter 3.3.

The detected organic substances are presented in Table 2. These results are presented and described in detail in Paper II, chapter 3.3. In the leachates, PCB7, Chlorobenzenes, Chlorophenols, Chlorinated hydrocarbons and PAH:s were not detected. One PAH derivative was found in the leachate from TPE. Phenols and its derivatives were found in the leachates from SBR, TPE and R-EPDM, similarly to previous studies (Nilsson et al., 2008). Phthalates were detected in the leachate of TPE. Phthalates have been detected in previous studies in leachates from both SBR, TPE and EPDM infill (Nilsson et al., 2008)
The leaching of other specified hydrocarbons was highest from R-EPDM, followed by EPDM and TPE and lowest from SBR. Most of these hydrocarbons contained nitrogen.
For R-EPDM, TPE and SBR, a large part of the leachate was of unspecified aliphatic oil types. The highest leaching was from R-EPDM. For organic substances, the total concentration in leachates were lowest from SBR (437 µg/l), followed by EPDM (702 µg/l) and TPE (1086 µg/l). From R-EPDM, the total leachate of organic substances was 39 023 µg/l, which is about 35- 90 times higher than from the other materials. Of the total organic substances leached from SBR, TPE and R-EPDM, a large share were aliphatic oils of which substances could not be specified.

Table 2. *Leached semi volatile organic substances from infill of SBR, TPE, EPDM and R-EPDM produced by a single stage shaking test at L/S 10 according to EN 12457-2 standard. (Paper II, chapter 3.3)*

	SBR [µg/l]	TPE [µg/l]	EPDM [µg/l]	R-EPDM [µg/l]
PAH16	-	-	-	-
PAH derivatives	-	3.3*	-	-
PCB7, Chlorobenzenes, Chlorophenols and Chlorinated hydrocarbons	-	-	-	-
Phenols, Alcylphenols	1.3	1.9	-	-
Phenol derivatives	1.4	1.8	-	11
Phthalates	-	6.2	-	-
Other specified hydrocarbons				
containing nitrogen	44.9	520	664	1012
not containing nitrogen	0.46	37.7	9.4	-
Oils, unspecified	390	520	-	38 000
Total leachate	**437**	**1 086**	**702**	**39 023**

*Possible derivative

Substances which are known to be harmful for the aquatic environment and / or humans were detected in all infill leachates. Highest concentrations were in the leachate from R-EPDM followed by EPDM, SBR and TPE.

4. DISCUSSION

The knowledge about urban excavated soil and rock flows and its quantities is poor. From the results it was found that of the 8 possible flows of soil and rock identified, only a few have been quantified. The importance of soil and rock management in terms of environmental impact and resource use in construction can be supported by the fact that the annual amount of excavated soil and rock that are landfilled can be as high as 5.5 ton / capita. This amount is significant in relation to the amounts of material that is extracted from quarries, which can be 11 ton / capita. In addition, one study suggest that materials extracted from quarries can correspond to 50 % of total material use in the construction sector (Ravetz, 2008). These findings show that resource efficiency in construction is closely linked to the management of soil and rock materials.

In order to increase recycling and to reduce transportation of soil and rock, information is needed about the material flows. The quantities of construction materials needed and soil and rock materials available for recycling have to be identified. From this information, the potential of sharing soil and rock material between construction projects could be assessed. Optimal locations for crushing, sorting and storage of soil and rock could be found based on where material will be generated and needed in future construction. Even though it was not possible to quantify the GHG emissions from soil and rock management, the recycling of soil and rock has shown to reduce GHG emissions. It can be seen that from the few studies available, there are CO_2 savings to be made when reusing excavated soil and rock in construction projects. This is mainly due to fuel savings that are achieved when the number of transports from quarries and to landfills are reduced (Chittoori et al., 2012; Eras et al., 2013; Kenley and Harfield, 2011).

For C&D waste, there are studies assessing some flows at the urban level in order to estimate the recycling potential. Generally, the resolution of measured waste flows from construction and demolition is low (Blengini and Garbarino, 2010). It is often a mixture of materials and it is not clear if soil and rock are included in the C&D waste definitions. To get around the problem of uncertain data, some C&D waste studies estimates the composition of the building material stock by looking at what construction materials have been used historically (Hiete et al., 2011, Hsiao et al., 2002). By estimating the construction and demolition rate, the waste composition and quantities can be estimated. A similar approach could be used to estimate future generation and need of soil and rock materials in construction. Estimations of excavated soil and rock at the urban level could be based on estimations on the generation of soil and rock from different construction types together with information about future construction needs.

From this background, a model which makes it possible to estimate future generation of soil and rock and construction material need in urban areas is developed within the Optimass project (www.optimass.se).

For the assessment of artificial turf, the energy use and GHG emissions were analyzed by using an LCA approach. It was shown that the infill choice has large impact on the total GHG emissions and energy use. The infill materials analyzed were SBR (Styrene Butadiene Rubber) which is recycled tire rubber, TPE (Thermoplastic Elastomer) which is a new plastic material, EPDM which is a new rubber, and R-EPDM which is recycled rubber such as from cables and automotive mats. From these aspects, SBR and R-EPDM infills were beneficial compared to TPE and EPDM infills. This is because the production of infills from recycled materials used less energy and emitted less GHG emission than from production of materials from new resources.

For this study, the system boundary included the artificial turf and the material and processes that were required during its construction, maintenance and disposal. The system boundaries were chosen to represent the life cycle of an artificial turf surface. However, the system could be expanded in order to include sports facilities and even arenas which includes associated buildings, paved areas, grandstands etc. From such environmental assessment, it would be able to identify what aspects are important at a higher system level. These aspects can be different from what has been found in this thesis since many other materials and processes would be included.

The choices made on data regarding the production of the different plastic and rubber materials can give significant effect on the results. Since the exact ingredients of polymers and additives in the production processes are different for each product, assumptions hade to be made in order to use more general compositions for each infill type.

It can be seen that previous studies about environmental impact of infill materials for artificial turf have used another system boundary than in this thesis. In those studies, the production of infill materials was compared against each other. The results showed environmental benefits for recycled materials compared to new EPDM and TPE infills. However, such system boundaries, do not answer if the choice of infill materials is an important aspect in terms of environmental impact of artificial turf, which includes both production, maintenance and final removal. In this thesis, the whole life cycle of artificial turf was considered and it was found that the infill choice has large significance for the total energy use and GHG emissions. Other important aspects for energy use and GHG emissions are transportation of excavated soil and rock, use of quarry materials and incineration of rubber and plastic materials. The environmental levels of ESA described in chapter 1 can be useful to identify differences between assessments and how the significance for an environmental aspect changes with different system boundaries. Such approach is useful to minimize the risks that decisions in construction results in environmental sub optimizations.

The chemical analysis of infill materials for artificial turf showed that all infill materials could leach substances of potential human and environmental concern. The study points out that there are many similarities between the materials in terms of leaching, and that infill materials must be analyzed and compared on the same premises. Today, assessments of local environmental impact are in most cases only focused on the use of SBR infill even though the plastic and rubber materials share some leaching characteristics. Due to this and the fact that studies of actual drainage water are few, a project has been initiated for field sampling and chemical analysis of drainage water from artificial turf fields using different infill materials.

For the analysis of leached organic substances, a screening method was used to primarily detect what substances are present in the leachates. In this method, leachate water was analyzed for semi-volatile organic compounds by using a Gas Chromatography Mass Spectrometry (GC-MS) column instrument. The accuracy of identifying what organic substances have been detected in the leachate is lower than with target analysis. However, the screening method is useful to give an overview of what substances could be expected in the leachates. Using target analysis assumes that it is already known what substances are of interest. As a complement to the screening analysis, the presence of some target substances, such as PAH:S and phenols, were also analyzed. Further studies should benefit from combining a screening approach and a more accurate target analysis of certain types of substances. Such approach makes it possible to analyze and compare the materials on the same premises.

In this thesis, questions were raised about how the use of recycled granular materials of soil, rock and rubber in construction perform at different environmental levels. The use of construction materials can be studied from different environmental perspectives, from using narrow to wide system boundaries. The environmental assessment levels presented by Roth and Eklund (2003) were originally developed for assessing the use of by-products in road construction. However, in this thesis, they were applied for environmental assessments of using stocked materials. For the study of urban soil and rock management, it was found that environmental assessments with wide system boundaries are missing and that such assessments are needed in order to identify how environmental impact and resource use can be reduced. For the case of artificial turf, it could be seen that narrow system boundaries are dominating assessments of environmental impact and that wider system boundaries were useful to identify significant environmental aspects at a higher system level that previously have been neglected. Environmental assessments of using stocked materials should take into consideration the material applications´ significance for the environmental impacts at higher system levels. Broader system boundaries in environmental assessments will reduce the risk for sub-optimizations when taking decisions on how materials should be used in construction.

5. CONCLUSIONS

- It was possible to develop a model for urban soil and rock flows.
- It was not possible to quantify the urban soil and rock flows from the literature.
- Landfilling of excavated soil and rock can be up to 5.5 ton / capita and year and the use of quarry materials in construction can be up to 11 ton /capita and year.
- It was not possible to quantify GHG emissions from soil and rock management from the literature.
- The recycling of soil and rock can reduce transport-related GHG emissions.
- The soil and rock model can be used for quantifying resource efficiency and environmental impact of soil and rock management.
- Assessing resource use and environmental impact of soil and rock management can benefit from prognosis of the generation and need of these materials.
- The artificial turf infill materials of recycled rubber lead to lower energy use and GHG emissions than new materials of TPE and EPDM infills.
- The choice of infill material has significance for the total energy use and corresponding GHG emissions from artificial turf.
- The leaching of substances of environmental concern were detected from all infill materials.
- Assessments of local environmental impacts need to include all infill materials and the materials need to be assessed on the same premises.
- Environmental assessments of using stocked materials in construction can benefit of taking into consideration the material applications´ significance for the environmental impacts at a higher system level.
- Broader system boundaries in environmental assessments will reduce the risk for sub-optimizations when taking decisions on how materials should be used in construction.

6. FURTHER RESEARCH

- Assessing resource use and environmental impact of soil and rock management would benefit from prognosis of the generation and need of these materials. Further research for such model has started in the Optimass project.
- Assessments of local environmental impact need to include all infill materials and the materials need to be assessed on the same premises. A study of drainage water from artificial turf fields has therefore been started in a project with Stockholm Municipality.

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PAPER I

SUSTAINABLE MANAGEMENT OF EXCAVATED SOIL AND ROCK IN URBAN AREAS – A LITERATURE REVIEW

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SUSTAINABLE MANAGEMENT OF EXCAVATED SOIL AND ROCK IN URBAN AREAS – A LITERATURE REVIEW

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Abstract

Construction in urban areas implies use of construction materials from quarries and excavation of soil and rock. From a resource perspective, there could be benefits from using excavated soil and rock as a construction material. The aim of this paper is to describe the material flow and management practices of urban excavated soil and rock from the perspective of resource efficiency. A conceptual model for the urban flow of excavated soil and rock was developed and a literature review concerning the management of excavated soil and rock was conducted. The conceptual model was subsequently used to clarify the different perspectives of the scientific literature and knowledge gaps. Conclusions drawn are that there is little knowledge about the quantities and the fate of excavated soil and rock in urban areas. Current research is focusing on the waste flows of construction material and little is known about the overall management practices of excavated soil and rock. Clearly, excavated soil and rock are often disposed at landfills and the recycling rate for high quality purposes is low. There is a need to evaluate the potential for an increased use of excavated soil and rock as construction material. However, the overall efficiency of urban construction material management can only be evaluated and improved by also including construction materials produced in quarries.

Keywords: Excavate soil rock construction mass flow

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Abbreviations: C&D: Construction and Demolition

1 INTRODUCTION

The need for resource efficiency and reduction of climate impact from urban areas is crucial for a global sustainable development. Ongoing urbanization and growth of cities will likely lead to significant increases in the demand for natural resources such as water, land, energy, and mineral resources (Huang et al. 2010). Urban areas emits about 80% of the global CO2 emissions (Heinonen & Junnila 2011) and is responsible for about 80% of global energy consumption (Grubler et al. 2012). In rapid growing cities, the construction sector has shown to be one of the major sources to CO2 emissions (Weber et al. 2007). In order to reduce climate impact from construction, there is a need to improve resource efficiency and increase the reuse of construction materials (Blengini & Garbarino 2010; Eras et al. 2013; Gangolells et al. 2014; Huang & Hsu 2003; McEvoy et al. 2004; Miliutenko 2012; Simion et al. 2013; Toller et al. 2011).

The use of natural resources for construction in urban areas was described by Wolman (1965) as one of the components in the metabolism of cities. The metabolic requirements of a city was defined as all the material and commodities required to sustain the city's inhabitants, such as food, water, clothes, durable goods, electric energy, and construction material. A metabolic approach can be helpful to evaluate the sustainability in urban management of construction materials. A methodology for material flow analysis where material flows are structured and quantified, gives a better understanding of the metabolism (Huang & Hsu 2003; McEvoy et al. 2004). Such methodology has been used in other research fields such as for biomass, phosphorus and energy to describe sustainability, self-sufficiency and resource security (Chowdhury et al. 2014; Decker et al. 2000; Rosado et al. 2014, Welfle et al. 2014).

Construction of buildings and infrastructure require use of construction materials, earthwork, transportation and management of large volumes of materials such as aggregates and excavated soil and rock. This paper is mainly focusing on the metabolism of excavated soil and rock which consist of all materials generated when digging/blasting in the ground for urban construction purposes.

Depending on local geological conditions and anthropogenic activities, excavated material can be rock, stones, gravel, sand, clay, organic material and materials from previous constructions or industrial activities. The quantities of excavated soil and rock can be considerably big and hauling and handling costs high. In infrastructure projects, on-site handling and hauling of excavated soil and rock and construction material from quarries, i.e. quarry material, can be up to 30% of the total project cost and generate significant amounts of CO2 emissions. Optimization of the management in projects has large potential to reduce both costs and climate impact (Kenley & Harfield 2011). The management alternatives of excavated soil and rock vary between construction projects. Lafebre et al. (1998) and Eras et al. (2013) has described possible management alternatives for excavated soil and rock as 1) use on-site 2) use in other projects 3) pretreated before use in other projects 4) store for later use, 5) use as landfill cover or dispose at landfill.

Other parameters affecting management possibilities are geotechnical properties, geo environmental properties, availability of recycling facilities, landfills and quarry materials (Chong & Hermreck 2010; Wilburn & Goonan 1998).

It is important to notice that geotechnical properties are basis for what functions can be achieved. Particle size, density, water absorption, hydraulic conductivity, deformation properties and bearing capacity are some of the most important aspects that have to be considered. Also, geo environmental properties such as PH value, organic content, total concentration and leachate concentration set the conditions for what material can be acceptable at the project site (Arulrajah et al. 2013). There is an increasing awareness of the possibilities of reusing materials such as soil and rock for construction purposes. Laboratory tests and field studies proves that excavated soil and rock, brick, glass, concrete, asphalt and ceramics can be used beneficially for civil engineering purposes and hence replace quarry materials (Arulrajah et al., 2012; COWASTE 2014, 2014; Gabr & Cameron, 2012; Kreft-burman et al. 2013; Mohammadinia et al. 2013; Rahman et al. 2014; Taha & Nounu 2009).

This paper is one of the outcomes from the research project “Optimass”. The project aim for “Optimass” is to provide conditions for a more sustainable management of soil and rock in dense city regions. The idea in both the “Optimass” project and this paper is that there is a potential to reduce environmental and economic costs by coordinating the soil and rock material produced in quarries and material excavated due to construction. In this paper this is done in a regional context.

The aim of this paper is to describe the material flow and management practices of urban excavated soil and rock. Focus is on excavated soil and rock due to construction and resource efficiency. This study will look at all materials as potential resources regardless of the specific material properties. This is done even though the geotechnical and geo environmental conditions and hence environmental risks affects the potential use of excavated soil and rock. In this study the primary focus is to reveal the quantities of material flows in urban regions from a resource perspective.

The paper is based on a literature review. The aim is also to identify knowledge gaps and needs of future research. For this paper, the research questions are:

- What is the knowledge about the flow of urban excavated soil and rock?
- What are the benefits of using excavated soil and rock in construction?

2 METHODOLOGY

The methodology of this study consists of a literature review in the research field of soil, rock and sustainable management. The purpose of the review is to give a presentation of literature related to the research field and the different perspectives on soil and rock. A model illustrating the urban flow of building materials was developed and used to clarify different types of scopes and material flows described in the scientific literature. Information was collected by using key words for the types, names, processes and objects significant to the flow of excavated soil and rock in urban areas and the methods relevant for describing its sustainability. The results were analyzed and conclusions drawn are presented in this paper.

2.1 Conceptual model for construction material flows

A conceptual model for construction material flows was developed and is presented in Figure 1. The model can be applied at different spatial system levels, from project level to transnational level to illustrate how construction materials are managed. The model illustrates the demand, supply, stock and internal flows of construction materials for a system.

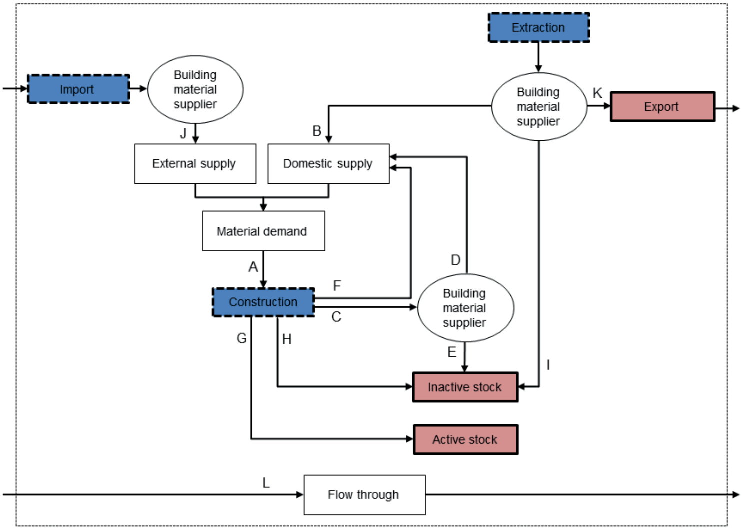

Figure 1. *Construction material flows and the demand and supply in construction.*

The demand for construction materials can be met by either domestic sources within the system or by imported sources which are illustrated with blue/dashed boxes. Sources can come from natural extraction such as production in quarries or residues from construction or industry. Construction materials are used domestically in the system or exported from the system and are illustrated with red/bold boxes. The construction material that is used domestically ends up as stocked material of two types: active stock and inactive stock. The active stock represents material that is typically used and accumulated in buildings, roads and other constructions while the inactive stock is material that has been permanently taken out of use and serves no purpose. The latter is usually construction and demolition (C&D) waste ending up at disposal sites (Johansson et al. 2013). In the considered system, there is also a flow through, i.e. external material passing through the system. The conceptual model was developed with inspiration from previous work on construction mineral flows by McEvoy et al. (2004), the work on biomass demand by Welfle et al. (2014), the work on urban material stocks by Johansson et al. (2013), the work on regional management of building materials by Frostell et al. (2009).

2.2 Applying the conceptual model to excavated soil and rock

In this paper, the flows in Figure 1 are referred to as flow *A* – *L*. Construction in Figure 1 was defined as all types of construction activities including demolition. Flow *A* refers to the demand and use of material in applications where excavated soil and rock can be used. Typical applications are in roads, e.g. as sub-base where crushed rock can be replaced (Arulrajah et al. 2012; Kreft-burman et al. 2013). Flow *B* corresponds to the supply of extracted material from domestic sources such as sand, gravel and rock that are usually produced in quarries. Flow *C* is the excess excavated soil and rock generated at the construction site which is sent away as waste to a building material supplier. The building material supplier processes the material before it is used in construction (flow *D)* or sends the material to landfill (flow *E*). Flow *F* is excavated soil and rock generated at the construction site which is either needed on-site for use in construction or needed in other construction projects, as described by Eras et al. (2013). The use of flow *F* materials is often preceded by some kind of treatment, for example crushing and sorting.

With exception for exported materials, excavated soil and rock is accumulated in the material stock. Flow *G* refers to all excavated soil and rock that is used in construction and hence accumulated in the active stock.

Flow *H* refers to excess excavated soil and rock which is sent to landfills or other sites where it is inactively stocked. At closing landfills and quarries, it is used as cover material. Such management practice is in the literature regarded as recycling. However, landfill covering has been described as recycling of very low quality (Blengini & Garbarino 2010). In addition, some argue that landfill cover material is a potential resource that could be extracted and used for other purposes (Frändegård et al. 2013). In the model, material ending up in landfills is therefore regarded as disposed material accumulated in the inactive stock. Flow *I* refer to fractions of extracted quarry materials that are not used, such as rock flour. The Flow *J*, *K* and *L* respectively refers to the import, export and flow through of excavated rock and soil in the considered system, and is similar to the flow of construction minerals presented by McEvoy et al. (2004).

3 QUANTIFYING EXCAVATED SOIL AND ROCK – EXAMPLES OF STUDIES AND METHODS

3.1 Material flow analysis of construction materials

Material flow analysis is a commonly used approach for describing the metabolism and regional flows of construction materials. It was used by McEvoy et al. (2004) to study the flow of rock, sand, gravel and other aggregates in the North West region of England and by Huang & Hsu (2003) to study regional flows of construction materials in Taipei with focus on sand, gravel, concrete, asphalt and construction waste (flow *A*, *C* and *H*). Regional material flows of construction materials and waste (Flow *A* and *H*) in 25 mega cities has also been studied by Decker et al. (2000). Further, Rosado et al. (2014) studied the flow of non-metallic minerals in Lisbon Metropolitan Area which was primarily stone, cement and sand (Flow *A*, *C, G* and *H*). Some results from material flow studies are presented in section 3.3.

The given examples of material flow studies used data both from measurements and estimations. McEvoy et al. (2004) collected statistics from authorities and business organizations. However, due to lack of data, some estimations had to be made. The quantities of excavated soil and rock were not studied specifically. However, the share of excavated soil and rock that is managed as waste (Flow *C* and *H*) was included in available data for construction mineral waste. Waste and recycling quantities was gathered from environmental agencies. Also data on crushed rock and soil used in construction and data on aggregates and soil used for covering purposes at landfills was available.

Huang & Hsu (2003) used estimations of construction material demand combined with waste statistics. Construction material demand (Flow *A*) was calculated by using building and infrastructure production data from regional and national authorities. Construction waste (Flow *C* and *H*) data was taken from previous waste generation estimations. Waste of excavated soil and rock was not calculated separately from other construction wastes.

Rosado et al. (2014) based calculations on consumption statistics for goods and services and the use of construction materials (Flow *A and G*) such as non-metallic minerals, sand, cement, clay and stone. Construction waste (Flow *C* and *H*) was quantified from data on total C&D waste flow.

The material flow studies presented are focusing on the metabolism at a city or regional level with a strong focus on the input and output of materials. The internal flows i.e. how materials are moving inside the considered system has not been studied. Most studies focus on the presented waste situation and status while Rosado et al. (2014) forecasts the future regional development in order to describe future challenges for material reuse.

3.2 C&D waste studies

The flow of excavated soil and rock has in some cases been described in C&D waste studies. However, available data on the generation of C&D waste is generally uncertain. This is mainly due to illegal dumping activities and lack of proper measurements of C&D waste facilities (Simion et al. 2013). Further, uncertainties are even larger due to the fact that the definition of C&D waste varies. Simion et al. (2013) and Coronado et al. (2011) defines C&D waste to be concrete, asphalt, wood, metal, drywall, paper and plastics while Hao et al. (2007) includes all surplus materials from construction. Further, Blengini and Garbarino (2010) include and Hiete et al. (2011) exclude excavated soil and rock from the C&D waste definition.

Different methods have been used to quantify C&D waste, some were based on the waste amounts processed at recycling facilities (Flow *C*) while other used estimations on generated waste (Flow *C* and *H*) from a certain amount of construction work. Hsiao et al. (2002) based C&D waste estimations on the number of square meters permitted for C&D in the region together with estimations on material need.

The environmental and economic benefits of C&D waste recycling (Flow *C* and *H*) within a region has been estimated in a few studies. Blengini and Garbarino (2010) evaluated the energy and climate performance of the C&D waste recycling chain in an Italian province. The flow of C&D waste including waste of excavated soil and rock was quantified by collection of data from recycling facilities (Flow *C*). Hiete et al. (2011) studied the federal state Baden–Württemberg in Germany and the generation of C&D waste. The amounts of waste of excavated soil and rock was recorded. Estimations on C&D waste (Flow *C* and *H*) generation for construction activities was combined with data on accumulated materials in buildings and construction, refurbishment and demolition rates.

3.3 Presentation of data from previous studies

A presentation of some data from a selection of studies is found in Table 1. In most studies, there have been no quantification of separate waste fractions. Instead, the total amounts of C&D waste have been used. Bold data represents pure quantities of waste of excavated soil and rock.

Table 1. *Generation and recycling of C&D waste and waste of excavated soil and rock, and the use of construction materials.*

	Capita	Generation of C&D waste including waste of excavated soil and rock		Recycling rate	Construction material use or material extracted in quarries	
Region or country	[10^6]	[Mton]	[ton/capita]	[%]	[Mton]	[ton/capita]
Taipei[a]	6,2	40,1	6,47	“minimal”	9,41	1,52
English region[b]	6,9	10,2	0,68	31,7%	31,4	4,55
Italy[c]	n/a	n/a	0,8	n/a	n/a	6-11
Italian region[c]	n/a	n/a	n/a	23%	n/a	n/a
German region[d]	11	**18,9**	**1,72**	**8,5%**	87,4	7,95
German region[d]	0,77	0,77	1	n/a	n/a	n/a
Finland[e]	5,46	**20-30**	**3,7-5,5**	n/a	n/a	n/a
Lisbon region[f]	3	1,25	0,42	n/a	19,5	6,51
European Union[g]	n/a	n/a	**0,10-0,32**	n/a	n/a	4,8
Chinese region[h]	8,46	6,0	0,71	n/a	n/a	n/a

[a] Data gathered from *Materials flow analysis and emergy evaluation of Taipei's urban construction* (Huang & Hsu 2003). Construction material use refers to the use of sand and gravel.

[b] Data gathered from *Managing the Flow of Construction Minerals in the North West Region of England - A Mass Balance Approach* (McEvoy et al. 2004). Recycling rate refers to reuse in roads and planings. Reuse in landfills (1,18 Mton) is here excluded. Construction material use refers to rock, sand and gravel.

[c] Data gathered from *Resources and waste management in Turin (Italy): the role of recycled aggregates in the sustainable supply mix* (Blengini & Garbarino 2010). Recycling rate refers to use in concrete, road, harbour and airport construction.

[d] Data gathered from *Matching construction and demolition waste supply to recycling demand: a regional management chain model* (Hiete et al. 2011). C&D waste only refers to excavated materials. Recycling rate refers to highest possible rate of recycling and has been calculated based on the fact that 79% of the 18,9 Mton i.e. 14,9 Mton excavated material was processed. The production of sand, gravel, cobbles and excavated material from recycling facilities was totally 1,6 Mton. Assuming that these 1,6 Mton of produced material origins from excavated soil and rock, the recycling rate is calculated to 1,6 Mton/ 18,9 Mton = 8,5%. Construction material use refers to the production of minerals in quarries in the studied area.

[e] Data gathered from *Experiences of utilizing mass stabilised low-quality soils for infrastructure construction in the capital region of Finland – Case Absoils Project* (Forsman et al. 2013), which has estimated the generation of excavated soil to 20-30 Mton. A value per capita was calculated by using population data gathered from Finnish public authority for statistics (Statistics Finland 2014).

[f] Data gathered from *A Material Flow Accounting Case Study of the Lisbon Metropolitan Area using the Urban Metabolism Analyst Model* (Rosado et al. 2014). Construction material use refers to the use of non-metallic minerals.

[g] Data gathered from *Comparing environmental impacts of natural inert and recycled construction and demolition waste processing using LCA* (Simion et al. 2013). Generation of C&D waste refers to excavated soil only.

[h] Data gathered from *A SWOT analysis of successful construction waste management* (Yuan 2013).

The scopes of presented studies in Table 1 are not identical and there are differences regarding the system boundaries and types of materials quantified. The majority of the studies make no quantification of the waste fractions which means that the data resolution is low. Further, it is important to stress that the waste quantities presented also include other types of construction waste than excavated soil and rock. From the example of studies in Table 1, the generation of C&D waste including waste of excavated soil and rock varies with a factor of 10. From the studies reporting specific quantities of waste of excavated soil (Flow *C* and *F*), it can be seen that the generation varies even more.

The literature gives some examples of recycling rates of waste from construction and excavated soil and rock. Blengini & Garbarino (2010) described the waste management in Torino. About 58% of the C&D waste was directly landfilled (Flow *H*). The other 42% were sent to recycling facility (Flow *C*). About 55% of the produced recycled aggregates, i.e. 23% of total C&D waste was used in high quality and medium quality construction purposes such as construction of concrete, road, harbor or airports (Flow *D*). This means that 77% was ending up in landfills and depleted quarries as waste or as covering materials and for rehabilitation purposes (Flow *E*). Another example is the management in the Federal state of Baden-Württemberg in Germany. Hiete et al. (2011) describe that about 79% of excavated waste and 91% of C&D waste was processed in recycling plants (Flow *C*). The recycling plant produced 4,4 Mton of which 2 Mton was processed concrete and 1,6 Mton was sand, gravel, cobbles and excavation materials. By assuming that all 1,6 Mton is excavated soil and rock, the maximum possible recycling rate (Flow *D*) can be estimated. The assumption implies that the use of excavated soil and rock in construction of roads and infrastructure could not be more than 8,5% of the total amount entering the recycling facilities while the rest was sent to landfill (Flow *E*). McEvoy et al. (2004) describe the recycling rate of construction waste in Northwest England where about 31,7% of C&D waste was recycled and used in construction (Flow *D*). Also, 11,5% of C&D waste was used at landfills as cover.

The demand of construction materials for construction purposes has been estimated in some of the studies presented in Table 1. The definition of construction materials varies but common for all is that the material has primarily been extracted from quarries. Also there is a difference between material demand and use of quarry materials, since some of the material that is generated due to construction can also be reused, such as excavated soil and rock that has never been accounted as a recycled/used material (Flow *F*). In the examples of studies presented in Table 1, the use of quarry materials (Flow *B*) is between 1,5 – 11 tons per capita.

4 IDENTIFYING ENVIRONMENTAL AND ECONOMIC BENEFITS OF REUSING EXCAVATED SOIL AND ROCK

4.1 Reusing excavated soil and rock on-site

Several studies describe the environmental gains with reusing excavated soil and rock (Flow *F*) at the construction site (Chittoori et al. 2012; Eras et al. 2013; Kenley & Harfield 2011; Lafebre et al. 1998). Eras et al. (2013) showed that by planning for mass balance of earthworks in an industrial construction project, it was possible to relocate and reuse 44% of the excavated materials, i.e. about 700 000 m3, and hence reduce earthwork and transports to landfill as well as the production and use of quarry materials. The total climate impact from transports could in this example be reduced with about 4000 tons of CO2 from fuel savings. Further, costs could be reduced with 1,76 million dollars. Similar results has been presented by Chittoori et al. (2012) who described the cost and environmental benefits of reusing excavated soil within a pipeline construction project. The increased reuse reduced the material management costs and climate impact by 85%.

Stabilization is a technology for improving geotechnical properties in terms of increased strength, reduced permeability and compressibility of soil. The technology makes it possible to use low quality materials in construction, such as soft soils that are usually landfilled (Makusa 2012). Chemical stabilization means that cement or other binder materials such as fly ash and lime is mixed with soil which leads to chemical reactions that stabilizes the soil. Forsman et al. (2013) described the experiences in the Absoils project of stabilizing excess soft soils that are usually excavated, transported to landfill and replaced with natural aggregates. The environmental and economic impact of soft soil stabilization for construction purposes compared to the use of new construction materials was not assessed.

In order to reuse excavated soil and rock on-site, there is a need for space at the construction site. In dense city regions, the availability for space at the construction sites is limited and the possibilities for on-site sorting of C&D waste including excavated soil and rock are often low. This has been described by Hao et al. (2007) regarding the situation for waste management in Hong Kong.

4.2 Reusing excavated soil and rock in other projects

Reuse of excavated soil and rock directly in other projects (Flow *F*) means that materials are transported between construction sites. Such reuse is possible when there are several construction projects going on in the same region. In Helsinki, landfills are starting to get full, due to the disposal of excavated soil and rock. This in turn leads to increased transportation to landfills further out from the city center, about 50 km. The coordination of excavated soil and rock between construction projects has been one of the aims of the Absoils project. However, no data on environmental or economic benefits has been published (Forsman et al. 2013). The benefits of using excavated soil and rock in other projects have been studied by the English non-profit organization CL:AIRE. They conducted a study of a cluster project which consisted of four remediation projects located relatively close to each other in Northwest England (CL:AIRE 2013). In these projects, large amounts of contaminated soil were excavated and transported to a temporary hub located at one of the construction sites where the materials were treated and thereafter transported to construction sites for reuse. The cluster approach resulted in an increased reuse of totally 30 000 m3 of excavated material and emission reductions of about 100 tons of CO_2. Furthermore, transportation, landfilling, and use of new construction material were reduced. The cost savings was estimated to 30% (CL:AIRE 2013).

In order to coordinate and exchange excavated soil and rock between construction sites, there is a need for joint planning. At an early stage of the planning process, it becomes important to evaluate the coordination benefits for all projects involved. The quantities and quality of excavated soil and rock over time would here be essential information.

4.3 Recycling at facility

Excavated soil and rock being classified as waste can be transported to a recycling facility (Flow *C*) where it is treated and prepared for use in other construction projects (Flow *D*). The environmental potential for recycling excavated soil and rock in such way has been studied by few. For example, Blengini and Garbarino (2010) concluded that there are environmental benefits when using recycled C&D waste including excavated soil and rock produced at a recycling facility compared to use of quarry materials. For 13 out of 14 environmental aspects studied, there were environmental gains. The CO_2 emissions were reduced with about 14 kg CO_2 equivalents per ton when recycling C&D waste compared to using quarry materials. Here, production of materials is included, not just transports. Transport distances were between 15 and 25 km for recycled aggregates and twice as high for natural aggregates. Even though transport distances are crucial for the environmental benefits, they could increase by a factor of 2 or 3 and still have a positive environmental effect. The work was developed in the EU project SARMa with the objective to achieve a common approach to sustainable management of aggregate resources (Blengini & Garbarino 2011). Simion et al. (2013) studied the climate effects of producing natural aggregates compared to recycling C&D waste. Climate impact from natural aggregate production was about 103 kg per ton, compared to about 16 kg per ton for recycled C&D waste.

There are examples of similar studies that focus on the economic potential for recycling other C&D waste flows than excavated soil and rock. For example, Coelho and de Brito (2013) assessed the economic potential and localization of a recycling facility in Portugal for C&D waste with a plant capacity of 350 tons per hour. Excavated soil and rock was not accepted and recycled at the facility. It was concluded that such investment is viable. The main income comes from input material gate fees at the facility. The main cost is the management of the fractions of C&D waste that can’t be recycled at the facility. This waste has to be transported to other treatment facilities or landfills where a fee must be paid. Different conclusions were drawn by Zhao et al. (2010) who evaluated the economic potential for a fixed recycling facility with a capacity of 100 tons per hour and a mobile recycling facility with a capacity of 50 tons per hour in the city of Chongqing in China. One conclusion was that such investment is risky due to low costs for the landfilling of C&D waste and the use of quarry materials in combination with high investment costs for the recycling facility. Galan et al. (2013) optimized localizations for waste recycling facilities due to transportation distances and total costs in the region of Cantabria in Spain. Recycling facilities with a capacity of 50 000 – 300 000 tons was evaluated. Assuming 8 hours of operation, the recycling capacity corresponds to 17 – 103 tons per hour. Cantabria has about 600 000 inhabitants and no recycling facilities are in place. All material is currently landfilled. An optimal combination of recycling facilities, according to Galan et al. (2013), was three processing plants or one processing plant combined with three transfer stations. In the suggested scenarios, transport distances for C&D waste could be reduced by 35% (Galan et al. 2013).

Robinson and Kapo (2004) analyzed the suitability for recycling facilities in Maryland and Virginia of United States and Yuan (2013) studied the strengths and weaknesses for improved construction waste management in Hong Kong, China. One of the conclusions drawn was that the selection of appropriate locations for recycling facilities is critical for its economic viability, and to reduce total transports and environmental impact.

5 ANALYSIS

The known quantities of excavated soil and rock, construction material demand and use are presented in Figure 2.

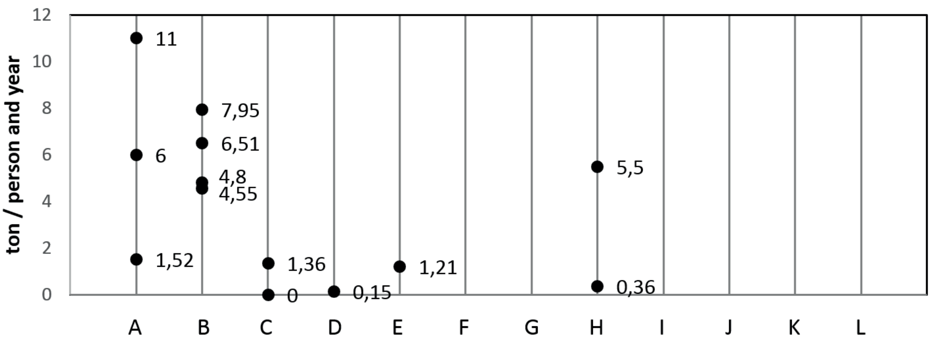

Figure 2. *Known quantities of excavated soil and rock, construction material demand and construction material use.*

Flow *A* corresponds to the demand and use of construction materials such as soil, sand, gravel and rock. From the review, it can be found that there is a lack of knowledge about what materials and how much is used in construction. The data presented for flow *A* are estimations on demand and use of construction minerals. Unfortunately, available data excludes excavated soil and rock. The use of construction minerals such as sand, gravel and rock, varies heavily, up to 7 times. This can partly be explained by fluctuations in economy, which impacts on the construction intensity (Huang & Hsu 2003). Even though data in flow *A* does not cover the total demand of excavated soil and rock, it is still valuable since it gives an idea of how much material is used.

The data quality for the use of construction minerals such as sand, gravel and rock (Flow *B*) is relatively high since it is based on real production data from the construction industry and since there are less differences in the definitions of this flow. Data on flow *B* have been presented in several studies.

Of the reviewed literature, there are only two studies that estimate the quantity of excavated soil and rock that is being sent to recycling facilities (Flow *C*). The studies have been conducted on two urban regions where there are significant differences between waste management practices for excavated soil and rock. In one of the regions, excavated soil and rock is sent to recycling facilities while such practices are minimal in the other region. Instead, excavated soil and rock is landfilled directly. The minimal flow (*C*) is illustrated in Figure 2 with a zero value. The knowledge base about flow *D* and *E* is even weaker since only one example of quantification was found.

There is a major lack of knowledge regarding excavated soil and rock used within the project or sent to other projects (Flow *F*) and this management is hardly ever described in literature. It could be discussed if such knowledge lack is due to a general attitude that Flow *F* is relatively insignificant in terms of quantities, resource demand, environmental impact and waste generation. It can also be explained by the fact that Flow *F* is an internal flow which is usually not considered in urban metabolism. In addition, Flow *F* is usually not recorded in any official waste statistics since it is not sent to recycling facilities or landfills but transported directly between construction projects.

The use of excavated soil and rock in construction is a fraction of Flow *A* and is accumulated in the active stock (Flow *G*). Since no study estimates the complete Flow *A* there is thus no estimation of the complete active stock.

The direct landfilling (Flow *H*) and the use of excavated soil and rock as cover material at landfills seems to be a common management approach. The European strategy for landfills is focused on the closing of landfills (European Environment Agency 2009). In Europe, the need for cover material at closing landfills is time-limited and there is a growing need for construction materials in urban regions. From this situation, it could be questioned if the accumulation of excavated soil and rock at landfills due to covering purposes should be labelled as recycling.

None of the flows *I* - *L* are estimated in the reviewed literature and it could be discussed how significant these flows are in the context of excavated soil and rock. Flow *I* usually consist of rock flour which is excess material from extraction in quarries. This flow has little significance to the flow of excavated soil and rock and is often disposed. Flow *J* and *K* can consist of both excavated soil and rock. At the local level, it is reasonable to assume that excavated soil and rock is imported and exported, i.e. shared with other construction sites. From a regional perspective, it is reasonable to assume that these flows primarily consist of quarry materials since they have a higher market value and higher margins and thus are able to carry higher transportation costs.

The flow through of excavated soil and rock (Flow *L*) could be significant when the model is applied at a small scale. At the project level or local level, it could be used to describe the effectiveness of construction material management such as logistics and transportation.

The consequence of the general lack regarding knowledge of quantities and management of excavated soil and rock is the almost impossible task to estimate resource efficiency and the potential for improvements. Since official statistics usually not record the flows of excavated soil and rock, it can be beneficial to make qualified estimations. Key estimations would be the generation of excavated soil and rock and the demand for rock, gravel, and sand caused by construction activities.

Illustrating and quantifying the flows of construction materials can facilitate a discussion on not only construction materials and waste but also on the metabolism of these materials, with a larger focus on how they are managed and the resource efficiency in the system. Such discussions are important for developing cities where construction activity is high and where there is a lack of sites for material reuse and limited landfill capacity.

The environmental benefits for using excavated soil and rock in construction have been evaluated from different handling perspectives. The climate impact from extracting, processing and handling quarry materials has been estimated to 7,8 kg and 10,3 kg of CO2 per ton material, respectively (Simion et al. 2013; Zuo et al. 2013). In Figure 3, estimations are presented on CO2 savings when reusing excavated soil and rock on-site, in other projects or by preparing the material at recycling facilities.

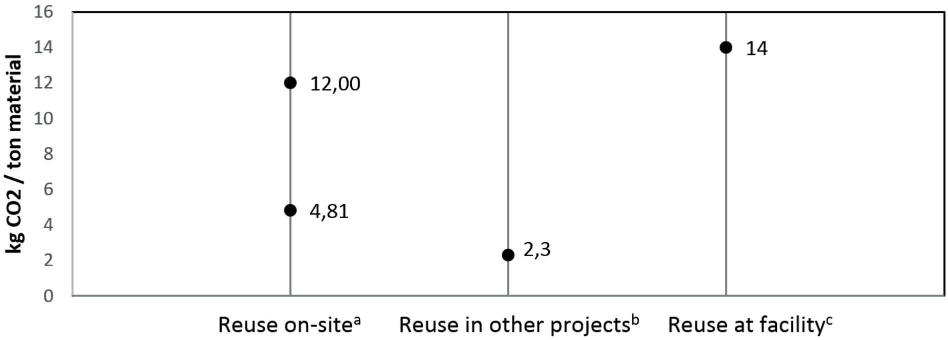

Figure 3. *CO2 savings due to excavated soil and rock reuse on-site, in other projects and at facility.*

[a] Data gathered from *Improving the environmental performance of an earthwork project using cleaner production strategies* (Eras et al. 2013) *and* from *Sustainable Reutilization of Excavated Trench Material* (Chittoori et al. 2012). Assuming a density of 1,6 tons per m3.

[b] Data gathered from *Remediation of Four Sites in Northwest England: A Successfully Completed Multi-Site, Multi-Consultant Cluster Project* (CL:AIRE 2013). Assuming a density of 1,6 tons per m3.

[c] Data gathered from *Resources and waste management in Turin (Italy): the role of recycled aggregates in the sustainable supply mix* (Blengini & Garbarino 2010).

CO2 savings can be achieved with all reusing strategies and can be significant. Increasing reuse on-site and hence reducing fuel consumption gives significant CO2 savings (Chittoori et al. 2012; Eras et al. 2013). The reuse of excavated soil and rock in other projects may also result in CO2 savings due to reduced transportation need to disposal sites and quarries. In these cases, the CO2 impact from using new quarry material and from disposal has not been included, so the saving potential is probably underestimated. The study by Blengini and Garbarino (2010) is the only example where the environmental impacts of recycling of excavated soil and rock is assessed from a complete life cycle perspective. The CO2 savings are mainly achieved by less landfilling of materials. Environmental data are scarce for all types of reuse. The largest knowledge gap seems to be for the environmental benefits of reuse in other projects. Coordination of construction projects and management of excavated soil and rock gave a lower climate impact, less transportation, less material landfilled and less need of quarry materials (CL:AIRE 2013). However, there is a need for scientific evidence and further research to assess the environmental potential of reuse. The coordination of excavated soil and rock between projects would require early planning. The knowledge about future resource needs and the quantities and qualities of excavated soil and rock generated in projects is a basis for such planning.

6 CONCLUSIONS

This study concludes that a resource perspective on excavated soil and rock in urban areas is missing. A waste perspective is prominent in the scientific literature. Main focus is on recycling potential and its environmental benefits. In this paper we show that it is possible to apply a resource perspective to describe the flow of excavated soil and rock. We identify 8 potential significant flows of excavated soil and rock in urban regions. The scientific literature deals only with a few of these.

This study concludes that it was not possible to reveal the quantities of excavated soil and rock in urban regions from the scientific literature. General management of excavated soil and rock are landfilling but can also be recycling at facilities, use on-site in construction or use directly in other construction projects. A few quantifications have been made, and they show that landfilling of excavated soil and rock is in the range of 0,4 to 5,5 tons per capita and year. The use of quarry materials ranged from 4,6 to 8,0 tons per capita and year. However, the flows are all largely unexplored and more research is needed.

The reuse of excavated soil and rock in construction can reduce costs and climate impact since transportation, landfilling and use of quarry materials are reduced. The few studies available indicate that saving potentials for reusing excavated soil and rock are up to 14 kg CO2 per ton. For a single construction project, reusing excavated soil and rock can reduce the material handling costs with 85%. However, more research is needed to clarify the environmental and economic benefits. It is concluded that the regional management of construction materials and excavated soil and rock could benefit from coordination of construction projects. This will need strategic planning at an early stage where the future demand and availability of construction material is assessed. Decisions on all levels will be needed, from construction project level for increasing reuse on-site to regional authority level for improving the conditions for reuse, such as establishments of hubs where material can be stored and sorted for later use in construction.

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PAPER II

ANALYSIS OF ENERGY USE, EMISSIONS OF GREENHOUSE GASES AND PRIORITIZED SUBSTANCES FOR ARTIFICIAL TURF AND ITS CONSTRUCTION MATERIALS

Submitted to Resources, Conservation and Recycling

ANALYSIS OF ENERGY USE, EMISSIONS OF GREENHOUSE GASES AND PRIORITIZED SUBSTANCES FOR ARTIFICIAL TURF AND ITS CONSTRUCTION MATERIALS

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Abstract

The environmental aspects of artificial turf for football purposes both involve local factors such as the exposure of chemicals to water resources and global aspects such as energy use and GHG emissions. This study assessed the energy use and GHG emissions for an artificial turf field with different types of infill materials. A life cycle analysis approach was used to identify significant posts for energy and greenhouse gas emissions for activities and construction materials throughout construction, use and removal of an artificial turf field. A chemical analysis of four types of infill typically used in Sweden was conducted in order to describe leachability of metals and organic substances. The infill types studied was SBR (recycled rubber from used car tires), TPE (new plastic material of thermoplastic elastomers), EPDM (new rubber of ethylene propylene diene monomer) and R-EPDM (Recycled rubber from used EPDM rubber products such as cables and automotive mats). It can be concluded that climate impact and energy use of an artificial turf field is significantly correlated to the choice of materials and maintenance but also to management when the turf is removed. The use of infill of recycled material, the reuse of soil and rock on site and the reuse of removed turf and infill could reduce climate impact and energy use. In this context, the use of machines for maintenance purposes is less significant. The differences in energy use and GHG emissions between infill materials are large, with highest impacts from TPE, followed by EPDM and lower impact from SBR and R-EPDM infills. Leachates from the recycled materials SBR and R-EPDM contained detectable concentrations of metal (zinc), where the zinc leaching from R-EPDM was relatively high. Organic substances which are known to be harmful for the aquatic environment and / or humans was found in detectable amounts in all infill leachates. The total concentration of these harmful substances in the leachates was highest from R-EPDM, followed by EPDM and lowest from SBR and TPE, the latter concentrations were at the same level. The results stress the need to include all infill types in risk assessments independent on their origin. There is an uneven distribution of risk assessments focusing on solely the recycled material SBR. For other infill types like TPE and EPDM the assessments are less extensive, and for R-EPDM no studies was found. Even though environmental risk assessments mostly concludes that the risks of artificial turf are small, the leachates produced in laboratory may not reflect leachates in the real world. It is therefore suggested that controlled measurements of actual field drainages are observed for further risk assessments.

Keywords: Artificial turf; infill material; crumb rubber; environment; health; life cycle assessment

Abbreviations: LCA, Life Cycle Assessment; SBR, Styrene Butadiene Rubber; TPE, Thermoplastic Elastomer; EPDM, Ethylene Propylene Diene Monomer; R-EPDM, Recycled EPDM; GC-MS, Gas Chromatography Mass Spectrometry; PVC, Polyvinylchloride; GHG, Greenhouse gas; MDI, Methylene diphenyl diisocyanate; SEBS, Styrene Ethylene Butylene Styrene; DOC, dissolved organic carbon

1. INTRODUCTION

1.1. Background and problem definition

Sport surfaces of artificial turf is used by both professional athletes and for spontaneous playing. The construction of artificial turf fields has greatly increased in Sweden and Norway (Swedish Football Association, 2015; Football Association of Norway, 2015) as it produces significantly more user hours than natural grass fields due to its durability and capacity in cold climate.

The design of an artificial turf pitch consist of several layers of materials. An illustration, based on Simpson et al. (2012) is given in Figure 1. The upper layer consist of a synthetic carpet where synthetic fibers are attached to a perforated backing of textile and latex. A layer of fine sand followed by a layer of elastic infill fills up the space between the synthetic fibers. Different types of infill are available on the market such as infill of recycled tires (SBR), new materials of thermoplastic elastomers (TPE) and Ethylene Propylene Diene Monomer (EPDM) or recycled rubber originating from products such as cables and automotive carpets, so called R-EPDM. A shock absorbing layer of a permeable elastic compound are sometimes used under the synthetic carpet. As subbase, an upper layer of fine sand is followed by a layer of crushed rock upon a drainage system.

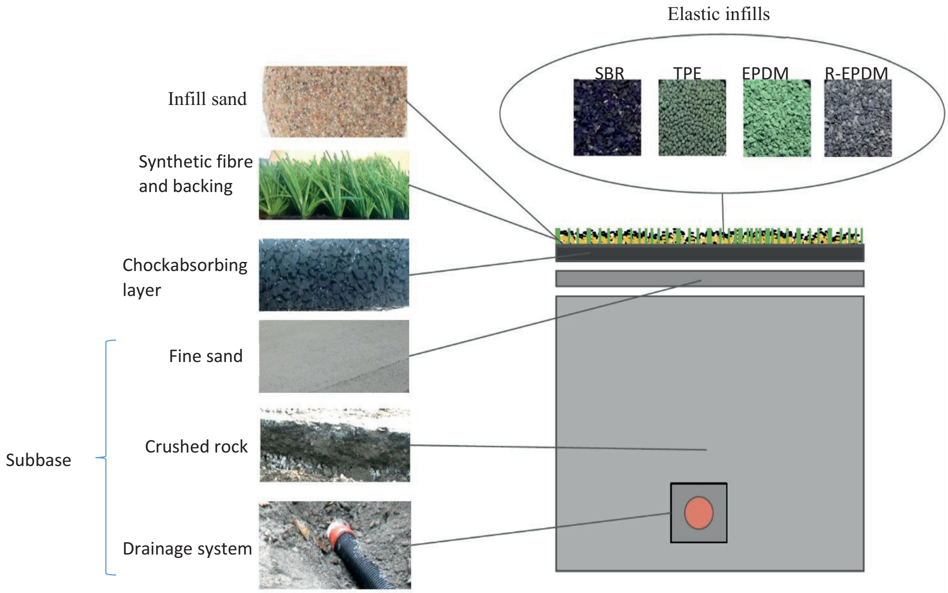

Figure 1. *Construction of artificial turf. Pictures produced by the article authors. Illustration based on Simpson et al. (2012).*

The environmental aspects of artificial turf materials has been described in a number of studies. Primarily focus have been on local environmental risks due to leaching to water and health risks (Birkholz et al., 2003; Lim & Walker, 2009; Moretto, 2007; Nilsson et al., 2008; Pavilonis et al., 2014; Ruffino et al., 2013; Verschoor, 2007; Vidair et al., 2007; USEPA, 2009) primarily due to the common use of infill made of used tire material which have shown to contain organic substances of priority such as polyaromatic hydrocarbons (PAH:s), phenols and metals (Plesser

& Lund, 2004). However, according to the conclusions and knowledge presented in environmental and health risk assessments and other studies in the field, the health risks are minimal (Birkholz et al., 2003; Ginsberg et al., 2011; Kim et al., 2012; Lim & Walker, 2009; Menichini et al., 2011; Moretto, 2007; Nilsson et al., 2008; Norwegian Institute of Public Health and the Radium Hospital, 2006; Pavilonis et al., 2014; Ruffino et al., 2013; Vidair et al., 2007). Further, the environmental risk from potential leaching of substances from tire infill to water has been concluded to be generally small (Moretto, 2007; Nilsson et al., 2008; Vidair et al., 2007; Birkholz et al., 2003), however there is a need to better understand the release and behavior of both metals and organic substances under relevant environmental conditions. This is also a question of importance for TPE and EPDM infill. Krüger et al. (2012; 2013) studied the potential leaching of some PAH: s and zinc and the ecotoxicology of infill materials and complete artificial turf systems. It was concluded that zinc in leachates was higher from SBR and EPDM compared to TPE while the PAH leaching was at same levels for all infill types. In addition, the ecotoxicology tests on leachates from column tests of complete artificial turf systems with the different types of infill all indicated a potential ecotoxicity (Krüger et al., 2013).

Other environmental aspects such as energy use and climate impact from a life cycle perspective has hardly been described in impact assessments of artificial turf. Life Cycle Assessment (LCA) is an established method that can be used to compare the environmental performance from a products or a services life cycle, from raw material extraction and production, use phase to final disposal (ILCD, 2012). A few studies with a LCA approach are available, and concluded that artificial turf are favorable to natural grass in terms of energy use, greenhouse gas emissions and water consumption in relation to user capacity, i.e. user hours (Uhlman et al., 2010; Holmström, 2013). Other LCA studies starting from the perspective of material recycling have focused on resource use, greenhouse gas emissions, energy use and water consumption of using SBR infill compared to EPDM and TPE infill and concluded that SBR was favorable to EDPM or TPE (Skenhall et al., 2012; Clauzade et al., 2010; Fiksel et al., 2011). Still, a life cycle perspective including global environmental impact from the construction, maintenance and final removal of artificial turf are missing. In addition, studies where leachates of inorganic and organic substances from different infill types are compared is missing.

1.2. Aim and objectives

This study aims to fill the gap of knowledge concerning life cycle impacts related to the construction, maintenance and removal of artificial turf. The study aims at describing the differences between infill materials in respect to environmental life cycle impact and the release of substances of environmental concern.

The objective is to quantify the energy use, greenhouse gas emissions in the life cycle of artificial turf and for the different infill materials types SBR, TPE, EPDM and R-EPDM.

Other objectives are to compare results from previous environmental and health risk assessments and analyze the leachability of primarily metals and organic substances from SBR, TPE, EPDM and R-EPDM.

2. MATERIALS AND METHODS

An analysis of energy use and greenhouse gas emissions for an artificial turf pitch was conducted by using an LCA approach based on the LCA guidelines presented in the ILCD handbook (2012). The system analyzed is presented in Figure 2. The energy use covered the energy consumed in raw materials acquisition, production, use and final removal and disposal. Energy use also included primary energy, i.e. use of energy resources. A certain focus was given to the different infill types due to its potential content of hazardous substances. These substances can be metals and organic substances, and can be emitted to water due to leaching. These substances are further referred to as prioritized substances. The infill types studied was SBR, TPE, EPDM and R-EPDM and has been chosen since they are used in Scandinavia (Nilsson et al., 2008).

A literature review of risk assessment on infill materials was conducted. Further, these infill material types were analyzed for prioritized substances. Here, each infill type underwent a leaching stress test where infill materials were analyzed for leachability to water of primarily metals and volatile organic substances.

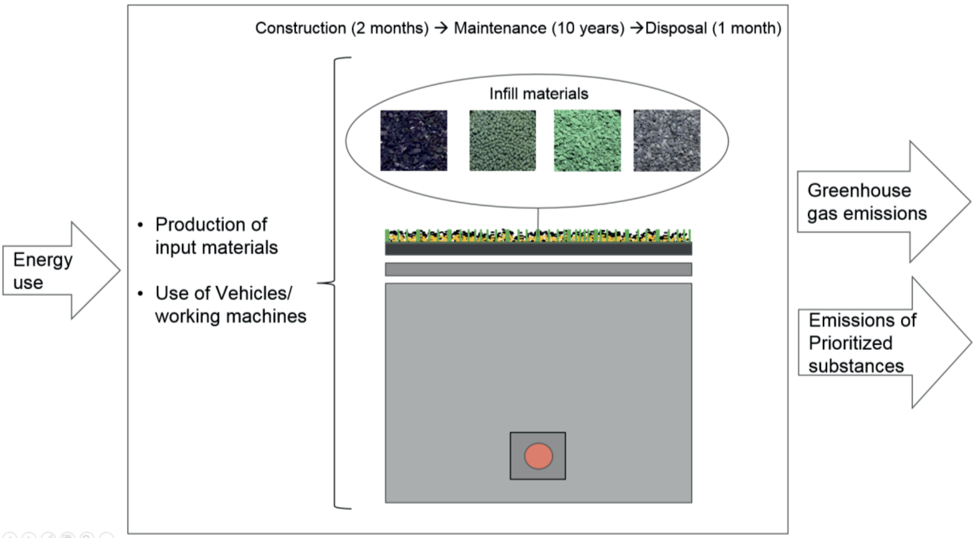

Figure 2. *System boundaries for analysis of energy use and emissions och greenhouse gases and prioritized substances throughout the life cycle of a turf pitch and different infill material*

2.1. Literature review of infill materials

In the literature review, articles with extensive environmental and health risk assessments associated with artificial turf and infill materials was studied. Risk assessments was classified due to how extensive they were, based on the compliance to the risk assessment methodology steps: 1) Information Statement / problem definition, 2) Impact Analysis, 3) Exposure analysis, 4) Risk Characterization, and 5) Overall risk assessment. These methodology steps are stated in the European Technical Guidance Document for Risk Assessment (TGD 2003 and 2003b). Further, risk assessments system boundaries and results was compiled. Extensive risk assessments that included risk characterization was further reviewed.

2.2. Analysis of energy use and greenhouse gas emissions

The study was based on a fictive case of an artificial turf field in Sweden, constructed and maintained over a ten year time period and finally removed. The system boundaries were limited to include production of input materials, use of vehicles and construction machines for providing an artificial turf surface with a service life of ten years and finally disposing it. Surrounding facilities were excluded such as lighting, paved areas, fencing and grandstands.

Material and energy flows were inventoried. Information about activities and dimensions for construction together with maintenance and removal activities was gathered from Swedish artificial turf field owners and the Swedish Football association's construction and maintenance recommendations (Swedish Football Association, 2016; 2016b). Data about construction activities was gathered from contractors. Manufacturing of machinery such as excavators and tractors used in construction, maintenance, and final removal was not included.

The studied artificial turf system was selected to reflect construction and maintenance praxis used in Sweden and in other Scandinavian countries. The fictive turf football field is a full sized field measuring 7881 m^2, according to Sweden's Football association's recommendations. The field was not heated during winter time, but was assumed to be brushed, harrowed, plowed and salted to allow playing during the whole year.

2.3. Materials

In Table 1, the material used during construction, maintenance and removal of artificial turf are presented. For the study, a layer of 0.5 m soil corresponding to 3 940 m^3, is excavated in order to prepare for a subbase and an extra 240 m^3 to prepare for drainage. A drainage system with a herringbone pattern comprising of 1 500 meters of drainage pipes of thermoplastic polyvinyl chloride (PVC) is installed. Drainage is sealed with geotextile and gravel. The subbase consist of a 0.4 m layer of crushed rock and a 0.05 m layer of fine sand produced in quarries. An SBR –based shock pad is manufactured and laid out on site. The artificial turf mat is manually rolled out and glued together with a strip of paper and two-component polyurethane-based adhesive. Infill sand and infill of either SBR, EPDM, TPE or R-EPDM are installed and lines are painted. For maintenance, the artificial turf was brushed and harrowed once a week. The annual refilling of infill is 5 ton. During winter time, snow is plowed. Annual snow amount is estimated to 30 % of total annual rainfall which is 600 mm (SMHI, 2015). Natrium chloride is spread to remove ice, annual use is 0.57 kg/m^2.
The artificial turf, infill, and shock absorbing pad is removed and incinerated after ten years of use. Incineration ash is disposed at a waste disposal site.

Table 1. *Material use in construction, maintenance and removal of 7881 m² of artificial turf.*

Material use	**Construction [kg/m²]**	**Maintenance [kg/m²]**	**Removal [kg/m²]**
Excavated soil and rock	840		
Drainage pipes	0.11		
Geotextile	0.06		0.01
Aggregates for subbase and drainage	49+648		
Fine sand for subbase	80		
Shock absorbing pad system	8.8		
Turf mat	1.87		
Two component turf adhesive	0.06		
Turf line paint	0.01		
Sand infill	22.5		
Infill alternatives			
SBR*	6.47	6.34	
TPE*	11.0	6.34	
EPDM*	7.7	6.34	
R-EPDM*	7.7	6.34	
Natrium chloride		5.7	
Snow		150	
Ash from rubber and plastic material incineration			0.95
Soil for recultivation layer and sealing layer			0.32+0.41
Crushed rock for upper / under drainage layers			0.9+0.9
Sand for upper and under protection layer			0.3+0.3
Bentonite			0.006

[1]Infill density varies between types (Skenhall et al., 2012)

In Table 2 the input materials are presented. Production and emission data for materials was gathered from Baitz et al. (2004) for drainage pipes, from Svingby & Båtelsson (1999) for geotextile, from Stripple (2001) for quarry products and Natrium chloride, from ECOBILAN (1993) for paint.

Table 2. *Composition of input materials*

Polymers based products	Composition
Turf mat	50 % polyethylene 50% polypropylene
Turf and Pad adhesive	Methylene diphenyl diisocyanate (MDI) 30 % and 70 % Calcium carbonate
Turf line paint	Water based, Styrene – Acrylic paint
TPE infill	40 % Calcium carbonate, 20 % Polypropylene, 20 % Polyethylene, 16 % Mineral oil, 4 % Phthalates
EPDM infill	68 % Calcium carbonate, 22 % EPDM, 8 % Mineral oil, 4 % Phthalates
Pad	12 % adhesive, 88 % SBR granules
PVC pipe	98 % PVC Polymer, 2 % stabilizers
Soil and rock materials	
Sand infill	Crushed rock, 1.5 ton/m3
All other aggregates	Crushed rock, 1.6 ton/ m3

For the energy use and greenhouse gas emissions of compounds in SBR and R-EPDM infill the compositions are not of interest since these materials are recycled and allocation of the environmental impact burdens the former rubber products. However, the composition of rubber and other materials affects the shredding process. The production of SBR and R-EPDM included collection, sorting and shredding of car tires respectively rubber products. The transportation distances for collection of waste tire and rubber was assumed to be 200 km with a fill ratio of 38 % based on assumptions by Skenhall et al. 2012. It was assumed that 1 kg of tire produced 0.67 kg SBR granules. About 40 % of the cutting of tires was run on electricity driven and 60 % was run on diesel. The shredding was run on electricity. The production of electricity was based on Swedish average electricity production and emission data (Stripple, 2001). The process for producing R-EDPM was assumed to be identical to the production of SBR.

Thermoplastics (TPE) are normally produced by using compounds including numerous additives and one or several polymers (Biron, 2015). Any production recipes from TPE infill producers could not been compiled. It was assumed, based on Skenhall et al. (2012) that TPE infill consisted of 40% calcium carbonate, 40% SEBS (Styrene Ethylene Butylene Styrene), 16 % mineral oils and 4 % phthalates. SEBS was assumed to be based on 50 % polyethylene and 50 % polypropylene. Production and emission data was gathered from (Biron 2015) for polyethylene and polypropylene, Shtiza et al. (2012) for calcium carbonate, McManus (2001) for mineral oils and Li (2013) for Phthalates.

Production recipes for EPDM infill producers could not be compiled either. Production and emission data was assumed to be equivalent to an average of production and emissions data for EPDM presented by Skenhall et al. (2012) and Stripple (2001).

The shock absorbing pad was assumed to consist of 88 % SBR granules mixed with 12 % two component polyurethane binder, based on a data from BASF (2006). The two component adhesive was assumed to consist of 30 % Methylene diphenyl diisocyanate (MDI) and 70 % Calcium carbonate. Same assumption was made for the turf adhesive. Production and emission data was taken from TEC (2015), Shtiza et al. (2012) and Pavlovich et al. (2011).

Any production recipes for turf could not be compiled. According to Nilsson et al. (2008), polypropylene, polyethylene and polyamide is used for turf production. Calculations were based on a simplified recipe which consisted of 50 % polypropylene and 50 % polyethylene. Production and emission data was gathered from Biron (2015).

For the final removal, it was assumed that all polymer based products were incinerated. Data for incineration of polymers was gathered from Eriksson and Finnveden (2009).

In Table 3 a presentation is given on the energy use and greenhouse gas (GHG) emissions for producing the input materials or its constituents

Table 3. *Energy use (MJ/kg) and GHG emissions (kg CO2 equivalents/kg) for polymers, polymer-based components, aggregates and minerals*

Material	Energy use [MJ/kg]	Total GHG emissions [Kg CO2 eqv./kg]	Reference
SBR and R-EPDM[1]	2.65 [2]	0.06	Skenhall et al., 2012
Polyethylene	80	1.8	Biron, 2015
Polypropylene	78	2.1	Biron, 2015
Calcium carbonate	0.74	0.040	Shtiza et al., 2012
Mineral oil	5.94	3.56	McManus, 2001
Phthalates	70	1.39	Li, 2013
EPDM	24.3	1.53	Skenhall et al., 2012 and Stripple 2001
MDI	57.4	2.77	Pavlovich et al., 2011
Water based, Styrene-Acrylic paint	43	1.90	ECOBILAN, 1993
PVC pipe	67.2	0.767 [3]	Baitz et al., 2004
Geotextile	26.1	3.12	Simon, 2008
Polymer based materials, energy recovery	-25.9	2.24	Skenhall et al., 2012, Eriksson & Finnveden, 2009
Quarry materials, crushed	5.4 [2]	0.0015	Stripple, 2001
Soil and gravel from quarries, excavated	0.0014 [2]	0.010	Stripple, 2001
Natrium Chloride	1.59	0.139	Stripple, 2001
Bentonite	0.0066	0.00152	Stripple, 2001

[1]*Recycled material. Energy use and GHG emissions from the production of tires and rubber products is allocated to previous tyre and rubber products.*
[2]*Process energy use only*
[3]*It is assumed that the process energy use is of 100 % electricity. Calculations on GHG emissions are based on Swedish electricity mix 2001*

2.4. Vehicles and working machines

In Table 4, transport distances and use of vehicles and working machines is presented. Excavation and construction of drainage and subbase was assumed to be performed with an excavator. In addition, a compactor and a planer machine was used for the subbase. Transportation of all materials was assumed to be made with a medium sized distribution truck. Maintenance work was assumed to be made with a small tractor. The final removal of the artificial turf materials was assumed to be made with an excavator.

Table 4. *Transport distances and use of vehicles and working machines*

Vehicle and working machines use	Construction	Maintenance	Removal
Lorry transport distances (Unit: kilometers, one way)			
Turf mat, elastic infills, pad , adhesives, paint, salt, drainage pipes	1 000	1 000	
Quarry materials, excavated soil and rock, and geotextile, snow	50	50	
Turf mat, infills, pad, adhesives and paint to incineration			50
Ashes from incineration to landfill			50
Working machines (Unit: hours, if not specified)			
Compactor	8		
Plainer machine	10 rounds*		
Brushing, harrowing with tractor		2 000	
Plowing, salting with tractor		600	

**Plaining is assumed to require 10 rounds of plaining*

Emission data for vehicles and machines was gathered from Stripple (2001), Holmström (2013) and Bauman & Tillman (2004). Loading and unloading of quarry materials and excavated materials are made with an excavator of class 1 with an energy use of 2.65 MJ diesel/ m^3. For other materials, loading and unloading are assumed to be made manually. The lorry for transportation was assumed to consume 1.7 MJ diesel/ ton-kilometer. The tractor that are used for maintenance purposes are assumed to consume 140 MJ diesel / operation hour. The plainer machine was assumed consume 0. 53 MJ diesel / m^2.

2.5. Material sampling and chemical analysis

Chemical analysis was performed on one material sample from new infill of SBR, TPE, EPDM and R-EPDM. Samples of about 10 kg of SBR, TPE, EPDM and R-EPDM infill was taken from previously not opened infill bags. Sample of SBR infill was newly produced and taken directly in a car tire shredding factory in Sweden. The TPE infill bag was about one meter high and 0.5 meters in diameter. Samples were taken about 0.1 m from top layer. The infill bag of EPDM was about 2 m high and 1.5 meters in diameter. Material sample was taken about 0.3 m from top layer. The infill bag of R-EPDM was about one meter high and 0.5 meters in diameter. Samples were taken about 0.1 m from top layer.

Chemical analysis was performed by an accredited laboratory. Leachate water from infill were produced by a single-stage shaking test at L / S = 10 in accordance to prEN 12457-2. Leachates were filtrated. Analysis of Antimony, Arsenic, Barium, Lead, Cadmium, Copper, Chromium, Mercury, Molybdenum, Nickel, Selenium, Zinc followed the EN 12457-2 standard. Analysis of chloride, fluoride and sulfate were conducted according to EN ISO 10304-1: in 2009. Distillable phenols and dissolved organic carbon (DOC) were analyzed according to EN 028 128: 1976 and EN 1484: 1997 respectively.

Leachate water was further analyzed by the laboratory for semi-volatile organic compounds by using a Gas Chromatography Mass Spectrometry (GC-MS) column instrument. The analysis detect organic compounds with boiling points from 100 to 500 degree Celsius consisting of 8 to 35 carbon atoms. Each substance detected is compared to a reference library and is further described with suggested name, cas number and quantity. The method has lower measurement certainties than in methods for full quantitative assessment. However, some substances are routinely search for, these are PAH 16, PCB7, phthalates, chlorobenzenes, chlorophenols, phenols and alkylated phenols, C9-10 aromatics, some chlorated organic substances and some nitrogen containing organic substances.

The substances toxicity was then further analyzed by the authors. The substances CAS numbers was used to find information on the substances toxicity to aquatic life and humans in the Classification and Labelling Inventory Database provided by the European Chemicals Agency (ECHA, 2016).

3. RESULTS AND DISCUSSION

3.1. Literature review

The results from the literature review is presented in Table 5. The health and environmental risk assessments has been sorted. Sorting was based on how extensive they are and which of the (RA) step (1-5) are included. In addition, the system boundaries and material studied are described.

Table 5. *Environmental and health risk assessments of artificial turf fields and its materials*

Reference	RA step	Materials/ system studied. (Material Supplied = MS)	Assessment results (Local environmental risk= ER, Health risk= HR)
Pavilonis et al., 2014	**1,2,3, 4**	Infill (SBR) and turf fibers. Outdoor or indoor not specified. **MS:** US	**HR:** Risk due to dermal, ingestion and inhalation exposure to infill and artificial turf was generally considered de minimus. Relatively high content of lead in one turf fiber.
Ruffino et al., 2013	**1,2,3, 4**	Outdoor fields with (SBR and TPE) infill. **MS:**: Italy	**HR:** For dermal and inhalation exposure, the cumulative carcinogenic risk was lower than 10−6 and the cumulative noncarcinogenic risk lower than 1.
Kim et al., 2012	**1,2,3, 4**	Turf, Infill (SBR, EPDM), back coating, elastic pavement. **MS:** Korea	**HR:** Minimal direct health risk regarding dermal, inhalation and ingestion exposure, except for ingestion exposure for children with pica.
Ginsberg et al., 2011	**1,2,3, 4**	Outdoor and indoor fields with infill (SBR). **MS:** N/A	**HR:** No elevated adverse health risks due to inhalation exposure. Adequate ventilation is recommended.
Menichini et al., 2011	**1,2,3, 4**	Outdoor field and Infill (Coated SBR, Uncoated SBR, TPE and R-EPDM). **MS:** Italy	**HR:** For the benzopyrene. an excess lifetime cancer risk of 1×10−6 due to inhalation was calculated for an intense 30-year activity at SBR fields
Lim & Walker, 2009	**1,2,3, 4**	Infill material (SBR) and outdoor field **MS:** USA	**ER:** No organics and low levels of metals detected in surface water. No impact on groundwater. SBR entirely from truck tires was estimated to possibly have an impact on aquatic life due to zinc exposure **HR:** Inhalation exposure does not indicate a concern for non-cancer or cancer effects. Football fields are not important contributors of exposure to particulate matter. SBR is no source for lead exposure when compared to federal hazard standard for lead in soil.
Nilsson et al., 2008	**1,2,3, 4**	Infill (SBR, Coted SBR, TPE, EPDM and Coir), turf mats, pad and road salt. **MS:**: Norway	**ER:** No major risk **HR:** Dermal and oral exposure is concluded to cause minimal risk. Potential allergic risk due to dermal exposure for benzothiazole and amines in SBR and EPDM for sensitive individuals.
Verschoor, 2007	**1,2,3, 4**	Infill (SBR) **MS:** Netherlands	**ER:** Potential ecotoxicological risk in surface water, groundwater and soil may occur.
Vidair et al., 2007	**1,2,3, 4**	Infill (SBR), rubber surfaces (SBR), soil **MS:** USA	**ER:** Small regarding exposure to soil and ground water **HR:** Minimal regarding ingestion and dermal exposure

Moretto, 2007	**1,2,3, 4**	Infill (SBR, EPDM, TPE) at outdoor and indoor fields **MS:** France	**ER:** Minimal impact on water resources and the aquatic environment in the short and medium term. **HR:** Health risks associated with the indoor inhalation of VOC and aldehydes present no actual cause for human health. No cause for concern as regards human health for the workers, general public and professional or amateur athletes, whether adults or children indoors. Good ventilation is recommended in case of workers installing artificial surfaces in small and poorly ventilated gymnasia
Birkholz et al., 2003	**1,2,3, 4**	Infill (SBR) **MS:** Canada	**ER:** Significant risk of contamination in surface water or groundwater is doubtful. **HR:** The cancer risk due to ingestion exposure is minimal
Norw. Inst. of Publ. Health & the Radium Hospital, 2006	**1,2,3, 4**	Indoor fields and Infill (SBR) **MS:** Norway	**HR:** No increased risk of leukemia due to inhalation exposure. No elevated risk for contact allergies due to dermal exposure. The possibility for latex allergy due to inhalation exposure cannot be entirely eliminated. SBR should not be used indoors when infill is replaced, due to lack of knowledge about potential latex allergy risk.
Schiliro et al., 2013	**1,2,3**	Outdoor fields with (SBR and TPE) infill **MS:** Italy	**HR:** Inhalation exposure present no more exposure risks than the rest of the city.
USEPA, 2009	**1,2,3**	Outdoor field with (SBR) infill **MS:** USA	**ER:** No conclusions on risks are made **HR:** No conclusions on risks are made
Joost et al., 2010	**1,2,3**	Outdoor field with infill (SBR)**MS:** Netherlands	**HR:** Minimal uptake of PAHs regarding all exposure ways
Johannesson & Sandén, 2007	**1,2,3**	Outdoor field with SBR infill **MS:** Sweden	**HR:** No increased risk for cancer regarding dermal, ingestive and inhalation exposure
Dye et al., 2006	**1,2,3**	Indoor halls with infill (SBR and TPE) **MS:** Norway	**HR:** The use of SBR causes a considerable burden on the indoor environment. For all three halls, organic chemicals are found in air.
Tekavec & Jakobsson, 2012	**1,2,3**	Outdoor field and SBR infill **MS:** Sweden	**HR:** Levels of PAH and Phtalates was similar to levels in general population. Due to the precautionary principle, other types of infill than SBR is recommended to be used.
Christensson & Antonsson, 2004	**1,2,3**	Indoor field with 50 % SBR and 50 % EPDM infill **MS:** Sweden	**HR:** Levels of heavy metals and benzoaporen was significantly below air limit standards.
Ottesen et al., 2011.	**1,3**	Shock-absorbing surfaces with SBR and EPDM **MS:**: Norway	**ER:** N/a. Leaching of THC (C12-C35), PAH, PCB , A health risk assessment needs to be conducted. THC (> C5 - C35) , zinc , nonylphenol , PAHs and PCBs was found in all products

Widenbrant, 2011	**1,3**	Outdoor fields with SBR infill **MS:** Sweden	**ER:** Water quality is within drinking water standard
Van Ulirsch et al., 2010	**1,3**	Turf fibers. **MS:** USA and South Korea	**HR:** Synthetic turf can deteriorate to form dust containing lead at levels that may pose a risk to children. Exposure pathways has not been specified.
Bristol & McDermott, 2008	**1,3**	Outdoor field and infill (SBR). **MS:** USA	**ER:** Aquatic toxicity not detected. Indicated levels of dissolved zinc in drainage but at concentrations less than the applicable Water Quality Standard. **HR:**: No detectable concentrations of volatile nitrosamines or 4-(tert-octyl) phenol existed in the air column. Benzothiazole: present at a very low concentration directly above one of the two fields sampled. Not detected in any of the upwind or downwind locations at either field
Zhang et al., 2008	**1,3**	Turf fibers and SBR infill **MS:** USA	**ER:** New SBR did not contain PAHs at levels above health-based soil standards. The zinc contents in SBR were found to far exceed the soil limit **HR:**: Zero or near-zero bioaccessibility in the synthetic digestive fluids regarding PAH in SBR .Generally relatively low concentrations of lead in SBR. Bioaccessability of lead from SBR and fiber in the synthetic gastric fluid, and from fiber in intestinal fluids.
Hofstra 2007	**1,3**	Infill (SBR). **MS:** Netherlands	**ER:** No risk due to air emissions, to soil zink is a relative parameter. **HR:** No risk regarding ingestion, inhalation and dermal uptake
Plesser & Lund, 2004	**1,3**	Infill (Recycled rubber, EPDM) and Turf. **MS:** Norway	**ER:** Leachate of zinc from turf and recycled rubber indicates that the leachate water is very strongly polluted water. With the exceptions of chromium and zinc, EPDM rubber contained smaller quantities of hazardous substances than the recycled rubber types overall.
Krüger et al. 2013	**1,2**	Turf fibers, Infill (SBR, TPE, EPDM), Shock Pad, Sub base	**ER:** Aside from recycled rubber compounds, synthetic plastics can also pose ecotoxicological risks, which might be even more serious

The assessment results show that for SBR infill, the health risks due to dermal exposure are minimal (Pavilonis et al., 2014; Ruffino et al., 2013; Kim et al., 2012; Nilsson et al., 2008; Vidair et al., 2007). However, one study concludes that there could be a potential allergic risk for sensitive individuals due to dermal contact (Nilsson et al., 2008). Further, the health risks due to ingestion of SBR infill are concluded to be minimal (Pavilonis et al., 2014; Kim et al., 2012; Nilsson et al., 2008; Vidair et al., 2007; Birkholz et al., 2003), except for children with Pica, i.e. children that are persistently eating of substances with no significant nutritional value. For these individuals, the risk due to ingestion is not minimal (Kim et al., 2012). Health risk assessment of exposure by inhalation of emissions from SBR infill on artificial fields outdoor concludes that the risks are minimal (Pavilonis et al., 2014; Ruffino et al., 2013; Kim et al., 2012; Ginsberg et al., 2011; Menichini et al., 2011; Lim & Walker, 2009; 2006; Norwegian

Institute of Public Health and the Radium Hospital, 2006; Moretto, 2007). Further, the health risks due to inhalation on indoor artificial fields with SBR infill is concluded not to increase the risks of cancer (Norwegian Institute of Public Health and the Radium Hospital, 2006; Ginsberg et al., 2011). However the potential latex allergy risk due to inhalation exposure should be further studied (Norwegian Institute of Public Health and the Radium Hospital, 2006).

Regarding local environmental impact due to SBR infill, the risks are generally estimated to be small (Moretto, 2007; Nilsson et al., 2008; Vidair et al., 2007; Birkholz et al., 2003), however it is assessed that SBR infill entirely made of truck tires could have an impact on aquatic life due to zinc exposure. Also, Verschoor (2007) estimate a potential risk on water, groundwater and soil due to zinc exposure from SBR infill. Here, a linear zinc leaching was assumed which implies that the zinc load limit for building materials decree is exceeded (Verschoor, 2007). In contrast, assessments that are based on measurements of drainage water quality, indicate that the local environmental risks are small (Lim & Walker, 2009; Nilsson et al., 2008).

For TPE infill, a limited number of health risk assessments have been conducted. The assessment results show that for TPE infill, the risks due to dermal exposure are minimal. (Ruffino et al., 2013; Nilsson et al., 2008). Further, Nilsson et al., (2008) assess that the risk for exposure due to ingestion of TPE infill are minimal. Regarding the health risk via exposure by inhalation on outdoor fields with TPE infill, cancer risk was assessed to a minimum (Ruffino et al., 2013) both at indoor and outdoor fields (Moretto, 2007). Regarding environmental risk, Moretto (2007) concluded that TPE infill has minimal impact on water and the aquatic environment in the short and medium term.

For EPDM infill, a limited number of health risk assessments has been conducted, Assessment results show that the health risk due to dermal exposure is minimal (Nilsson et al., 2008; Kim et al., 2012) but the risk for allergy for sensitive individuals cannot be excluded (Nilsson et al., 2008). The risk due to exposure from ingestion and inhalation is minimal (Kim et al., 2012; Nilsson et al., 2008). Moretto (2007) examined the health effects of exposure via inhalation and concludes that the risks due to exposure from inhalation are minimal at outdoor and indoor courts.
Only a few assessments of local environmental impact due to exposure from EPDM infill have been made. One environmental risk assessment concluded that the risk of impact on groundwater and aquatic life is small (Nilsson et al., 2008). Moretto (2007) concluded that EPDM has minimal impact on water and the aquatic environment in the short and medium term.

For R-EPDM infill, no health or environmental risk assessments have been found in the literature.

The review of health and environmental risk assessments shows that most studies have been focused on SBR infill, and that many studies does not include TPE and EPDM infill. For R-EPDM no risk assessments was found. Environmental assessments of SBR infill has primarily been focused on zinc leaching to water. It can be seen that there are generally low risks from using the studied infill materials, however studies where different infill materials are analyzed and compared on the same premises are few. The local environmental and health risks assessments that was found in the literature is presented in Table 8.

Table 8. *Health risk and local environmental risk assessment results.*

Infill type / Risk	SBR	TPE	EPDM	R-EPDM
Health				
Dermal	-Minimal [1,2,3,4,5] -Potential allergic risk for particularly sensitive persons[6]	Minimal [2,6]	-Minimal [3] -Potential allergic risk for particularly sensitive persons[6]	**Not found**
Ingestion	Minimala [1,5,6,7]	Minimal[6]	Minimal[6]	**Not found**
Pica-children	Not minimal[3]	**Not found**	Not minimal[3]	**Not found**
Inhalation, indoors	Minimal [1,2,3,4,8,9, 12]	Minimal[2,12]	Minimal[3, 6, 12]	**Not found**
Inhalation, outdoors	No increased cancer risk [8, 10] Minimal[12]	Minimal[12]	Minimal[12]	**Not found**
Local environment				
Ground water	-Potential risk[11] -Small effect[4,5,6] - Minimal[12]	- Minimal[12]	- Minimal[12]	**Not found**
Surface water	-Potential risk/effect[4,11] -Small risk/ effect[6,7] - Minimal[12]	- Minimal[12]	- Small risk[6] - Minimal[12]	**Not found**
Soil	-Potential risk[11] -Small risk[7]	**Not found**	**Not found**	**Not found**

1 Pavilonis et al., 2014
2 Ruffino et al., 2013
3 Kim et al., 2012
4 Lim & Walker, 2009
5 Vidair et al., 2007
6 Nilsson et al., 2008
7 Birkholz et al., 2003
8 Ginsberg et al., 2011
9 Menichini et al., 2011
10 Norwegian Institute of Public Health and the Radium Hospital, 2006
11 Verschoor, 2007
12 Moretto, 2007

3.2. Energy use and GHG emissions

In figure 3, 4 and 5, the energy use and GHG emissions for the use of materials (including transportation) and the use of working machines throughout the life cycle of artificial turf is described.

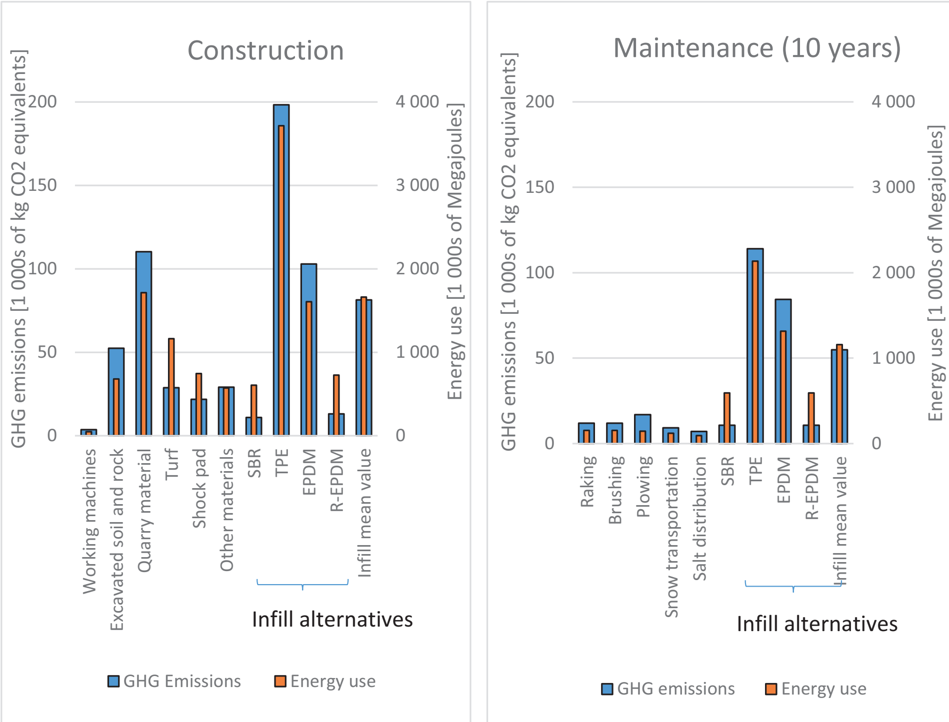

Figure 3 and 4. *The energy use and GHG emissions of materials (including transports) and the use of working machines from construction and maintenance of 7 881 m ^{2} artificial turf.*

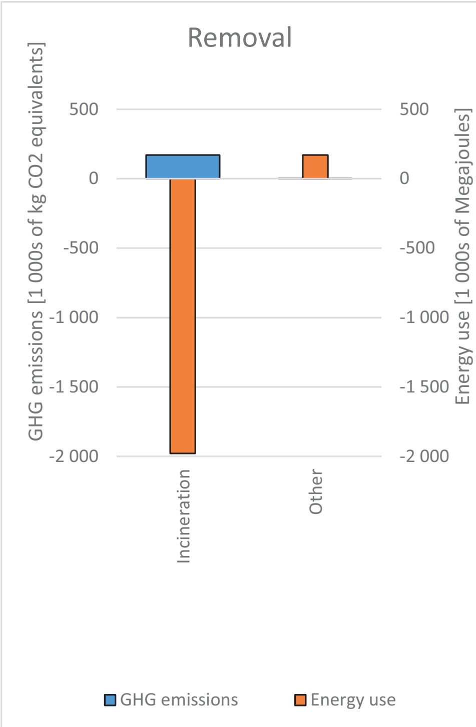

Figure 5. *The energy use and GHG emissions of materials (including transports) and the use of working machines from removal of 7 881 m^2 artificial turf.*

The infill material alternatives have large impact on total energy use and GHG emissions. Throughout construction and maintenance, the energy use for TPE was 5.85 Gigajoule (GJ), about 2 times higher than for EPDM, about 4.4 times and 4.9 times higher than for R-EPDM and SBR respectively. Similarly GHG emissions from TPE was 312 ton CO2 equivalents, about 1.7 times higher than for EPDM, and about 13 times higher for R-EPDM and 14 times for SBR respectively. The large differences between the energy use and GHG emissions of infill materials was mainly due to the differences in production, where TPE and EPDM infill are produced by new polymer compounds while SBR and R-EPDM infill are produced by waste materials. Here, the polymer compounds of these waste materials does not burden the SBR and R-EPDM infill.

The excavation and use of quarry materials for subbase was large contributors to energy use and GHG emissions in the construction phase, corresponding to about 2.4 GJ and 163 ton CO2 equivalents. Here, the transportation distances was of major importance. Studies have revealed potentials in reducing energy use and GHG emission by reusing aggregates (Magnusson et al. 2015; Hossain et al. 2016) or by constructing the subbase with alternative lightweight materials (Williams et al. 2010). With the exemption for transportation of infill materials, the vehicle use for maintenance practices contributed to 0.58 GJ in energy use and emissions of 50 ton CO2 equivalents. Here, transportation distance for snow is of importance. For the final artificial turf removal, the incineration contributes to a negative energy use, i.e. energy recovery, and the major part of GHG emissions. Incineration contributed to 1.9 GJ of energy recovered and emission of 171 ton CO2 equivalents.

In Figure 6 and 7, the energy use and GHG emissions for construction, maintenance and removal is compared.

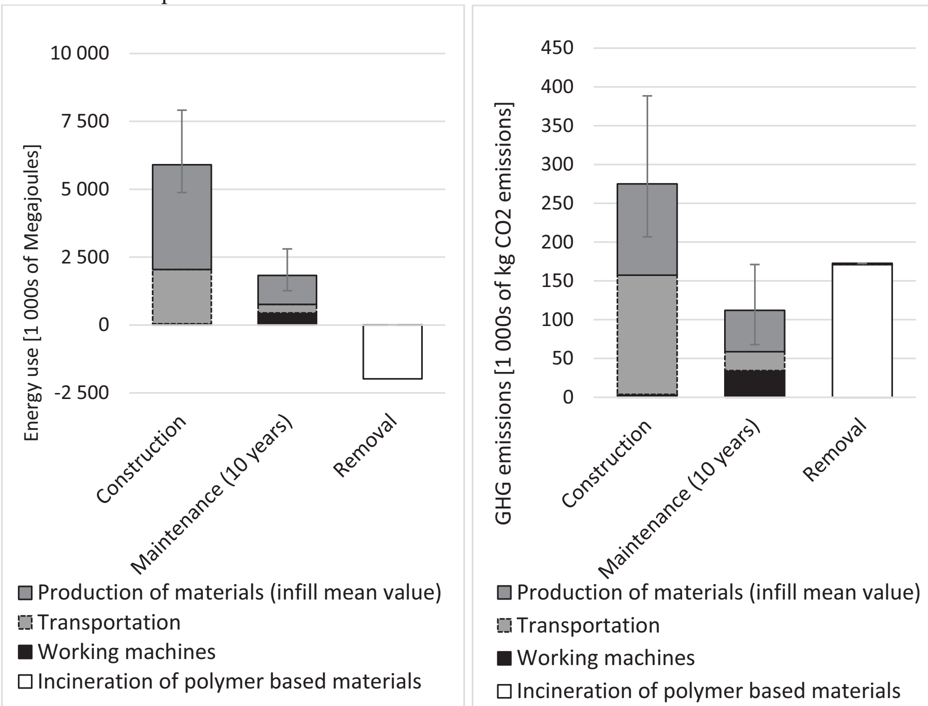

Figure 6 and Figure 7. *Energy use and GHG emissions from construction, maintenance and removal of 7 881 m ^{2} artificial turf. Energy use and GHG emissions for the infill is the mean value of SBR (min value), TPE (max value), EPDM and R-EPDM infill.*

The total energy use was 5.8 GJ and the GHG emissions was 560 ton CO2 equivalents. An average infill was here used and corresponds to an average of GHG emissions and energy use from SBR, TPE, EPDM and R-EPDM infills. The maximum and minimum values for production of materials is related to the infill alternative used. The most significant contribution to GHG emissions was primarily the construction (275 ton CO2 equivalents) followed by removal (172 ton CO2 equivalents) and maintenance (112 ton CO2 equivalents). It can be seen that the production of materials makes up the major part of energy use while also transportation and incineration makes up the major contribution to GHG emissions. Energy recovery from incineration of polymer based materials contribute to reduce the energy use in the life cycle, however it should be stressed that material recycling which prevents production of new materials may be preferable from an energy perspective.

3.3. Chemical analysis

Results from laboratory analysis of infill from single material samples for metals, chloride, fluoride and sulfate, distillable phenols and dissolved organic carbon (DOC) are presented in Table 6.

Table 6. *Leaching of metals and inorganic substances of SBR, TPE, EPDM and R-EPDM produced by a single stage shaking test at L/S 10 according to EN 12457-2 standard. Differences between leachates are marked as bold.*

	SBR [µg/l]	TPE [µg/l]	EPDM [µg/l]	R-EPDM [µg/l]
Antimony Sb	<0.6	<0.6	<0.6	<0.6
Arsenic A	<5	<5	<5	<5
Barium Ba	<200	<200	<200	<200
Lead Pb	<5	<5	<5	<5
Kadmium Cd	<0.4	<0.4	<0.4	<0.4
Copper Cu	<20	<20	<20	<20
Chromium Cr	<5	<5	<5	<5
Mercury Hg	<0.1	<0.1	<0.1	<0.1
Molybdenum Mo	<5	<5	<5	<5
Nickel Ni	<4	<4	<4	<4
Selenium Se	<1	<1	<1	<1
Zinc ,Zn	**94**	**<40**	**<40**	**5 000**
Chloride	**<1 000**	**10 000**	**<1 000**	**1 300**
Fluoride	**120**	**<100**	**<100**	**<100**
Sulphate	**3 000**	**1 000**	**2 200**	**<1 000**
Distillable phenols	**190**	**17**	**<10**	**18**
Dissolved organic carbon	**24 000**	**8 600**	**2 000**	**93 000**

For many substances, the concentrates was under detection limit. For the concentrates above detection, the infill leachates was primarily different for the substances zinc, chloride, sulphate, distillable phenols and dissolved organic carbon (DOC).

The leaching of zinc and DOC was comparably high from R-EPDM. Zinc leaching from R-EPDM was about 53 times higher than from SBR and about 125 times higher than from TPE and EPDM. The leaching of zinc from SBR, EPDM and TPE infill was relatively low compared to previous studies (Ruffino et al., 2013; Plesser & Lund, 2004). Reference values for R-EPDM was not found. The leaching of DOC from R-EPDM was about four times higher than from SBR, about 11 times higher than from TPE and 47 times higher than from EPDM. The leaching of DOC from SBR and EPDM was relatively low compared to results presented by Plesser and Lund (2004). Any reference values for TPE and R-EPDM could not be found.

SBR had comparably high leaching of distillable phenols, about 11 times higher than from TPE and R-EPDM and at least 19 times higher than from TPE. Reference values for phenol leaching has not been found.

Results from laboratory analysis of S-VOCs from single infill material samples are presented in Table 7.

Table 7. *Leached semi volatile organic substances from infill of SBR, TPE, EPDM and R-EPDM produced by a single stage shaking test at L/S 10 according to EN 12457-2 standard.*

	SBR [µg/l]	TPE [µg/l]	EPDM [µg/l]	R-EPDM [µg/l]
PAH16	-	-	-	-
PAH derivatives	-	3.3*	-	-
PCB7, Chlorobenzenes, Chlorophenols and Chlorinated hydrocarbons	-	-	-	-
Phenols, Alcylphenols	1.3	1.9	-	-
Phenol derivatives	1.4	1.8	-	11
Phthalates	-	6.2	-	-
Other specified hydrocarbons				
containing nitrogen	44.9	520	664	1012
not containing nitrogen	0.46	37.7	9.4	-
Oils, unspecified				
Residual oil type, Primarily straight aliphatic hydrocarbons within C20-C36	390		-	
Motor oil type, primarily branched aliphatic hydrocarbons within C20-C35		520	-	
Unknown oil type, aliphatic hydrocarbons within C24-C35			-	38 000
Total leachate	**437**	**1 086**	**702**	**39 023**
Leachate of specified hydrocarbons not found in CLP register	**0.95**	**522**	**321**	**11**

*Possible derivative

The analysis of S-VOC showed that the infill materials of SBR, EPDM, TPE and R-EPDM contained all leachable S-VOC substances. The total leaching of S-VOC was highest from R-EPDM, about 39 mg/l, and comparably much lower from TPE (1.1 mg/l), EPDM (0.7 mg/l) and SBR (0.4 mg/l). The results are similar to previous studies where total VOC leaching from SBR, TPE and EPDM was within the range of 1- 44 mg/l. any reference values for R-EPDM was not found.

No substances of PAH16, PCB7, chlorbenzenes, chlorinated hydrocarbons and chlorophenols was detected in the leachate from the infill materials with the exception of one possible PAH derivative (2,6-Dihydroxynaphthalene) which was detected in the leachate from TPE. Phenols and its derivatives was detected from SBR, TPE and R-EPDM, however previous studies have detected phenols also in EPDM (Nilsson et al., 2008). No reference values was found on R-EPDM.

Phtalates was detected from TPE. Phthalates have been detected in previous studies in leachates from both SBR, TPE and EPDM infill (Nilsson et al., 2008). The leaching of other specified hydrocarbons was highest from R-EPDM, followed by EPDM and TPE and lowest for SBR. Most of these hydrocarbons contained nitrogen.

For R-EPDM, TPE and SBR, a large part of the leachate was of unspecified oil types. The highest leaching was from R-EPDM where aliphatic hydrocarbons within C24-C35 contributed to about 38 mg/l. TPE leachate was of motor oil types, primarily branched aliphatic hydrocarbons within C20-C35 in the level of about 0.5 mg/liter. SBR leachate was with about 0.4 mg/liter of residual oil, primarily straight aliphatic hydrocarbons within C20-C36. No unspecified oils was detected from EPDM.

Some of the specified hydrocarbons detected was not found in the Classification and Labelling Inventory Database (CLP) provided by the European Chemicals Agency (ECHA, 2016). The leaching of such substances was highest from TPE (522 µg/l), followed by EPDM (321 µg/l), and in comparison low from R-EPDM (11 µg/l) and SBR (0.95 µg/l).

Substances which are known to be harmful for the aquatic environment and / or humans was detected in all infill leachates. Eight harmful substances were detected from SBR with a total of 46 µg/l in the leachate. Six substances were detected from TPE with a total of 45 µg/l in the leachate. Four substances were detected from EPDM with a total of 381 µg/l in the leachate. Three substances were detected from R-EPDM with a total of 1012 µg/l in the leachate.

From SBR, toxic substances for aquatic life found was primarily leachate of N-1,3-Dimethylbutyl;-N'-phenylbenzenediamine (14 µg/l) and Acetone anil (11 µg/l). The latter substances has been detected in a previous study (Danish EPA 2008) in concentrations up to 690 µg/l. It was assessed that, based on an uncertain determination of degradation product, this concentration in an aquatic environment may be above the no-effect concentration (Nilsson et al., 2008).

From TPE and EPDM the aquatic toxic substance Kodaflex txib was found in concentrations of 5.5 µg/l from TPE and 9.4 µg/l from EPDM. Kodaflex txib has been detected from EPDM infill in a previous study (Nilsson et al., 2008). In addition, Alkofen B was detected in the leachate from TPE in a concentration of 1.9 µg/l. No previous risk assessments of these substances in infill material was found.

From R-EPDM, the toxic substance for aquatic life detected was Benzenesulfonanilide at a concentration of 960 µg/l. No previous risk assessment of this substance in infill was found.

The results show that all infills tested produced leachates containing substances harmful to aquatic life. For the leachates from TPE, EPDM and R-EPDM, information about potential toxicity could not be found for a large share of the total S-VOCs identified and seems to be missing. However, the analysis are made on only a single material sample from each type of studied infill.

4. CONCLUSIONS

For the construction, maintenance and final removal of artificial turf, the total energy use was 5.9 GJ and the GHG emissions was 572 ton CO_2 equivalents. Here, an average energy use and GHG emission for the infill was assumed. However, the differences between infill materials are large, where the use of TPE and EPDM contributes to relatively high energy use and GHG emissions compared to the use of R-EPDM and SBR. Throughout construction and maintenance, the energy use for TPE was 5.85 Gigajoule (GJ), about 2 times higher than for EPDM, about 4.7 times and 5.2 times higher than for R-EPDM and SBR respectively. Similarly GHG emissions from TPE was 312 ton CO2 equivalents, about 1.7 times higher than for EPDM, and about 17.3 times higher for R-EPDM and 18.3 for SBR respectively. These differences give that the total energy use and GHG emissions for the artificial turf system studied as a whole can vary with a factor of 1.9 and 1.6 respectively, depending on the infill type. However, the complete composition of infills was not possible to find, i.e. the assumptions on infill composition may have significant impact on the energy and climate performance.

The reuse of soil and rock on site in construction, local snow handling and the reuse of removed turf and infill can potentially reduce energy use and GHG emissions of the artificial turf.

The leachates from SBR and R-EPDM contained detectable concentrations of metal (zinc), where the zinc leaching from R-EPDM was comparably high. The leaching of organic substances was highest from R-EPDM followed by TPE and EPDM and lowest from SBR.

Organic substances which are known to be harmful for the aquatic environment and / or humans were detected in all infill leachates. Eight harmful substances were detected from SBR with a total of 46 µg/l in the leachate. Six substances were detected from TPE with a total of 45 µg/l in the leachate. Four substances were detected from EPDM with a total of 381 µg/l in the leachate. Three substances were detected from R-EPDM with a total of 1012 µg/l in the leachate. A possible PAH derivative (2,6-Dihydroxynaphthalene) was detected in the leachate from TPE at a concentration of 3.3 µg/l. Phenols and its derivatives was detected in leachates from SBR, TPE and R-EPDM and phtalates was detected in the leachate from TPE. Other specified substances was nitrogen containing hydrocarbons, where the concentrations was highest in leachates from R-EPDM followed by EPDM and TPE and lowest from SBR. For a large share of these substances in the TPE and EPDM leachates, no information was found in the CLP database. The leaching of unknown organic substances was highest from R-EPDM followed by lower levels from TPE and SBR and no unknown substances from EPDM.

The chemical analysis was made on single material samples, however the results stress the need to include all infill types in future risk assessments. There is an uneven distribution of risk assessments focusing on solely SBR. The lack of risk assessment where all infill materials are evaluated makes it difficult to compare them. For other infill types like TPE and EPDM the assessments are less extensive, and for R-EPDM no studies was found. However, environmental risk assessments mostly concludes that the risks of artificial turf are small.

Since the leachates produced in laboratory may not reflect leachates in the real world, it is therefore suggested that controlled measurements of actual field drainages are developed for further risk assessments.

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