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Handbook of recycled concrete and demolition waste

Edited by F. Pacheco-Torgal, V. W. Y. Tam, J. A. Labrincha, Y. Ding and J. de Brito

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1 Introduction to the recycling of construction and demolition waste (CDW)

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Abstract: The chapter starts with an overview on the recycling of construction and demolition wastes (CDW), followed by a brief analysis on the EU 70% recycling target for 2020. The chapter also includes a book outline.

Key words: construction and demolition wastes (CDW), recycling, resource efficiency, waste management, life-cycle assessment (LCA).

1.1 Introduction

The high volume of construction and demolition waste (CDW) generated today constitutes a serious problem. CDW in the United States is estimated at around 140 million metric tons (Yuan *et al.*, 2012). Eurostat estimates the total for Europe to be 970 million tons/year, representing an average value of almost 2.0 ton/per capita (Sonigo *et al.*, 2010). It should be noted that the figures for CDW generation per capita in Europe have a wide geographical variation (e.g. 0.04 tons for Latvia and 5.9 tons for France). These figures must be viewed as lower estimates, as this type of waste is often dumped illegally. The data are also hard to interpret because of the different waste definitions and reporting mechanisms in different countries (Sonigo *et al.*, 2010).

Recycling of CDW is of paramount importance because it reduces environmental pressure. It prevents an increase in the area needed for waste disposal and also avoids the exploitation of non-renewable raw materials. Environmental impacts caused by the extraction of non-renewable raw materials include extensive deforestation, top-soil loss, air pollution and pollution of water reserves. It should be noted that 40% of all materials are used by the construction industry (Kulatunga *et al.*, 2006). Wang *et al.* (2010) records that construction in China consumes approximately 40% of total natural resources and around 40% of energy. During the last century, the use of global materials increased eight-fold. As a result, almost 60 billion tons (Gt) are currently used per year (Krausmann *et al.*, 2009). It has been forecast that demand for materials will reach at least double the current levels by 2050 (Allwood *et al.*, 2011).

In the context of housing life-cycle assessment (LCA), the environmental gains associated with the recycling of CDW constitute a very small fraction (just 3% in United Kingdom) of the total global warming potential (GWP). Ninety percent of the GWP relates to the use stage (Cuéllar-Franca and Azapagic, 2012). This situation is not confined to the UK residential sector and applies generally in Europe and in other parts of the world where high energy-efficient buildings are the exception (Pacheco-Torgal *et al.*, 2013). However, the recast of the Energy Performance of Buildings Directive (EPBD), which was adopted by the European Parliament and the Council of the European Union on 19 May 2010, sets 2020 as the deadline for all new buildings to be 'nearly zero energy'. This will dramatically increase the percentage (Pacheco-Torgal *et al.*, 2013a).

The benefits of effective CDW recycling are economic as well as environmental. For example, the Environment Agency of the US (EPA, 2002) states that the incineration of 10 000 tonnes of waste can mean the creation of 1 job, landfill can create 6 jobs, but recycling the same amount of waste can create 36 jobs. The recent report *Strategic Analysis of the European Recycled Materials and Chemicals Market in Construction Industry* records that the market for recycled construction materials generated revenues of €744.1 million in 2010 and is estimated to reach €1.3 billion by 2016 (Frost and Sullivan, 2011). This is a low estimate as it does not take into account a near 100% CDW recycling scenario, which will be the future of construction (Phillips *et al.*, 2011).

During the last 15 years, investigations in the field of CDW have focused on three major topics: generation, reduction and recycling. This is guided by the '3Rs' principle (Lu and Yuan, 2011). However, as it is a more complex issue, zero-waste will demand a much wider approach requiring 'strong industry leadership, new policies and effective education curricula, as well as raising awareness and refocusing research agendas to bring about attitudinal change and the reduction of wasteful consumption' (Lehmann, 2011).

1.2 EU 70% recycling target for 2020

According to the revised Waste Framework Directive 2008/98/EC (WFD), the minimum recycling percentage of 'non-hazardous' CDW by 2020 ('excluding naturally occurring material defined in category 170504 (soil and stones not containing dangerous substances) in the *European Waste Catalogue*') should be at least 70% by weight (Saez *et al.*, 2011; del Rio Merino *et al.*, 2011). This target and also the communication *A Resource Efficient Europe* (COM, 2011) indicates the determination of the EU to emphasise the importance of recycling. As the current average recycling rate of CDW for EU-27 is only 47% (Sonigo *et al.*, 2010), increasing it by 70% in just a decade seems an ambitious goal.

CDW are often used as aggregates in roadfill, constituting a down-cycling option. Worldwide aggregate consumption is around 20 000 million tons/year and an annual growth rate of 4.7% is expected (Bleischwitz and Bahn-Walkowiak, 2011). The environmental impact of primary aggregates includes the consumption of non-renewable raw materials, energy consumption and more importantly, the reduction of biodiversity at extraction sites. The cost of aggregates is dependent on transport distances and the price per ton doubles for every 30 km (Van den Heede and De Belie, 2012). Extraction operations therefore have to be near construction sites, which increase the number of quarries and the biodiversity impact. More than one-third of aggregate consumption is related to the production of concrete, which is the most widely used construction material, currently standing at about 10 km^3/year (Gartner and Macphee, 2011). By comparison, the amount of fired clay, timber and steel used in construction represents, respectively, around 2, 1.3 and 0.1 km^3 (Flatt *et al.*, 2012).

Although the use of CDW as recycled aggregates in concrete has been studied for almost 50 years, there are still too many concrete structures made with virgin aggregates (Pacheco-Torgal and Jalali, 2011; Pacheco-Torgal *et al.*, 2013b). This is due to their low cost, lack of incentives, low landfill costs and in some cases, a lack of up-to-date technical regulation (Marie and Quiasrawi, 2012). Recycled aggregates also contain impurities which can be deleterious in Portland cement concrete. It is therefore difficult for the concrete industry to use these materials unless uncontaminated recycled aggregates are used. This issue highlights the importance of developing new binders, which are more suitable for CDW recycling. The WFD 70% target increases the need for effective recycling methods and it is the purpose of this book to make a contribution in this area. It also addresses new techniques for the remediation and/or immobilisation of hazardous wastes such as asbestos and for CDW prevention/reduction, which remains the best option (EC, 2006).

1.3 Outline of the book

Part I is concerned with the management of CDW (chapters 2 to 6).

Chapter 2 considers waste management plans, reviews existing methods in several countries and presents the results of a wide survey assessing the effectiveness of the requirements in implementing waste management plans for the Hong Kong construction industry. Attitudes, benefits and difficulties related to the implementation of waste management plans are discussed.

Chapter 3 covers methods of estimating CDW and includes studies and tables to estimate the amount of CDW. Seventeen examples and case studies on the factors affecting such estimates, the improvement achieved in management scenarios and the benefits of its implementation are presented.

In Chapter 4, waste management plants and technology for recycling CDW are addressed. The types and choices of waste management plants are discussed. Particular attention is given to the health and safety of workers.

Chapter 5 covers the use of multi-criteria decision-making methods for optimal CDW recycling facilities.

Chapter 6 reviews the relevant economic issues in the management of CDW facilities.

Part II is concerned with the processing and properties of recycled aggregates from CDW (chapters 7 to 13).

Chapter 7 compares conventional demolition and deconstruction techniques. A local case study is used to perform a thorough economic analysis, which directly compares these two options.

Chapter 8 reviews demolition techniques in the production of CDW.

Chapter 9 discusses the preparation of concrete aggregates from CDW. This chapter contains data on the relevant technological, economic and environmental aspects of operating CDW recycling facilities, which produce average to high-quality recycled concrete aggregates.

Chapter 10 covers mortar/aggregate separation processes for the improvement of quality in recycled aggregates. In addition to classical mechanical beneficiation, it also analyses cases in which thermal, acid and microwave treatments are used.

Chapter 11 addresses the quality control of recycled aggregates.

Chapter 12 discusses compressive strength, tensile splitting and flexural resistance, elastic modulus, shrinkage and creep.

Chapter 13 addresses several durability parameters (carbonation and abrasion resistance, chloride permeability and resistance to freeze/thaw).

Part III (chapters 14 to 18) deals with the applications of recycled aggregates from CDW.

Chapter 14 reviews the use of recycled aggregates in roads and analyses the properties that make them suitable for use as unbound materials and cement-treated materials in road construction.

Chapter 15 looks at the use of recycled aggregates for asphalt materials. Because recycled aggregates may contain 'hazardous wastes, such as adhesives, lead-based paints (LBP), phenols, formaldehyde resins, polychlorinated biphenyls (PCB), polycyclic aromatic hydrocarbons (PAH) and others', their environmental performance must be assessed by a leaching test. This chapter also discusses the appropriate leaching test and its characteristics. It addresses volumetric properties, rutting, stiffness, fatigue, stripping and durability of asphalt materials containing recycled aggregates.

Chapter 16 is concerned with recycled asphalt for pavements, including pavement removal, properties, mix design and recycling methods.

Chapter 17 reviews the current knowledge on concrete made with recycled aggregates, with a special focus on the importance of impurities in rendering aggregates unsuitable for the production of high performance concrete. The potential of geo-polymers in producing high performance concrete based on high volume recycled aggregates is discussed.

Chapter 18 considers the geo-polymerisation of recycled aggregates.

Part IV (chapters 19 to 24) deals with the environmental issues affecting recycled aggregates from CDW.

Chapter 19 describes the environmental and technical problems arising from gypsum contamination and methods for its removal.

Chapter 20 is concerned with the recycling of materials containing asbestos. Among other issues, it analyses the recycling of thermally treated cement asbestos in the production of concrete (Portland cement based as well as geo-polymeric based).

Chapter 21 describes the decontamination processes for wood treated with organic and inorganic preservatives. Recycling this kind of waste requires the prior removal of organic or inorganic preservative agents. The chapter reviews remediation technologies based on inorganic and organic compounds removal by physical, biological or chemical processes.

Chapter 22 analyses the processing of recycled aggregates contaminated with alkali-silica reaction concrete.

Chapter 23 deals with the LCA of concrete with recycled aggregates. It includes results of LCA case studies on two different RCA applications and discusses the potential and limitations of LCA.

Chapter 24 covers assessment of the potential environmental hazards of concrete made with recycled aggregates. It presents ‘methodologies for environmental assessment with specific overview on their application to construction materials’. It describes relevant experimental tools dedicated to hazard identification and environmental performances and reviews leaching properties and the chemical behaviour of pollutants in the cement matrix. Several examples related to the leaching behaviour of concrete and of recycled aggregates are presented.

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2
Improving waste management plans in construction projects

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Abstract: Growing awareness of managing construction and demolition (C&D) waste has led to its development in construction project management. In 2003, the Hong Kong Government implemented a waste management plan (WMP) for all construction projects. The trial period showed that WMP affects companies' productivity. Its implementation was investigated through a survey and structured interviews. The main benefits were found to be the on-site reuse of materials and methods for reducing waste. Financial incentives and costs present major difficulties and the use of prefabricated building components was considered as the most effective measure in encouraging the implementation of WMP.

Key words: waste-management-plan, construction and demolition waste, effectiveness, construction, Hong Kong.

2.1 Introduction

Historically, waste management in the construction industry has not been successfully controlled and much work remains to be done if a satisfactory standard is to be reached. Construction waste in places such as Hong Kong was commonly dumped in landfills. However, landfill space is expected to run out in the next few years (Hong Kong Government – Environmental Protection Department, 2006), and there is an urgent need to discover alternative methods of waste management. The Hong Kong Government has employed various waste control measures during the past few years, including:

- issuing a waste disposal ordinance;
- developing and launching a green manager scheme;
- drafting a waste reduction framework plan;
- commissioning a pilot concrete recycling plant;
- stipulating implementation of waste management planning (WMP) methodology in industry;
- promoting a public landfill charging scheme (Hong Kong Government, 2006).

The use of a new scheme using WMP methodology has received negative feedback from the industry. Construction organisations complained of procedures which had a damaging effect on their productivity and were impractical to implement. The method therefore appears to be ineffective in construction projects.

This chapter focuses on the examination of waste generated in construction and on the existing waste controlling measures employed by the Hong Kong Government. It explores the effectiveness and difficulties of the existing WMP methodology and proposes ways of addressing the problems which exist in the implementing of WMP methods.

2.2 Existing waste management planning (WMP) measures and methods of control

There has been an overwhelming promotion of waste management tools and sustainable development activities in recent years. As a result, there is a growing awareness of waste management issues and the potential problems from negative impacts on the environment. Construction is not an environmentally friendly activity. Comprehensive reviews were encouraged to provide waste management measures in construction activities (Tam *et al.*, 2005b, 2006b). Various factors affect the environment, including land use and deterioration, resource depletion, waste generation and various forms of pollution (Tam *et al.*, 2005a, 2006a).

Debris from construction and demolition (C&D) works constitutes a large proportion of solid waste (Table 2.1). In the United Kingdom, more than 50% of landfill waste comes from construction (Ferguson *et al.*, 1995), while 70 million tons of waste are produced by C&D activities annually (Sealey *et al.*, 2001). In Australia, about 14 million tons of waste go to landfill annually and 44% of this total is attributed to the construction industry (Craven *et al.*, 1994; McDonald, 1996). In the United States, around 29% of solid waste is produced by construction (Hendriks and Pietersen, 2000) and in Hong Kong, the figure is around about 38% (Hong Kong Government – Environmental Protection Department, 2006).

Waste management promotion worldwide can be summarised as:

- **China**: C&D waste management was promoted and placed under scrutiny by the law on Prevention of Environmental Pollution Caused by Solid Waste, (Regulations on the Urban Environmental Sanitation Management and the Administration Measures for Urban Living Waste) (Lu and Yuan, 2010).
- **Finland**: the promotion of waste management is based on the Waste Management Law activated in 1993 (Ref no. 1072/1993). Treatment of construction waste is more closely regulated by the government decision Ref no. 295/1997 (Hendriks and Pietersen, 2000). The main points in the decision are:
  - a goal of 50% of construction waste (not including soil materials, aggregate and dredging waste) to be recycled in 2000;
  - the decision to be followed on construction sites with more than 800 tons of soil, rock material and dredging waste, or more than 5 tons of other waste;

Table 2.1 Comparison of proportions of construction solid waste

Country	Proportion of construction waste to total waste (in %)	C&D waste recycled (in %)	Source(s)
Australia	44	51	Hendriks and Pietersen, 2000
Brazil	15	8	Hendriks and Pietersen, 2000
Denmark	25–50	80	Hendriks and Pietersen, 2000
Finland	14	40	Construction Materials Recycling Association, 2005; Hendriks and Pietersen, 2000
France	25	20–30	Construction Materials Recycling Association, 2005; Hendriks and Pietersen, 2000
Germany	19	40–60	Construction Materials Recycling Association, 2005; Hendriks and Pietersen, 2000
Hong Kong	38	No information	Hong Kong Government – Environmental Protection Department, 2006; Poon, 2000
Japan	36	65	Construction Materials Recycling Association, 2005; Hendriks and Pietersen, 2000
Italy	30	10	Construction Materials Recycling Association, 2005; Hendriks and Pietersen, 2000
Netherlands	26	75	Construction Materials Recycling Association, 2005
Norway	30	7	Hendriks and Pietersen, 2000
Spain	70	17	Hendriks and Pietersen, 2000
United Kingdom	Over 50	40	Hendriks and Pietersen, 2000
United States of America	29	25	Construction Materials Recycling Association, 2005; Hendriks and Pietersen, 2000

- construction work to be planned and carried out so as to separate various kinds of construction waste, including concrete, bricks, mineral boards, ceramics, gypsum products, wood products, metal products, soil, material and dredging waste;
- finally, about 40% (by weight) of the C&D waste to be reused.

- **Germany**: According to the *Closed Substance Cycle and Waste Management Act* (Hendriks and Pietersen, 2000), the producer is responsible for products in the fields of commercial waste and C&D waste and is obligated to give priority to the recycling of these wastes. Finally, all parties involved in the cycle of building materials or the building itself are responsible for waste resulting from the use, maintenance, refurbishment or demolition of a structure in the construction industry, the building materials industry, the back building industry and the recycling industry. A member of an action group in the building industry, Kreislaufwirtschatstrager Bau (KWBT), presented 'Voluntary Self-Commitment for the Involved Parties of the Building Industry and Associations on the Recovery of C&D Waste with Regard to Environmental Impact' to the responsible Federal Minster of the Environment in 1996. The Ministry is informed of success and progress resulting from self-commitment by monitoring.

- **Ireland**: the government issued a report on recycling C&D waste entitled 'Task Force B4: Recycling of C&D Waste, Final Report on the Development and Implementation of a Voluntary Construction Industry Programme to Meet the Government's Objectives for the Recovery of C&D Waste' (Forum for the Construction Industry, 2001). This aimed to coordinate the development and implementation of a voluntary construction industry programme to meet government objectives for recovery of C&D waste as set out in the policy statement 'Changing our Ways', and to present this programme with an implementation timetable to the Minister for the Environment and Local Government in July 2000.
- **Japan**: the Construction Recycle Law outlines recycling targets as a basic policy. It requires clients and contractors to produce agreements which obligate sorting and recycling activities for particular projects and materials. The projects concerned are construction sites with an area of around 500 m^2 or above, demolition sites with an area of around 80 m^2 or above, or with a construction contract cost equal to or above 100 million Japanese Yen (6 million Hong Kong dollars or 1.26 million US dollars). The materials concerned include concrete, wood and asphalt concrete. The main contractors are required to prepare a demolition plan before beginning activities, report the completion of recycling and record the results of recycling to the clients. It should be noted that this system requires high technical skills from contractors who must submit detailed procedures before starting C&D activities. In addition, the government sets a target for the recycling rate of specified construction materials to reach about 95% by 2010 (Japanese Government – Ministry of Land Infrastructure and Transport, 2006).
- **Norway**: new legislation on recycling has been implemented, which requires companies to make a management plan for waste produced during C&D activities. A lack of documentation on deposition results in heavy fines. This has resulted in a dramatic increase of turnover for recycling companies. There is a proposal for deposition tax of about ECU$56 (US$76) per ton for deposition with gas outtake and about ECU$61 (US$83) per ton for deposition without gas outtake).
- **Spain**: The National Plan for C&D Waste developed by Royal Decree 105/2008 (RD105/2008) is a specific piece of state level legislation for waste

production and management (Marrero *et al.*, 2011). After an adaptation period, the legislation is now compulsory for all C&D waste. The legislation objectives are summarised in a waste hierarchy, which runs from the most to the least efficient measures: prevention, reuse, recycling, energy recovery and adequate waste disposal.

- **United Kingdom**: there is a legal requirement on all C&D projects worth over £300 000 to complete a site WMP. More detailed requirements apply in projects worth over £500 000 (Safety Agenda Ltd, 2012). Prosecutions can result in £50 000 fines or on-the-spot penalties, and both companies and individuals can be held liable. This aims to improve efficiency and profitability, reduce fly-tipping, and increase the environmental awareness of workforce and management.
- **Hong Kong**: Several measures have been implemented under Hong Kong government initiatives to reduce waste generation, including:
  - enacting the Waste Disposal Ordinance;
  - issuing a white paper for a comprehensive 10-year plan to reduce construction waste;
  - launching a green manager scheme;
  - issuing a note on the use of recycled aggregate;
  - commissioning a pilot recycling plant to supply recycled aggregate to public projects;
  - issuing a circular on waste management; and
  - issuing charges for waste dumping.

  The government of Hong Kong implemented a variety of regulations and methods, which aimed to minimise waste production. Among them, the mandatory system of implementing the WMP method in all construction projects has received varied feedback during its trial period. Some contractors found the WMP method to be useful and effective. The contractor managers required site staff to carry out monthly reviews, with checks to ensure that operations on-site follow the plan provided (Poon *et al.*, 2004a,b). However, some argued that detailed descriptions of waste management procedures in the WMP method adversely affected companies' productivity. Many construction organisations lack sufficient experience in drafting to use the WMP method. Companies also found difficulties in recycling construction materials on site.
- **China**: rather than implementing global legislation, a framework has been developed by Lu and Yuan (2011) for understanding C&D waste management research, by means of the qualitative social research software package NVivo. It was found that waste generation, reduction and recycling are the three major areas in C&D waste management. The following were identified as critical factors for successful waste management:
  - waste management regulations;
  - waste management systems;

- awareness of C&D waste management;
- low-waste building technologies;
- fewer design changes;
- research and development in waste management; and
- vocational training in waste management (Lu and Yuan, 2010).

Another study carried out in Shenzhen (Wang *et al.*, 2010) identified:

- manpower;
- market for recycled materials;
- waste sortability;
- better management;
- site space; and
- equipment for sorting construction waste

to be the critical success factors for effective on-site sorting of construction waste.

- **France**: Differing strategies for demolition waste management with the aid of multi-criteria analysis were proposed (Roussat *et al.*, 2009). The choice of waste management strategies takes into consideration sustainable development objectives, including economic aspects, environmental consequences and social issues. Research trends in C&D waste management have been explored (Yuan and Shen, 2011) and reveal increasing research interest in this area in recent years. Researchers from developed economies have made significant contributions to research in the discipline. Some developing countries, such as Malaysia and China, have also been effective in promoting C&D waste management research.

2.3 Assessing the effectiveness of WMP methodology

A survey was conducted to examine the effectiveness of existing requirements in implementing the WMP method for all construction projects. The survey was sent to 250 parties including contractors, consultants, developers, governmental departments and environmental professional associations. Seventy-eight were received and the response rate was around 31.2%. However, two of the questionnaires were not properly completed and only 76 were therefore valid. The respondents can be classified into five categories, G1 to G5. Table 2.2 shows the details of G1 to G5 and the distribution of respondents in each group.

Data collected from questionnaires were analysed using the Statistical Package for Social Sciences (SPSS) Version 14.0 for Windows. The mean values of the five groups were derived first. Then the values were tested for concordance between groups and an F-test performed with a demarcation level of significance at 0.05. The test was used to assess any similarity of opinion on the issues of waste management practices between groups.

Table 2.2 Distribution of respondents

Group	Description	Frequency	Percentage
G1	Registered General Building Contractor	34	44.7
G2	Registered Specialist Contractor	22	28.9
G3	Consultant	10	13.2
G4	Governmental Department and Developer	5	6.6
G5	Environmental Professional Association	5	6.6
Total		76	100

To determine the relative ranking of factors, the scores were transformed into indices based on Equation [2.1] (Tam *et al.*, 2000):

$$RII = \Sigma w / AN \qquad [2.1]$$

where w is the weighting given to each factor by the respondent, ranging from 1 to 5, in which 1 is the least important and 5 the most important; A is the highest weight, in this study $A=5$; N the total number of samples; and RII the relative important index.

After the questionnaire responses were received, individual structured interviews were arranged with eight respondents selected from different business sectors: one from a government department, one from a building developer, one from an environmental consultant, two from registered building contractors, two from registered specialised contractors and one from a professional environmental association. The interviews were intended for the gathering of further comments to elaborate upon and interpret the results obtained from the questionnaire and would reduce the limitations arising from the low response rates from G4 and G5.

2.3.1 Attitudes to implementing the WMP method

The results in Table 2.3 show that concrete is the major source of construction waste, corresponding to about 28.9% of total waste. This result is consistent with

Table 2.3 Survey results on major sources of construction waste

Major sources of construction waste	Percentage
Concrete construction	28.9
Steel reinforcement bar	21.1
Formwork	15.8
Temporary hoarding	14.0
Scaffolding	7.9
Material handling	7.0
Finishing	5.3
Total	100.0

the findings of Li's (2002) study. Concrete was found to be the most significant element with about 75% collected from construction sites, 70% from demolition sites, 40% from general civil work and 70% from renovation work. Interviews with governmental staff indicate that recycling concrete waste to be used as recycled aggregate and as new concrete construction materials, are new directions for the construction industry. This offers opportunities to reduce landfill and improve the environment.

In considering the willingness of various parties to minimise waste, results show government to be the most willing at around 78.9% (Table 2.4). A contractor explained in an interview discussion that implementing waste management requires considerable investment in the early stages of projects, including implementing site planning, the use of environmentally friendly materials and the installation of waste management facilities and equipment. These were the main factors affecting his willingness to undertake waste minimisation.

Across five major project areas: cost, time, quality, environmental and safety, most of the respondents pointed out that minimising construction costs is the most important project goal, while environmental concern is the least important factor (Table 2.5). One of the contractors explained that a low tendering price effectively helps them to award projects and that companies increase their profits if they can lower construction costs. A member of a professional environmental association

Table 2.4 Survey result on willingness to minimise waste

Willingness to minimise waste	Contractor	Client	Designer	Government
	Percentage			
Yes	26.3	18.4	21.1	78.9
Neutral	57.9	55.3	47.3	15.8
No	15.8	26.3	31.6	5.3
Total	100.0	100.0	100.0	100.0

Table 2.5 Survey results on project goal of construction projects

Project goal of construction projects	Most important	Least important
	Percentage	
Cost	39.5	5.2
Time	15.8	18.4
Quality	18.4	15.8
Environmental	0.00	47.4
Safety	26.3	13.2
Total	100.0	100.0

Table 2.6 Survey results on reducing waste production in implementing a waste-management-plan method

Responses	Reducing waste production in implementing a WMP method (in %)	Satisfaction on WMP (in %)
Yes	60.5	13.2
Neutral	29.0	39.4
No	10.5	47.4

argued that implementing waste management may help in achieving a long-term cost saving goal, although it will increase the short-time investment cost. A government staff member agreed that the lowest tendering price is commonly considered the most important factor in awarding a project. However, a new trend from government is to require all tendering contractors to provide documentation on waste management awareness, including the implementation of an ISO 14000 environmental management system.

Table 2.6 shows that around 60.5% of the respondents agreed that implementation of the method can be effective in reducing waste. A member of a professional environmental association pointed out that the association had received assistance in implementing the method from construction organisations. It was clear that those construction organisations lack experience in using the WMP method for their projects. Once sufficient experience is gained, their waste levels will be lowered.

Table 2.6 also shows that around half the respondents (47.4%) were dissatisfied with their companies' existing implementation of the method. During interview discussions with a building contractor and a specialised contractor, it was explained that most construction organisations lack experience in implementing the method, and find it too time-consuming to draft their WMP methods. On-site applications also affect implementation, which need to be continuously revised. A member of a professional environmental association recommended periodic auditing of waste generated from various construction activities to help construction organisations in providing early on-site WMP or making modifications to an existing management system. The audits should cover all aspects of waste related activities, including waste generation, storage, recycling, transport and disposal.

2.3.2 Benefits of implementing the WMP method

Implementation of the WMP method delivers several benefits. Pollution prevention, better allocation of resources, better regulatory compliance, evaluation of risks and plans for preventing potential problems can be achieved (Tibor, 1996). Previous studies have identified a number of benefits in implementing the method for construction (Jasch, 2000; Kuhre, 1998; Tam *et al.*, 2006a). A list of benefits are shown in Table 2.7, ranked by relative importance.

The results of F statistics show all benefits to be consistent among the five groups of respondents. It is clear that all groups provided the same arguments for the benefits in implementing the method. The survey results show that 'Propose methods for on-site reuse of materials' and 'Propose methods for reducing waste' are recognised as the most important and require a detailed procedure to show how construction organisations reuse, recycle and reduce construction waste before starting their projects. A registered building contractor pointed out that he designed a detailed methodology in reusing, recycling and reducing major types of construction waste, including concrete, tiles, steel, glass and plastic. He claimed that these methods may be effectively used in planning the use of materials and reducing material consumption. Some local examples in material recycling are also recommended by the Hong Kong Government (Hong Kong Government – Environmental Protection Department, 2006), for example, the use of recycled aggregate as a sub-base material for road construction, of asphalt as aggregate fill and sub-base, and glass as a substitute for sand and aggregate in pipe-bedding materials.

However, 'Propose lists of materials to be reused or recycled' was not considered to be a major benefit in the survey. An environmental consultant pointed out that

Table 2.7 Relative important index for benefits in implementing a WMP method

Benefits in implementing a WMP method	Σw	Relative important index	Ranking
Propose methods for reducing waste	292	0.768	1
Propose methods for on-site reuse of materials	292	0.768	1
Propose methods for on-site waste separation	290	0.763	3
Propose disposal outlets	284	0.747	4
Identify different types of waste	276	0.726	5
Propose methods of dealing with packing materials	262	0.689	6
Develop an organisation structure for waste management	256	0.674	7
Estimate quantities of waste requiring off-site disposal	254	0.668	8
Help implementing trip ticket system	248	0.653	9
Monitor and audit waste management programme	246	0.647	10
Propose methods of processing, storing and disposal of hazardous waste	246	0.647	10
Estimate quantities of identified waste	240	0.632	12
Propose areas for waste storage	222	0.584	13
Propose lists of materials to be reused or recycled	220	0.579	14

although the WMP method can provide a draft list of materials to be reused or recycled before a project starts, it may sometimes show a disparity between the reusable/recyclable proposal and the actual practices of a project. A contractor claimed that most construction organisations try to minimise waste in other ways, including the use of prefabricated building components and reducing the use of wet trades, before reusing or recycling construction waste.

2.3.3 Difficulties in implementing the WMP method

Although there are several benefits in implementing the WMP method, many construction organisations find the implementation difficult. The principal reason is the high investment costs, such as resource input for training courses (Shen and Tam, 2002). Kuhre's (1998) study showed that support for implementation of the method from top management was crucial at the early stage of the process. Tron's (1995) study considered the lack of relevant empirical experience of the method in supporting the development of a practical guideline to be one of the main concerns in its implementation. Other major difficulties include lack of resources and expertise, lack of staff involvement and poor co-ordination between government, industry and business (Chan and Li, 2001). Based on these previous works, nine major difficulties are identified and are shown in Table 2.8 ranked by relative importance.

The F statistics results show 'Low financial incentive' to be significant. The mean score for G4 (governmental departments and developers) is 5.00, which is higher than the overall average score of 4.11, indicating that government departments and developers do consider the low financial incentive in

Table 2.8 Relative importance index for difficulties in implementing a WMP method

Difficulties in implementing a WMP method	Σw	Relative importance index	Ranking
Low financial incentive	312	0.821	1
Increase in overhead cost	294	0.774	2
Complicated subcontracting system	282	0.742	3
Lack of promotion of waste minimisation measures	280	0.737	4
Construction culture and behaviour	280	0.737	4
Lack of well-known effective waste management methods	276	0.726	6
Low disposal cost	264	0.695	7
Lack of proper training and education	258	0.679	8
Competitive market	252	0.663	9

implementing the method. In discussions with one of the developers, it was claimed that most clients still saw reducing cost as their main aim, although the reduction of waste generation is becoming a significant trend in the construction industry. In implementing waste management, clients and developers will usually make a budget part of the project's specification and their contractors can use this information to help in facilitating waste management equipment on site. A developer claimed in interview that investment in implementing waste management may prove a burden for them in improving the environment. However, he also pointed out that large developers may be eager to implement waste management, as it can improve their image and reputation with the public and industry.

'Low financial incentive' and 'Increase in overhead cost' are considered to be the major difficulties in implementing the method. As facilities and equipment need to be provided on site, the investment cost will increase in the short term. These difficulties can only be alleviated in the long term, when waste generation is reduced and savings improved.

Nevertheless, 'Lack of proper training and education' minimally affects companies in implementing the method. A registered specialised contractor explained that many academic professionals, for example, Green Council, Hong Kong Productivity Council and Centre of Environmental Technology, are actively organising environmentally related seminars and conferences. These training programmes can help in improving knowledge of waste management and providing proper training for different levels of employees.

2.3.4 Implementing more effective measures for the WMP method

Previous studies have identified a number of ways of encouraging the implementation of the WMP method in construction activities. Applying environmentally friendly technology on site is one of the most effective measures of improving waste management (Tan *et al.*, 1999). McDonald's (1998) study emphasised the significance of establishing the WMP method during the construction phase. Chen *et al.*'s (2000) study classified four groups of measures against construction pollution; namely, technological, managerial, planning and building material methods. Based on the findings from previous researchers, 15 tools for encouraging the adoption of the method are identified, as shown in Table 2.9 ranked by relative importance.

The F statistics results show all the effective measures identified to be consistent among the five groups of respondents. It is clear that all groups provide the same arguments on effective measures for implementing the method. By using the relative importance index, 'Use of prefabricated building components' is considered to be the most effective measure. Using prefabricated building components instead of wet trade construction can significantly reduce waste

Table 2.9 Relative importance index for effective measures in implementing a WMP method

Effective measures in implementing a WMP method	Σw	Relative importance index	Ranking
Use of prefabricated building components	306	0.805	1
Purchase management	294	0.774	2
Education and training	294	0.774	2
Proper site layout planning	290	0.763	4
On-site waste recycling operation	288	0.758	5
Implementation of environmental management systems	280	0.737	6
High level management commitment	278	0.732	7
Install underground mechanical wheel washing machines	256	0.674	8
Identification of available recycling facilitate	254	0.668	9
On-site sorting of construction and demolition materials	252	0.663	10
Use of metal formwork	244	0.642	11
Central areas for cutting and storage	238	0.626	12
On-site waste conservation	228	0.600	13
Use of information technology on-site	224	0.589	14
Use of non-timber hoarding	222	0.584	15

generation. A building contractor clarified at interview that reducing waste generation is the best option in implementing waste management, reusing and recycling being the only alternative methods for reducing waste on site. 'Purchase management' and 'Education and training' are considered to be the next most effective measures. A building contractor also explained that if an organisation has senior management support in implementing waste management, environmental awareness for their projects can easily be enhanced, thus significantly reducing waste generation.

2.4 Conclusions

Waste management is a burning issue in the Hong Kong construction industry, which is one of the biggest polluting sectors in the region. The WMP method was introduced in Hong Kong in 2003 and initially received negative feedback from the industry. This paper has examined the effectiveness in implementing the WMP method in the construction industry. The questionnaire surveys and structured interview discussions show the government to be the most willing party in waste minimisation. The major problem for other parties is the high investment cost. 'Cost' is still considered to be the most important project area, while 'Environment' is considered the least important. Although costs are increased by implementation of the method, there are benefits including 'Propose methods for on-site reuse of materials' and 'Propose methods for reducing waste'. The use of prefabricated building components was found to be the most effective means of encouraging implementation.

2.5 Acknowledgement

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3
Methods for estimating construction and demolition (C&D) waste

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Abstract: This chapter contains information about the main issues in the estimation of construction waste (CW), for the purpose of promoting waste recovery within the framework of sustainable construction. Several studies and quantisation tables estimating construction and demolition (C&D) waste, both regionally and on a smaller scale and in different types of buildings, are given. The difficulties and importance of obtaining a greater knowledge of CW from the initial conception of the project are discussed. Seventeen examples and case studies highlight the factors affecting the estimation of C&D waste, the improvement achieved in the management scenarios and the benefits of implementation in attaining sustainable buildings.

Key words: estimation; source; composition; quantification; construction and demolition (C&D) waste methods.

3.1 Introduction

This chapter provides an overview of estimating construction and demolition waste (CDW) as good practice in planning and managing construction projects and waste disposal schemes. It discusses ways of estimating CDW from an individual construction project and in estimating the flow of waste for a district or region. It is structured into five main sections. In the first section, key terms are defined. The variables affecting waste generation are then discussed, differentiating between those affected by planning and design prior to starting work, and those affected by building operations carried out on site.

This chapter introduces two main issues: waste classification considered as types and amounts. The section dealing with the composition of C&D waste differentiates between two types: waste generated during construction and waste originating from demolition. It also emphasises the hazardous waste that may be generated. This section also deals with the European Waste List (EWL).

The section on waste quantification differentiates between two areas: the origin of a project and the waste stream within a region. Several studies are shown which deal with these differences and some data, arranged by countries and types of buildings, are shown. Examples and case studies highlight the factors affecting the estimation of C&D waste, the benefits of making an estimate and its implementation in attaining sustainable building. The role of architects, designers and technicians will be fundamental in the design phase, as well as legislative issues.

3.2 Definitions and documents

'C&D waste' is the generic term used to refer to any material produced as waste from construction sites, from the demolition of buildings or structures, or of a combination thereof (Figs 3.1 and 3.2). With regard to the term 'waste', it is important for legislation to clarify two main issues: when a material, substance or product has the status of waste and when it ceases to be waste.

3.1 Construction waste is a heterogeneous mixture composed mainly of soil, packaging waste, debris and a low percentage of hazardous waste.

3.2 There may be other types of waste, such as garbage, furniture, etc.

3.2.1 'Waste status'

Within the legislative framework of the European Union, a substance or object has 'waste status' when the holder discards, intends or is required to discard it (European Parliament, 2008). However, the following cases are not considered as waste:

- Materials, objects or substances intended for reuse.
- Materials, objects or substances obtained as a by-product of a production process and which can be directly used as raw materials in the same or another production process without undergoing prior transformation. These substances have the same characteristics as those obtained by conventional processes and have the status of by-products.
- Defective materials, objects or substances generated in a production process which are returned to it.
- Uncontaminated soil and other natural material excavated in the course of construction activities, where it is certain that the material will be used for the purposes of construction in its natural state on the site from which it was excavated.

3.2.2 'End-of-waste status'

Material attains 'end-of-waste status' when it has undergone recovery, including recycling, and complies with specific criteria for development in accordance with the following conditions:

- The substance or object is commonly used for specific purposes.
- There is a market or demand for such substance or object.
- The substance or object fulfils the technical requirements for the specific purposes and meets existing legislation and standards applicable to products.
- The use of the substance or object will not lead to overall adverse environmental or human health impacts.

This differentiation is significant because while prevention and reuse avoid the generation of waste, preparation for reuse, recycling or other recovery signifies the end of waste status.

3.2.3 Documents that estimate construction and demolition (C&D) waste

A further legal aspect, which must be considered when estimating C&D waste, is the agent required to perform the operation. In Spain, for example, the two documents which constitute the estimate are differentiated according to a National Decree (Spanish Government – Ministry of the Presidency, 2008). The first is called the Waste Management Study (WMS) and is included in the construction project, prior to starting the work. This obligation rests with the 'waste producer' who is defined as: 'Anyone whose activities produce waste (original waste producer) or anyone who carries out pre-processing, mixing or other operations resulting in a change in the nature or composition of this waste.' This study is performed by the technical designer and includes, *inter alia*, an estimate of the amount (expressed in tons and cubic metres) of C&D waste, which will be generated, coded under the EWL (Commission Decision 2000/532/EC and 2001/118/EC). Key aspects of this estimate are the tools and methods used during the design phase, which are adapted to the technologies used in the project. Another legal requirement in making the estimate is the obligation to separate some categories of waste when they exceed certain amounts. In Spain, these categories and their quantities are:

- concrete: 80 tons;
- bricks, tiles and ceramic: 40 tons;
- metal: 2 tons;
- wood: 1 ton;
- glass: 1 ton;
- plastic: 0.5 ton;
- paper and cardboard: 0.5 ton.

However, there are also wastes which have traditionally been separated on site due to the existence of a secondary market (Fig. 3.3a and b).

Once the work is contracted, the 'waste holder' will carry out a Waste Management Plan (WMP) in accordance with the WMS. The 'waste holder' is the

(a)

3.3 Traditionally, construction waste suitable for a secondary market has been separated into work: (a) steels.


(b)

3.3 (Continued) (b) Old ceramic shingles.

waste producer, person or legal entity in possession of the waste. The plan is usually carried out by the builder, who has the main role in setting the amount and nature of the waste that will be generated at each stage of the work and demolition. A key aspect of these estimates is the experience of the construction company in recording its own data according to the construction and auxiliary methods used.

3.3 Sources of construction and demolition (C&D) waste

C&D waste has its origins within the three stages of the life cycle of buildings:

- the construction project, where dimensions, building materials and procedures necessary for construction are specified;
- execution, where materials are supplied and the project is realised according to its purpose;
- partial or total demolition, when the building or part of it reaches the end of its life cycle and the building elements are demolished.

3.3.1 The design phase

During the construction project, the technician designs, locates, describes, quantifies and specifies the different elements used in the building. As a result of these decisions, the execution of the work and the future demolition of the building will inevitably involve the production of waste materials. Some studies have identified poor concepts and decisions in the design stage as the main causes of generation of a substantial amount of construction waste (CW) (Jaillon *et al*., 2008; Osmani *et al.*, 2006; Innes, 2004; Chandrakanthi *et al.*, 2002; Ekanayake and Ofori, 2000; Faniran and Caban, 1998; Bossink and Brouwers, 1996).

3.3.2 The construction phase

The building processes will determine the type and quantity of waste generated throughout.

- During the early stages of the work, waste is mainly generated by the enclosure of the works, the cleaning and clearing of land, the implementation of access and infrastructure and the construction of premises for storage and staff work.
- During the subsequent stage of receiving materials, waste is often generated by product damage in transport, incorrect purchases due to lack of time or lack of provision in the project and the supply of products of inadequate quality.
- Adequate organisation and conditions for protection of building materials against atmospheric elements is important during the storage stage.
- The main sources of waste during the execution stage are: excavated soil, remains of building materials and components resulting from offcuts, breakages and losses, etc.; the remains of packaging and of auxiliary elements resulting from redesign, testing, temporary work, formwork, etc.; debris from handling; demolition and reconstruction due to faulty execution.
- Design changes, project errors or new customer requirements may necessitate elements of demolition and reconstruction.

3.3.3 The demolition phase

At the end of the life of a building, the waste generated will depend primarily on its typology (i.e. residential, industrial), the deconstruction criteria adopted in the project (i.e. dry anchoring systems), the construction procedures and materials used in the building (Figs 3.4 and 3.5) (i.e. concrete, wood or metal framework) and the demolition techniques used (i.e. selective demolition versus traditional demolition).

3.4 Composition of C&D waste

The majority of C&D waste is generally inert (Franklin Associates, 1998) and therefore will not pose an environmental threat as great as that of hazardous waste or typical Municipal Solid Waste (Wang *et al.*, 2004). For this reason, in many regions, it is not controlled in the same way as other sources of waste, with a consequent lack of data and statistics. C&D waste is usually mixed waste, in which the degree of heterogeneity depends on the category of work (demolition or construction).

3.4 Construction wastes vary from one project to another and depend on the construction techniques.

3.5 Losses of wooden formworks due to concrete executed on site.

3.4.1 Demolition waste

Demolition waste (DW) is a product of dismantling at the demolition stage, or of the restoring and repairing of buildings and facilities. It is usually of a stony nature and more homogeneous than CW due to the absence of soil and packaging waste and has a greater volume and weight. Its composition depends on the construction techniques and materials used in the building to be demolished, and falls into four main categories:

1. Non-stony waste (building elements consisting of steel, iron, aluminium, copper, glass, wood, plastic, etc.).
2. Stony waste (concrete, mortar, ceramics, aggregates and mixtures thereof).
3. Hazardous waste (materials containing asbestos, lead, zinc, paints, varnishes, batteries, fluorescent tubes, lubricants, oils, grease, air conditioning facilities, etc.).
4. Others (i.e. organic material).

There is a greater likelihood of hazardous waste in demolition works than in construction. This is mainly due to the absence of past restrictive regulations on the use of certain hazardous materials, such asbestos and lead. In Spain, for example, a WMS of demolition and rehabilitation projects must include an inventory of hazardous waste to ensure separation and specific treatment.

3.4.2 Construction waste

Construction waste (CW) is generated as a result of work executed on buildings from the foundations up, and by civil engineering works such as roads, railways, canals, dams, sports and leisure facilities, ports and airports, etc. Its composition depends on the type of construction work and the techniques used. It falls into five main categories:

1. Soil (sand, clay, stones, mud, etc.) generated from the excavations prior to construction. This waste can be mixed with organic vegetable elements.
2. Packaging waste from building materials (wooden pallets, plastic, cardboard, etc.), which have a lower presence in civil engineering works.
3. Remains of building materials (of a stony nature: concrete, ceramics, aggregates and mixtures thereof; non-stony: steel, iron, aluminium, copper, glass, wood, plastic, asphalt, etc.), which are more homogeneous in civil engineering works.
4. Hazardous waste (contaminated soil and dredging spoil, materials and substances that may include some dangerous features: flammable concrete additives, adhesives, sealants and mastic (flammable, toxic or irritant), tar emulsions (toxic, carcinogenic), asbestos-based materials in the form of breathable fibre (toxic, carcinogenic), wood treated with fungicides, pesticides, etc. (toxic, ecotoxic, flammable), coatings of halogenated flame retardants (ecotoxic, toxic, carcinogenic), equipment with PCBs (ecotoxic, carcinogenic), mercury lighting (toxic, ecotoxic), systems with CFCs, gypsum-based elements (possible source of sulphide in landfills, toxic, flammable), containers for hazardous substances (solvents, paints, adhesives, etc.), and the packaging of contaminated waste likelihood).
5. Others (i.e. organic material).

3.4.3 C&D waste classification

Waste estimates for C&D projects should be classified according to their composition, treatment and selective separation. In the EU, the unified basis for the classification of waste is the EWL (Commission Decision 2000/532/EC and 2001/118/EC). Its two main goals are protection of the environment and the establishment and good functioning of the home market. It is also important for waste statistics and waste transport (waste transport directive) and is valid for all waste (for both disposal and recovery). Packaging waste comes under category number 15 EWL; residues and soils are identified in chapter number 17 EWL. If the waste fulfils at least one criterion for danger, it is marked with a star, *, and considered as hazardous waste. Other studies have developed C&D Waste Materials Check-lists for use in planning the salvage, reuse and recycling of demolition materials (Kincaid *et al*., 1995).

3.5 Quantification of C&D waste studies

Thirty-five percent of the world's solid waste is debris from C&D projects (Construction Materials Recycling Association, 2005; Hendriks and Pietersen, 2000), the majority of which ends up in landfills, uncontrolled sites or in other inappropriate sites. In recent decades, several studies have been conducted to collect data on the composition and amount of C&D waste generated in a geographical area or in the construction of buildings of varied types.

3.5.1 Quantification of C&D waste in regions

Estimates of CW streams within a region are focused on the objectives of forecasted resource demands, the provision of facilities for their management and the targets for reduction, recovery or disposal within a legislative framework. This depends mainly on economic cycles, the degree of industrialisation and the technologies applied in each region, the degree of structural and functional obsolescence of the buildings and the regulations governing the conservation of buildings:

Estimations based on the range of waste generated in each activity

In the United States, a study showed that C&D waste generation based on per capita factors, in much the same way as MSW estimates, lead to estimation rates with more than a 10-fold variation (Yost and Halstead, 1996). Consequently, a methodology based on the financial value of building permits for a variety of construction projects based on a database from the US Census Bureau was proposed.

The National Methodology for the US Environmental Protection Agency and the Industrial Solid Waste Division (Franklin Associates, 1998) was applied with a similar approach across six sectors: dwellings, non-dwelling buildings, dwelling rehabilitations, non-dwelling rehabilitations, dwelling demolitions and non-dwelling demolitions. The statistical data on the number, value and area of each debris-generating activity was multiplied by the range of waste generated in each activity.

The report for the Florida Centre for Solid and Hazardous Waste Management (Reinhart *et al.*, 2003) was also based on financial value to calculate the amount of construction, rehabilitation and demolition activity in Florida. The values obtained for each activity were multiplied by the previously estimated quantities of each type of C&D waste.

A subsequent study estimating waste generated in the construction, rehabilitation and demolition of buildings in Florida was based on the same principle as the National Methodology (Cochran *et al.*, 2007). However, this study used data from more reliable sources and took into account new estimation ranges allowing for waste generation to be estimated according to the construction techniques. Results were obtained for nine types of waste: wood, concrete, block, drywall, asphalt, metal, plastic, ceramic and other debris.

In Spain, the following information has been inter-related to meet the need for collecting data: the index of C&D waste generated by m^2 of C&D of building used by the Instituto de Tecnología de la Construcción de Cataluña (ITEC, Institut of Technological Construction in Cataluña), the Colegios Profesionales de Arquitectos Técnicos (COOAATT, Professional Associations of Architects' Technicians) and the statistics of the Ministerio de Fomento (MF, Ministry of Public Works). The data was gathered from supervised projects specifying the following parameters: the surface to be constructed, rehabilitated or demolished. After analysis of all available information, the use of the indices indicated in Table 3.1 was chosen to establish the amount of C&D waste generated in each type of construction. By combining all the information, it can be estimated that the waste generated in Spain between 2001 and 2005 is as detailed in Table 3.2.

A study carried out in Thailand employed this principle of estimation more generally (Kofoworola and Gheewala, 2009). Information obtained from building permits was subjected to waste generation factors equal to 21.38 kg/m^2 for the

Table 3.1 Weighted average C&D waste generation rates in Spain (kg/m^2)

Type of construction	
New building construction	120.0
Rehabilitation	338.7
Total demolition	1129.0
Partial demolition	903.2

Source: Draft National Integrated Waste Plan 2007–2015 (Spanish Government – Ministry of the Environment, 2007).

Table 3.2 C&DW generated in Spain (2001–2005) by type of construction and civil works) (t)

Type of construction	Years				
	2001	2002	2003	2004	2005
New building construction	10 270 920	10 274 640	11 649 720	13 139 640	14 149 080
Rehabilitation	914 490	865 040	1 006 278	1 010 342	909 748
Total demolition	4 493 420	4 399 713	5 444 038	6 446 590	7 860 098
Partial demolition	1 147 064	1 122 678	1 231 965	1 360 219	1 297 898
Works without license	841 295	833 104	966 600	1 097 840	1 210 841
Civil works	6 543 403	6 479 649	7 518 000	8 538 752	9 417 654
Total C&DW generated	24 210 592	23 974 824	27 816 601	31 593 383	34 845 319

Source: Draft National Integrated Waste Plan 2007–2015 (Spanish Government – Ministry of the Environment, 2007).

construction of dwellings and 18.99 kg/m^2 for the construction of non-dwelling buildings.

Estimations based on material stocks and flows by dynamic model technique

Another means of estimating C&D waste is through material stocks and flows using the dynamic model technique. The quantities of C&D waste expected within a region are studied by the dynamics of collection and the flow of materials in construction activity and future building demand.

In Taiwan, for example, a study estimated the concrete waste that would be generated in the period 1981 to 2011, based on permits issued for the construction and the demolition of buildings up to 1999 (Hsiao *et al*., 2002).

In Canada, simulation models based on the schedule of works activities were used to establish the generation of waste at construction sites in five categories (metals, wood, plasterboard, concrete, others) (Chandrakanthi *et al*., 2002).

In the US, an evaluation of a materials flow analysis was used to estimate the mass of solid waste generated as a result of construction activities, taking into account the consumption of construction materials and the typical waste factors used in construction materials purchasing (Cochran and Townsend, 2010).

Studies based on the dynamic model technique have also been conducted in Europe. In the Netherlands, the collection and flow dynamics of lead from cathode ray tubes within the EU were studied, anticipating needs by simulating the amounts of lead generated up to the year 2031 (Elshkaki *et al*., 2005). This procedure was also used to determine the resource demands of a region and its corresponding waste generation and emissions (Müller, 2006). More specifically, the model was applied to simulate the demand for concrete in dwelling buildings and waste generated in the Netherlands up to 2100. In Norway, a study obtained estimates of ten types of waste, which would be generated in the construction, rehabilitation and demolition of buildings in Trondheim by 2020 in the following categories: concrete, wood, plasterboard, metal, paper, plastic, glass, insulation, asbestos, hazardous and others (Bergsdal *et al*., 2007).

Estimates of C&D waste in waste treatment facilities

Studies have been carried out in waste treatment facilities to determine the composition of waste from C&D work. In Spain, the composition of debris brought to the Autonomous Community of Madrid landfills was found to be as follows:

- bricks, tiles and other ceramics: (54%);
- concrete: 12%;
- garbage: 7%;
- stones: 5%;
- asphalt: 5%;
- sand, gravel and other aggregates: 4%;
- wood: 4%;
- metals: 2.5%;
- plastic: 1.5%;
- glass: 0.5%;
- paper: 0.3%;
- gypsum: 0.2%; and
- other: 4% (Spanish Government – Ministry of the Environment, 2001, 2007).

The volume of CW treated in plants was estimated on the basis of replies to a questionnaire.

3.5.2 Quantification of C&D waste in buildings

The estimates of the types and quantities of waste in buildings have the following objectives: efficient planning of waste management on site, increased awareness and promotion of the reduction, recycling and recovery of waste on site and an estimate of the economic and environmental cost of waste management.

The amount of waste generated at source depends mainly on: construction techniques, design of the components, waste reduction criteria adopted, documentation and technical quality of the project, training and skills of the workers, quality of implementation processes and the optimisation of the materials supplied (containers, packaging, products, materials, etc.).

Estimation of C&D waste on site

One of the first estimates of the quantities of C&D waste was conducted in the Netherlands by studying 184 dwellings in 5 different projects (Bossink and

Brouwers, 1996). C&D waste was classified and weighed in nine categories: debris, bricks, concrete, blocks, tiles, mortar, aggregates, packages and others. It was concluded that, depending on the type of building materials supplied on site, between 1 and 10% by weight becomes waste. Another study estimated C&D waste across nine sites, including two construction sites, three demolition sites, three recycling plants and one landfill, for the purpose of evaluating the prevailing waste recycling practices in Southeast Queensland (Tam *et al*., 2009).

This information is generally handled by construction companies and as there is greater control of C&D waste management on site, the estimated quantities of waste generated are closer to the actual data. In a recent study, six quantification methods for CW generated at construction sites and/or waste audit tools available from the literature are discussed (Masudi *et al*., 2010).

Installing recycling plants next to dumping sites is one of the five major issues for improving waste management and recycling for both profitable and non-profitable materials (Tam and Tam, 2008).

Estimation of C&D waste in construction projects

As construction projects are the origin of waste, they are also the most effective tools for its efficient prevention and management.

Based on data from previous research, several methods for management and quantification have been developed in Europe, such as SMARTWasteTM association of BRE in the UK (SMARTWasteTM, 2010). This software tool helps in the preparation, implementation and reviewing of the Site WMP for Construction Works. It includes a quantification tool that calculates the overall volume of waste across 13 different categories.

Another example is the BEDEC database tool, designed by ITeC (BEDEC, 2010). Estimates of waste are based on results obtained by studying several C&D works, where the volume of waste is obtained by the constructed or demolished area.

Other tools designed for the life-cycle assessment of buildings are useful in the management of C&D waste (Fatta and Moll, 2003). Examples are the Danish Building Research Institute (SBI) designed in 1999 by Peterson and The Environment Agency for England, known as Wisard (waste-integrated systems assessment for recovery and disposal) designed in 1999.

Other methods are based on databases of construction costs. In Massachusetts, a study was carried out in which the authors obtained the waste factor for each building material from the company R.S. Means. This published the costs of construction and assumed losses of 100% of building materials in demolition projects and 10% in construction projects. This made it possible to establish the amount of wood, tile, asphalt, carpets and plasterboard waste (Wang *et al*., 2004).

In Spain, a quantitative method of C&D waste based on the Andalusian Construction Costs Database (BCCA) (Andalusian Government – Dwelling Counselling, 2010) was established with the aim of estimating site waste and planning waste withdrawal within the specifications and budget documentation of the project (Ramirez de Arellano *et al*., 2002). Another method of quantification was developed in parallel and applied to newly-constructed dwelling buildings (Llatas, 2000). The aim was the identification and quantification of the types of waste expected in each element of the building/site work, with the purpose of determining the sources of waste within the construction process. This method was refined by studying 20 residential buildings, redefining the quantising factors (Llatas, 2011).

3.6 Estimate procedures and case studies

The types and amounts of waste generated on site are directly related to the classification characteristics and construction techniques employed in each building; CW will therefore vary between projects. This chapter provides the literature from which tools, methodologies and tables for estimating waste may be obtained prior to starting work. There are usually two procedures for making estimates:

- a quantification procedure to obtain approximate estimates by the use of waste quantisation tables;
- a quantification procedure to obtain specific estimates for each project. This is advisable.

3.6.1 Estimates based on quantisation tables

These tables are mainly provided by construction companies, organisations and associations in the construction sector. The methodology in these cases is as follows:

- **Step 1**: Quantisation tables classified by project type are obtained (demolition, construction, rehabilitation); uses (residential, non-residential: industrial, commercial, etc.); and similar technologies relevant to the project (structure, masonry, etc.).
- **Step 2**: The features of the project are identified: type of project (demolition, construction, rehabilitation); use (residential, non-residential: industrial, commercial, etc.); and the main technologies (generally in relation to structure: metal, concrete or masonry).
- **Step 3**: The surface area of the project is calculated (in m^2).
- **Step 4**: The total waste amount (volume and/or weight) is obtained from the floor area of the project.
- **Step 5**: The waste composition is obtained (amounts by type of waste).

Waste quantification tables are available from the literature and shown in Section 3.8, 'Sources of further information and advice'. This chapter provides guidance tables from which approximate C&D waste data can be obtained (Tables 3.3 to 3.6).

Table 3.3 Weighted average C&D waste generation rates (kg/m^2)

Type of construction	Heavyweight construction: masonry, concrete, etc.		Lightweight construction: precast elements, drywalls, wood frame, etc.	
	Residential	Non-residential	Residential	Non-residential
New building construction	120–140	100–120	20–22	18–20
Rehabilitation	300–400	250–350	90–120	80–90
Demolition	800–1000	1000–1200	500–700	700–800

Table 3.4 Volume average C&D waste generation rates (m^3/m^2)

Type of construction	Heavyweight construction: masonry, concrete, etc.		Lightweight construction: precast elements, drywalls, light frame, etc.	
	Residential	Non-residential	Residential	Non-residential
New building construction	0.12–0.14	0.10–0.12	0.02–0.03	0.02–0.03
Rehabilitation	0.30–0.40	0.25–0.35	0.10–0.15	0.09–0.10
Demolition	0.80–1.00	1.00–1.20	0.50–0.70	0.70–0.80

Table 3.5 Rounded average percentage of waste composition by volume in constructions (%)

Type of waste			Heavyweight construction: masonry, concrete, etc.	Lightweight construction: precast elements, drywalls, light frame, etc.
15	Packaging waste		60–70	30–60
	15 01 01	Paper cardboard pack	2–4	1–4
	15 01 02	Plastic packaging	5–7	2–3
	15 01 03	Wooden packaging	50–55 and 17 02 01 wood*	25–45
	15 01 04	Metallic packaging*	2–3 and 17 04 metals	2–7
	15 01 06	Mixed packaging	<1	<1
17	C&D waste		30–40	40–70
	17 01 01	Concrete	15–20	10–30
	17 01 03	Ceramics-bricks	10–13	
	17 01 07	Mixed concrete ceramics	2–3	
	17 08 02	Drywalls		20–25
	17 09 04	Mixed C&D waste	3–4	10–15
17 05	Soil and stones		Varies	Varies

*Waste fulfils at least one criterion for danger.

Table 3.6 Rounded average percentage of waste composition by volume in demolitions (%)

Type of waste		Residential		Non-residential	
		Masonry	Concrete	Metal	Concrete
17 01 01	Concrete	5–10	40–50	15–20	35–40
17 01 03	Ceramics-blocks mixtures	65–70	20–30	15–20	5–10
17 01 07	Concrete-ceramics	5–10	5–10	35–40	40–45
17 02 01	Wood	1–5	1–5	0.3	0.2
17 02 02	Glass	0.1	0.1	0.2	0.1
17 02 03	Plastics	0.1	0.1	0.8	0.3
17 03 02	Asphalt	0.5	0.5	0.1	4
17 04 01/05	Metals	1–2	2–3	10–15	1–5
	Potentially hazardous*	2–10	2–10	0.6	0.2
17 09 04	Mixed C&D waste			5–10	5–10

*Waste fulfils at least one criterion for danger.

The following case studies can be utilised as examples:

- **Case Study 1**: New building construction – Residential – Heavyweight construction.
  Input data: surface area (m^2): 1220 m^2. Output data: total waste weight (Table 3.3): 117.12 t; total waste volume (Table 3.4): 146.40 m^3; waste composition by volume (Table 3.5): 15 packaging waste: 93.70 m^3; 15 01 01 paper/cardboard packaging.: 4.39 m^3; 15 01 02 plastic packaging: 7.78 m^3; 15 01 03 wooden packaging: 76.06 m^3; 15 01 04 metallic packaging*: 2.63 m^3; 15 01 06 mixed packaging: 2.84 m^3; 17 debris: 52.70 m^3; 17 01 01 concrete: 29.28 m^3; 17 01 03 ceramics/bricks: 20.50 m^3; 17 01 07 mixed. Concrete/ceramics: 1.67 m^3; 17 09 04 mixed C&D waste: 1.25 m^3; 17 05 soil and stones: varies.
- **Case Study 2**: New building construction – Non-residential – Heavyweight construction.
  Input data: surface area (m^2): 970 m^2. Output data: total waste weight (Table 3.3): 85.36 t; total waste volume (Table 3.4): 106.70 m^3; waste composition by volume (Table 3.5): 15 packaging waste: 68.29 m^3; 15 01 01 paper cardboard pack.: 3.20 m^3; 15 01 02 plastic packaging: 5.67 m^3; 15 01 03 wooden packaging: 55.43 m^3; 15 01 04 metallic packaging*: 1.92 m^3; 15 01 06 mixed packaging: 2.07 m^3; 17 debris: 38.41 m^3; 17 01 01 concrete: 21.34 m^3; 17 01 03 ceramics-bricks: 14.94 m^3; 17 01 07 mixed. Concrete ceramics: 1.22 m^3; 17 09 04 mixed C&D waste: 0.91 m^3; 17 05 soil and stones: varies.
- **Case Study 3**: New building construction – Residential – Lightweight construction.
  Input data: surface area (m^2): 1500 m^2. Output data: total waste weight (Table 3.3): 31.50 t; total waste volume (Table 3.4): 39.30 m^3; waste

composition by volume (Table 3.5): 15 packaging waste: 22.36 m^3; 15 01 01 paper cardboard pack: 0.39 m^3; 15 01 02 plastic packaging: 0.79 m^3; 15 01 03 wooden packaging/ 17 02 01 wood*: 20.00 m^3; 15 01 04 metallic packaging*/17 04 metals: 0.79 m^3; 15 01 06 mixed packaging: 0.39 ^{3}; 17 debris: 16.94 m^3; 17 01 01 concrete: 3.93 m^3; 17 08 02 drywalls: 7.86 m^3; 17 09 04 mixed C&D waste: 5.15 m^3; 17 05 soil and stones: varies.

- **Case Study 4**: New building construction – Non-residential – Lightweight construction.
  Input data: surface area (m^2): 1250 m^2. Output data: Total waste weight (Table 3.3): 22.50 t; Total waste volume (Table 3.4): 28.12 m^3; waste composition by volume (Table 3.5): 15 packaging waste: 12.39 m^3; 15 01 01 paper cardboard pack.: 0.28 m^3; 15 01 02 plastic packaging: 0.84 m^3; 15 01 03 wooden packaging/17 02 01 wood*: 8.44 m^3; 15 01 04 metallic packaging*/17 04 metals: 2.81 m^3; 15 01 06 mixed packaging: 0.02 m^3; 17 debris: 15.73 m^3; 17 01 01 concrete: 8.43 m^3; 17 08 02 drywalls: 5.62 m^3; 17 09 04 mixed C&D waste: 1.68 m^3; 17 05 soil and stones: varies.
- **Case Study 5**: Demolition – Residential – Masonry – Heavyweight construction.
  Input data: surface area (m^2): 2500 m^2. Output data: total waste weight (Table 3.3): 2500 t; total waste volume (Table 3.4): 2500 m^3; waste composition by volume (Table 3.6): 17 01 01 concrete: 250 m^3; 17 01 03 ceramics-blocks: 1697 m^3; 17 01 07 mixtures concrete/ceramics: 250 m^3; 17 02 01 wood: 250 m^3; 17 02 02 glass: 2.5 m^3; 17 02 03 plastics: 2.5 m^3; 17 03 02 asphalt: 0.5 m^3; 17 04 01/05 metals: 2.5 m^3; 17 09 04 mixed C&D waste and potentially hazardous*: 45 m^3.
- **Case Study 6**: Demolition – Residential – Concrete – Heavyweight construction.
  Input data: surface area (m^2): 2500 m^2. Output data: total waste weight (Table 3.3): 2500 t; total waste volume (Table 3.4): 2500 m^3; waste composition by volume (Table 3.6): 17 01 01 concrete: 750 m^3; 17 01 03 ceramics-blocks: 1500 m^3; 17 01 07 mixtures concrete/ceramics: 50 m^3; 17 02 01 wood: 150 m^3; 17 02 02 glass: 2.5 m^3; 17 02 03 plastics: 5.0 m^3; 17 03 02 asphalt: 0.5 m^3; 17 04 01/05 metals: 2.5 m^3; 17 09 04 mixed C&D waste and potentially hazardous*: 39.5 m^3.
- **Case Study 7**: Demolition – Non-residential – Concrete – Heavyweight construction.
  Input data: surface area (m^2): 2500 m^2. Output data: total waste weight (Table 3.3): 3000 t; total waste volume (Table 3.4): 3000 m^3; waste composition by volume (Table 3.6): 17 01 01 concrete: 1050 m^3; 17 01 03 ceramics-blocks: 600 m^3; 17 01 07 mixtures concrete/ceramics: 900 m^3; 17 02 01 wood: 30 m^3; 17 02 02 glass: 3 m^3; 17 02 03 plastics: 3 m^3; 17 03 02 asphalt: 45 m^3; 17 04 01/05 metals: 30 m^3; 17 09 04 mixed C&D waste and potentially hazardous*: 339 m^3.

- **Case Study 8**: Demolition – Non-residential – Metal – Heavyweight construction. Input data: surface area (m^2): 2500 m^2. Output data: total waste weight (Table 3.3): 3000 t; total waste volume (Table 3.4): 3000 m^3; waste composition by volume (Table 3.6): 17 01 01 concrete: 600 m^3; 17 01 03 ceramics-blocks: 750 m^3; 17 01 07 mixtures concrete/ceramics: 900 m^3; 17 02 01 wood: 30 m^3; 17 02 02 glass: 3 m^3; 17 02 03 plastics: 6 m^3; 17 03 02 asphalt: 0.5 m^3; 17 04 01/05 metals: 350 m^3; 17 09 04 mixed C&D waste and potentially hazardous*: 360.5 m^3.

Other case studies that can be considered in respect of demolitions are:

- **Case Study 9**: Demolition – Residential – Masonry – Lightweight construction.
- **Case Study 10**: Demolition – Residential – Concrete – Lightweight construction.
- **Case Study 11**: Demolition – Non-residential – Concrete – Lightweight construction.
- **Case Study 12**: Demolition – Non-residential – Metal – Lightweight construction. Input data: surface area (m^2). Output data: total waste weight (Table 3.3). Total waste volume (Table 3.4). Waste composition by volume (Table 3.6).

Waste estimates for rehabilitation are more variable than those for demolition and construction. The waste generated will depend on the type of rehabilitation. However, with the tables provided, the following case studies could be resolved:

- **Case Study 13**: Rehabilitation – Residential – Heavyweight construction.
- **Case Study 14**: Rehabilitation – Non-residential – Heavyweight construction.
- **Case Study 15**: Rehabilitation – Residential – Lightweight construction.
- **Case Study 16**: Rehabilitation – Non-residential – Lightweight construction. Input data: surface area (m^2). Output data: Total waste weight (Table 3.3). Total waste volume (Table 3.4). Waste composition by volume (Table 3.7).

3.6.2 Estimates based on detailed tools

Specific quantification methods may be applied in these cases. These tools are generally based on the measurements and budget documents for a project. The following is a case study based on a model obtaining waste quantification in accordance with the EWL. The procedure has been applied to the project measurement document:

- **Case Study 17**: New building construction – Residential.
  Input data: project measurements. Additional data: specifications (surface, typology, functional aspects, materials, building elements, etc.): footings, reinforced concrete, reinforced beams, brick closings, flat roof, exterior aluminium, etc. Method of quantisation: Waste quantification by building

Table 3.7 Rounded average percentage of waste composition by volume in rehabilitations (%)

Type of waste		Heavyweight construction: masonry, concrete, etc.	Lightweight construction: precast elements, drywalls, light frame, etc.
15	Packaging waste	35–75	30–70
15 01 01	Paper cardboard pack	1–6	2–4
15 01 02	Plastic packaging	3–8	2–5
15 01 03	Wooden packaging	25–45	20–40
15 01 04	Metallic packaging*	5–15	5–20
15 01 06	Mixed packaging	<1	<1
17	C&D waste	25–65	30–70
17 01 01	Concrete	5–10	5–10
17 01 03	Ceramics-bricks	5–15	
17 01 07	Mixed concrete ceramics	10–25	
17 08 02	Drywalls		20–35
17 09 04	Mixed C&D waste	5–15	5–25
17 05	Soil and stones	Varies	Varies

*Waste fulfils at least one criterion for danger.

elements in accordance with the EWL (Llatas, 2011). Output data: CW by building element, by building system and entire building (Table 3.8), coded in accordance with the EWL.

3.6.3 Benefits and management scenarios based on estimates prior to starting work

The construction company will subsequently manage the waste generated on site. Several studies have indicated the importance of including plans for recycling waste materials in the construction project prior to starting work, identifying the types of waste which will be generated, the method of handling, and the recycling and disposal procedures (Batayneh *et al*., 2007). The opportunities for reducing waste and planning the waste separation and selective removal at each phase of the work will support recycling and recovery (Fig. 3.6).

Three scenarios can be identified: 'low management', which is naturally adjusted to market needs; 'minimal management' tailored to the legal requirements of waste management; and 'efficient management', in which there is a greater awareness of the optimisation of waste beyond the statutory requirements:

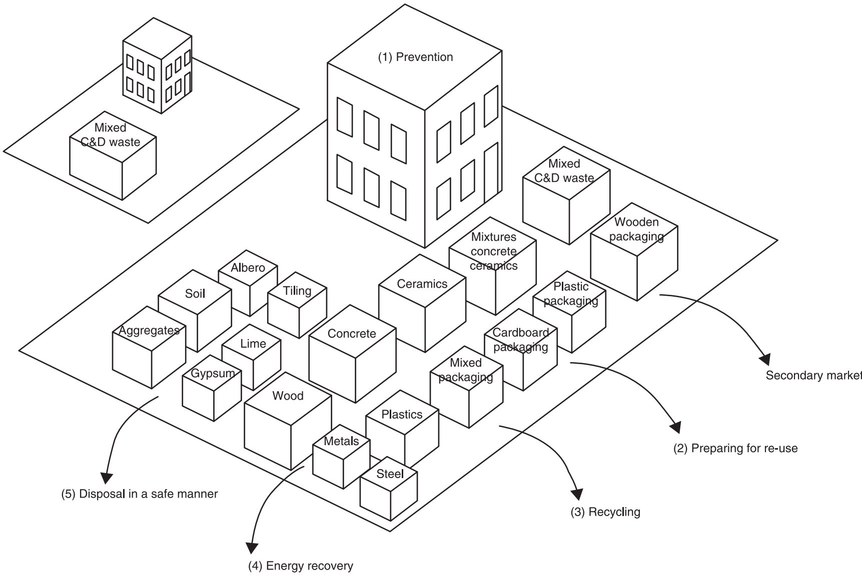
- **Scenario 1**: Low management: The waste is separated into categories which provide an economic benefit through the existence of a secondary market, or the existence of high penalty rates for the disposal of mixed waste. In Case Study 17, for example, the steel and wooden pallets, which are withdrawn by

Table 3.8 Estimates of composition and quantities of construction waste in a new housing project by applying a detailed model for quantification of C&D waste[1]

European Waste List Code[2]	Type of waste	Volume S: 3200 m^2 (m^3)	Volume average rate (m^3/m^2)	Percentage by total waste %	Percentage by group of waste %
15	**Packaging waste**	**256.00**	**0.080**	**16.00**	
15 01 03	Wooden packaging	179.20	0.056	11.20	70.00
15 01 02	Plastic packaging	32.00	0.010	2.00	13.00
15 01 01	Paper and cardboard packaging	28.80	0.009	1.80	11.00
15 01 04	Metallic packaging*	12.80	0.004	0.80	5.00
15 01 06	Mixed packaging	3.20	0.001	0.20	1.00
17	**Remains**	**192.00**	**0.060**	**12.00**	
	Main waste categories (>10% by group of waste)	**160.00**	**0.050**	**10.00**	
17 01 03	Ceramics	54.40	0.017	3.40	28.24
17 01 01	Concrete	41.60	0.013	2.60	20.90
17 01 07	Mixtures of concrete, bricks, tiles and ceramics	25.60	0.008	1.60	12.97
17 01 01	Mortar small vault	22.40	0.007	1.40	12.46
17 01 01	Mortar	16.00	0.005	1.00	8.87
	Secondary waste categories (1–10% by group of waste)	**32.00**	**0.010**	**2.00**	
01 04 09	Aggregates	9.60	0.003	0.60	4.23
17 01 01	Mortar blocks	6.40	0.002	0.40	2.63
17 08 02	Gypsum*	3.20	0.001	0.20	2.20
17 01 01	Terrazzo	3.20	0.001	0.20	1.93
17 02 01	Wood*	3.20	0.001	0.20	1.86
17 09 04	Mixed construction and demolition wastes*	3.20	0.001	0.20	1.23
17 06 04	Fibreglass	3.20	0.001	0.20	1.08

	Minority waste categories (<1% by group of waste)	**0.00**	**0.000**	**0.00**	
10 13 99	Lime	0.00	0.000	0.00	0.44
17 01 01	Mortar beams	0.00	0.000	0.00	0.28
17 06 04	Polystyrene	0.00	0.000	0.00	0.19
17 05 04	Marble	0.00	0.000	0.00	0.16
17 05 04	Pipe clay	0.00	0.000	0.00	0.12
17 02 02	Glass	0.00	0.000	0.00	0.07
17 03 02	Asphalt*	0.00	0.000	0.00	0.07
17 01 01	Cement	0.00	0.000	0.00	0.05
08 01 12	Paintings	0.00	0.000	0.00	0.04
17 02 03	PVC	0.00	0.000	0.00	0.04
17 04 05	Iron and steel	0.00	0.000	0.00	0.04
17 08 02	Scagliola	0.00	0.000	0.00	0.03
08 01 12	Release agent*	0.00	0.000	0.00	0.03
17 02 01	Plant reeds	0.00	0.000	0.00	0.02
17 02 03	Polyethylene	0.00	0.000	0.00	0.01
17 04 01	Copper	0.00	0.000	0.00	0.01
17 03 02	Mastic	0.00	0.000	0.00	0.00
08 01 12	Plasticisers	0.00	0.000	0.00	0.00
08 01 12	Adhesive pastes	0.00	0.000	0.00	0.00
17 02 03	Neoprene	0.00	0.000	0.00	0.00
17 05	**Soil and stones**	**1152.00**	**0.360**	**72.00**	
17 05 04	Soil and stones (earth-works)*	950.40	0.297	59.40	82.51
17 05 04	Soil and stones (site-clearing)	198.40	0.062	12.40	17.25
	Total C&D waste	**1600.00**	**0.500**	**100.00**	

Notes:
[1] Data obtained using a model for the quantification of waste (Llatas, 2011) in a 4-storey housing project.
[2] Waste listed in order of magnitude.
* Potentially hazardous waste.



3.6 Benefits and management scenarios based on estimates prior to commencement of work.

the manufacturers themselves, are separated, so permitting 11.2% of the waste to be recovered.

- **Scenario 2**: Minimal management: This follows the minimal European and Spanish legal requirements. In Case Study 17, for example, the soil is used as fill in neighbouring works, the hazardous waste is separated, as are the following categories: concrete, ceramics, metals, wood, glass, paper and plastic. In this case, reused soil represents 70% of the total C&D waste and recovered waste represents 80% of total waste (excluding soil). Forecasting receptacle size and removal of waste by the respective managers are crucial. Prior knowledge of the waste generated on site is essential and this scenario improves management.
- **Scenario 3**: Efficient management: Estimates of waste generated are made before commencing work, so ensuring the maximum recovery of on-site waste and the avoidance of waste generation. In Case Study 17, for example, 40 categories of waste are identified according to origin, which enables high recovery rates above the minimum legal requirements.

3.7 Future trends

Industry is capable of meeting the social need for the development of sustainable construction. However, the amount of waste in the construction sector will continue to grow without appropriate policies. Becoming a recycling society is a major European challenge and all EU member states will have to take the necessary measures to ensure that by 2020, 70% by weight of non-hazardous C&D waste (with the exclusion of non-contaminated soil and rocks from excavation) are intended for reuse, recycling and material recovery operations. This includes filling operations which use waste to replace other materials.

All EU member states will have to implement control mechanisms to reach this Community target. Leaving the management of on-site waste to chance wastes the potential for prevention and recovery. Higher rates for discharge, regulatory techniques for recycled materials, the provision of legal requirements to separate waste on site and the development of adequate facilities for waste treatment are some of the mechanisms that encourage recycling. However, only prior knowledge of waste and its origin in a project will enable assessment of the possibilities for further on-site management.

One of the key aspects in this task will be the development and promotion of tools, methods and models, which will help designers and architects in estimating CW for each project. The factors these tools should take into account are the direct link between the decisions of the project, with the types and quantities of waste generated on site and the adaptation of tools to the technology of each context and place. The application of a quantification model adapted to the construction technology in Andalusia and interrelated with decisions on the project, allowed estimation during the design stage of the types and quantities of waste listed according to the EWL. Without this operation, both the setting up of a waste management scenario in accordance with minimum legal requirements and the next step in improving management efficiency would not be possible.

The development of detailed procedures for quantifying C&D waste in projects will be advanced by green building, social demand and legal requirements. However, the next step will be the development of an increase in awareness and interest among architects and technical designers in applying these tools to their projects.

3.8 Sources of further information and advice

Much work on the efficient management of C&D waste is being carried out by industrial and environmental associations and companies, with the consequent establishment of work groups and the publication of environmental literature.

Organisations

Examples of these organisations and major studies on the estimation of CW:

- www.epp.eurostat.ec.europa.eu; Statistical Office of the European Communities, Eurostat (2010), European Commission, Environmental Data

Centre on Waste, Waste generation and management, Studies: C&D waste management practices and their economic impacts. Chapter 2: Types of C&D waste and generation of waste by the construction sector, by country, year, and waste category, in kg per inhabitant and tonnes, 2012.

- www.eea.europa.eu; European Environment Agency. Studies on C&D waste by country, by topic, total waste generation and waste generation and management.
- www.eionet.europa.eu; EIONET, European Environment Information and Observation Network. Study: WasteBase. Electronic database with historic information on waste and waste management in Europe, Waste Quantities by Countries.
- www.defra.gov.uk; Department for Environment Food and Rural Affairs. Studies: Methodology for estimating annual waste generation from the Construction, Demolition and Excavation (CD&E) Sectors in England and total waste generation 2008–2010.
- www.wbdg.org; WBDG – The Whole Building Design Guide. Tool: Construction Waste Management Database.
- www.ategrus.org;
- www.buildinggreen.com; Building Green Company. Studies: useful bibliography on CW.
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- www.epa.gov; EPA, US Environment Protection Agency. Studies: State C&D reports/Estimating 2003 Building-Related Construction and Demolition Materials Amounts/Characterization of Building-Related Construction and Demolition Debris in the United States.
- www.anr.state.vt.us; Vermont Agency of Natural Resources.
- www.wastecap.org; WasteCap Resource Solutions.
- www.ihobe.net; Environmental Management Public Company Basque Government. Study: Quantisation tables for waste in demolition projects in methodological guide for the development of selective demolition projects in the Basque Autonomous Community.
- www.csostenible.net; Agenda for sustainable construction, Study: Tables for calculating the volume of waste in a construction.

Tables to estimate C&D waste

The following sources provide tables, case studies, or collections of case studies, on C&D estimation and characterisation.

- Innovative Waste Management (1993), *Construction Materials Recycling Guidebook*, Minneapolis. Useful Section 2.A: Composition of Construction and Remodelling Waste in the Metro Area and Section 2.B: Table Construction and Remodelling Wastes Currently Recycled in Minnesota.

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- Franklin Associates (1998), *Characterisation of Building Related Construction and Demolition Debris in the United States*, EPA-530-R-98-010. US Environmental Protection Agency. Useful for obtaining weight of waste/light weight construction/demolitions, renovation, new construction/residential and non-residential buildings.
- Air Force Center for Engineering and the Environment (2012), *C&D Waste Management Guide*, Available from: http://www.afcee.brooks.af.mil/green/resources/resources.asp. Useful Appendix D – 19 case studies.
- Homestead FL (2012) 'Habitat for humanity', *Construction Waste Management Handbook*, Available from: http://www.recyclecddebris.com/rCDd/Resources/Documents. Useful section: 'Waste composition and quantities waste'.
- Ihobe (2005), Environmental Management Public Company Basque Government. Methodological guide for the development of selective demolition projects in the Basque Autonomous Community. Useful for obtaining volume of waste/heavy weight construction/demolitions/residential and non-residential buildings.

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4

Waste management plants and technology for recycling construction and demolition (C&D) waste: state-of-the-art and future challenges

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Abstract: Waste arising from construction and demolition activities in civil and structural engineering, so-called C&D waste, represents a major share of total waste generation, showing its high importance from both a waste management and a resource efficiency perspective. Whereas re-use of construction products or elements is rarely practised, there is an increasing effort to foster C&D waste recycling in many countries requiring processing of the waste in waste management plants. The current and future framework for waste management plants is analysed with a focus on techno-economic and environmental life cycle, as well as resource policy aspects. Remaining and upcoming challenges with some C&D waste fractions, as well as future changes in quantity and quality of C&D waste supply and recycled aggregate materials demand, are discussed.

Key words: construction and demolition (C&D) waste, waste management plants, recycling rate, life-cycle assessment (LCA), resource policy instruments.

4.1 Introduction

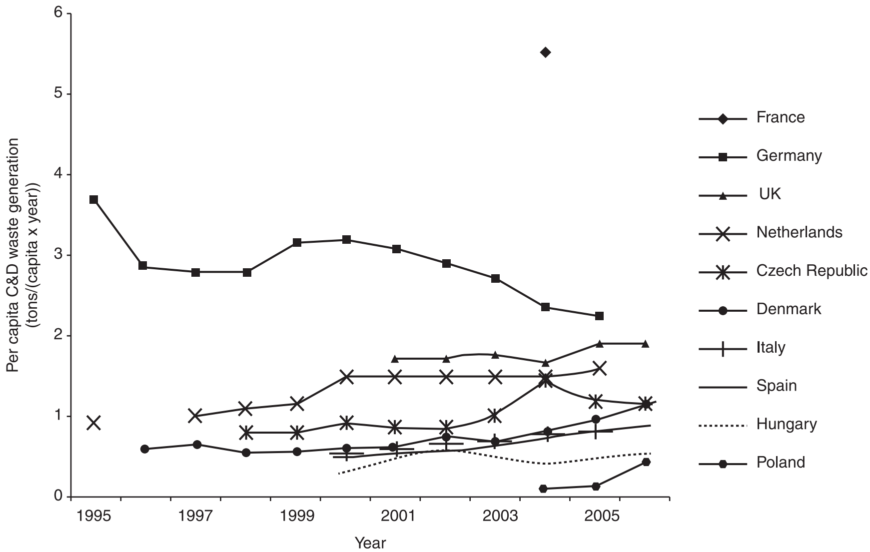
Waste arising from construction and demolition (C&D) activities in both structural and civil engineering, as well as from earthworks, is referred to as C&D waste. Like its sources, C&D waste is a mixture of different components such as concrete, wood, bricks, glass, metals and asphalt (European Commission, 2000, Table 4.1). Generation rates of C&D waste in EU countries vary between 0.2 and 5.5 tonnes per capita, with most new EU member countries showing rates below 1 tonne per capita and countries such as the United Kingdom and Germany around 2 tonnes per capita, subject to variation in construction activities (Fischer and Werge, 2009, Fig. 4.1). Normalisation to value added in the construction sector leads to slightly less variation. However, this introduces additional variations resulting, for example, from differences in labour costs. Overall, in the EU, C&D waste accounts for about one-third of the total waste generated and represents three times the waste from households (EEA, 2012), highlighting its importance in terms of waste management and resource efficiency.

Whereas in former times C&D wastes, such as dimension stones or wooden beams, were re-used as far as possible, for most of the 20th century, disposal of

Table 4.1 European list of wastes (EC, 2000)

17	Construction and demolition wastes (including road construction
17 01	**Concrete, bricks, tiles, ceramics, and gypsum-based materials**
17 01 01	Concrete
17 01 02	Bricks
17 01 03	Tiles and ceramics
17 01 04	Gypsum-based construction materials
17 01 05	Asbestos-based construction materials
17 02	**Wood, glass and plastic**
17 02 01	Wood
17 02 02	Glass
17 02 03	Plastic
17 03	**Asphalt, tar and tarred products**
17 03 01	Asphalt containing tar
17 03 02	Asphalt not containing tar
17 03 03	Tar and tar products
17 04	**Metals (including their alloys)**
17 04 01	Copper, bronze, brass
17 04 02	Aluminium
17 04 03	Lead
17 04 04	Zinc
17 04 05	Iron and steel
17 04 06	Tin
17 04 07	Mixed metals
17 04 08	Cables
17 05	**Soil and dredging spoil**
17 05 03	Soil and stones containing dangerous substances
17 05 04	Soil and stones other than those mentioned in 17 05 03
17 05 05	Dredging spoil containing dangerous substances
17 05 06	Dredging spoil other than those mentioned in 17 05 05
17 06	**Insulation materials**
17 06 01	Insulation materials containing asbestos
17 06 02	Other insulation materials
17 07	**Mixed construction and demolition waste**
17 07 02	Mixed construction and demolition waste or separated fractions containing dangerous substances
17 07 03	Mixed construction and demolition waste other than those mentioned in 17 07 02

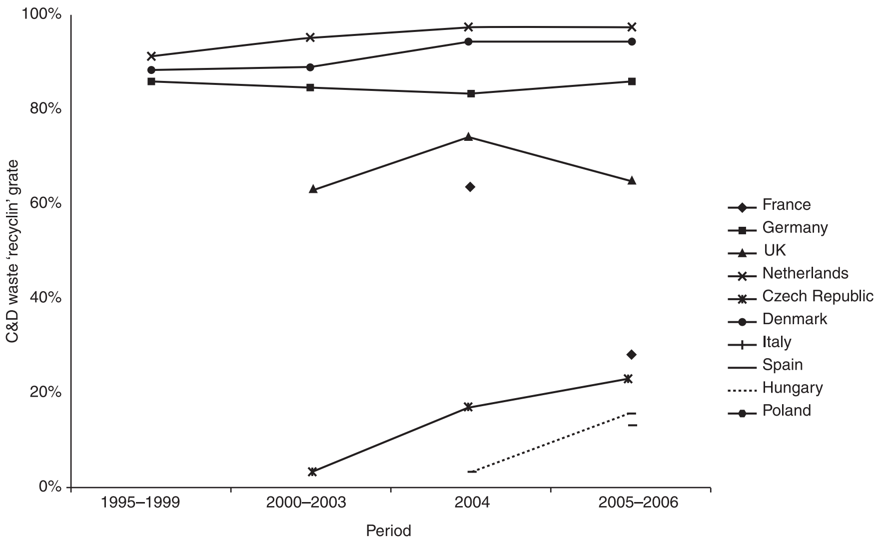
C&D waste dominated for a variety of reasons. First, machinery allowed a fast and inexpensive but also destructive demolition, inhibiting the re-use of building elements. Second, as building materials partly changed, these also became to some extent obsolete. Finally, building materials became less expensive, favouring new production over re-use.



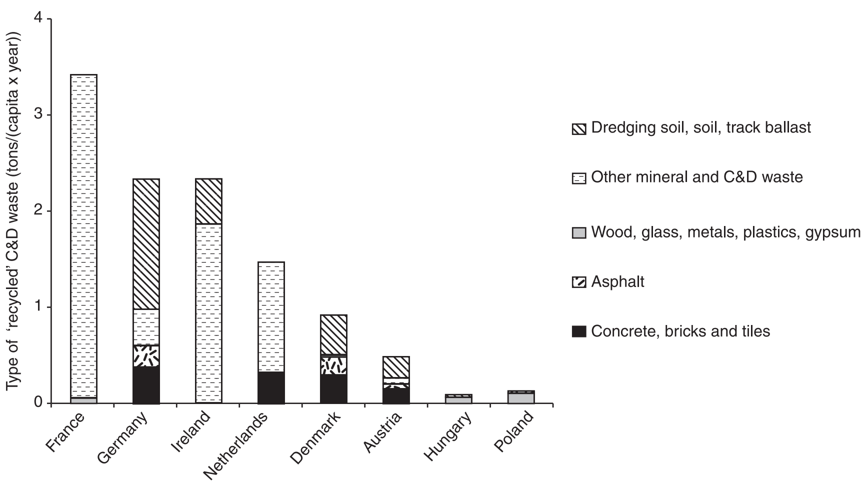
4.1 Per capita C&D waste generation rates in selected EU countries (Fischer and Werge, 2009).

Therefore, C&D waste was dumped together with municipal solid waste or was used due to its mainly inert character directly for landfilling. With the advent of controlled waste management in the second part of the 20th century, inert C&D waste was deposited in building rubble dumps with lower protection requirements than municipal waste dumps. Nowadays, the waste hierarchy (3Rs) orders the waste management options: prevention or reduction as the most preferable, preparing for re-use, recycling, other recovery, and disposal (JRC-IES, 2011). To meet these requirements, generated C&D waste needs to be separately collected and processed in waste management plants to allow recycling of the materials. C&D waste recycling rates vary strongly between EU member countries and range from, for example, around 14% in Spain to 98% in the Netherlands (Tojo and Fischer, 2011; Fig. 4.2). This variation can be partially attributed to differences in reporting between countries.

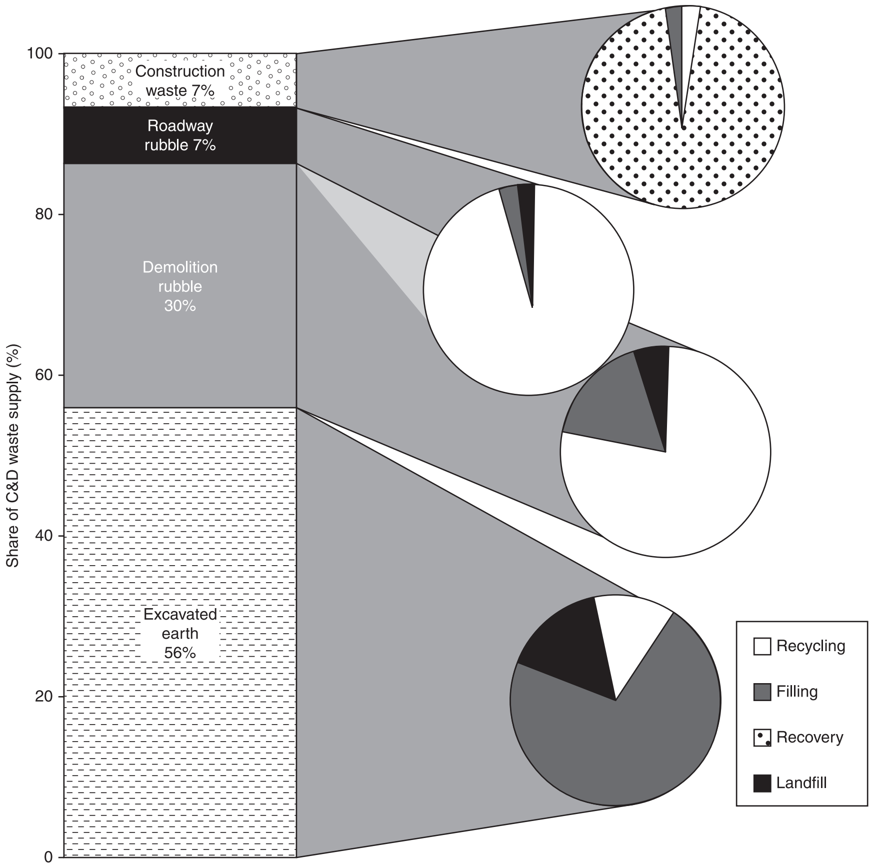
Some countries include dredging soil, soil and track ballast in C&D waste statistics, while others do not (Fig. 4.3). Countries that do include such materials generally show higher C&D waste generation and recycling rates, as these materials, which represent 39 to 74% of the recycled C&D waste streams, are to a higher degree recycled than the ‘core’ C&D waste (Fischer and Werge, 2009; Symonds Group, 1999; Tojo and Fischer, 2011). However, for Germany, the share of (directly) landfilled C&D waste is highest for excavated earth (Fig. 4.4).



4.2 Recycling rates of C&D waste in selected EU countries (Fischer and Werge, 2009).



4.3 Composition of recycled C&D waste in selected EU countries (Fischer and Werge, 2009).



4.4 Composition and fate of C&D waste in Germany in 2008 (Kreislaufwirtschaft Bau, 2011).

In the following sections, the current and future framework for waste management plants is analysed.

4.2 Types of waste management plants

Processing of C&D waste in waste management plants has the following main objectives:

- separation of mixed waste fractions, e.g. removal of wood, plastics and steel from aggregates, to allow reuse, recycling and safe disposal;
- crushing, grinding and sorting to achieve defined grain sizes and thus marketable products;

- removal of contaminants such as asbestos, gypsum, heavy metals and tar from the main waste streams by sorting and sieving (note: deconstruction seeks to minimise the entering of such contaminants in the aggregate waste streams, as later removal is more difficult).

Typical processing steps include two-stage crushing, screening, sieving and removal of impurities and materials such as plastics, iron and steel (Corinaldesi and Moriconi, 2010; Kohler, 1994; Marinkovic *et al.*, 2010; Nunes *et al.*, 2007; Tam, 2008, 2009; ZEBAU, 2006). The main components of a waste management plant comprise (Penzel and Kircher, 1997):

- a bunker with a feeder serving as storage, assuring a constant supply of C&D waste;
- pre-sieving to remove the fine-grained fraction, including soil;
- a crusher, such as an impact or a jaw crusher, to reduce the grain sizes;
- control sieving to separate larger parts needing a further shredding;
- a magnet to remove ferrous metals;
- screening to achieve defined grain size classes;
- mechanical sorting device(s), such as an air knife, an air separator or a jig, to remove contraries such as wood, paper, plastics, etc.;
- manual sorting for further removal of contraries;
- conveyors for dumping;
- machinery for further sieving and mixing, to achieve defined recycled aggregate products.

The selection of different components and their configuration affect the achievable type, quantity and quality of recycled aggregate materials and thus the recycling rate. Besides stationary plants, which also often process primary aggregate material (Blengini and Garbarino, 2010), there are mobile and semi-mobile plants, which are transported to the construction site and recycle the waste on site. If recycled aggregate materials can also be used on site, transport of both C&D waste and recycled aggregate materials can be avoided. However, avoided transport comes at the expense of a lower sorting performance, so that application of (semi-)mobile plants is best for rather homogenous and 'clean' C&D waste, for example concrete waste.

Figures of C&D waste processed in waste management plants in Germany show that in 2002 63% of total concrete C&D waste but only 40% of mixed C&D waste were treated in (semi-)mobile plants (Statistisches Bundesamt, 2004). Schultmann (2005) highlights that contaminants are enriched in the finer fractions, which can be better separated in stationary rather than in mobile plants. Mobile and semi-mobile plants are particularly used in road renovation, as recycled aggregate materials are pure and can then be used in the base course and sub-base of the road. Nevertheless, the share of material going into disposal was twice as high in (semi-)mobile plants compared to stationary plants in 2002 (11.2 and 5.6%, respectively) (Statistisches Bundesamt, 2004).

The choice of the best techniques and their configuration is challenging and depends on technical, legal and economic aspects (Penzel and Kircher, 1997; Symonds Group, 1999):

- C&D waste input:
  - type and composition of C&D waste;
  - expected amounts of different C&D waste streams;
  - maximum sizes of pieces in C&D waste;
- desired output of recycled aggregate materials:
  - types of recycling materials to be separated;
  - grain form, grain sizes and sorting accuracy of recycled aggregates;
  - sorting purity;
- local site conditions:
  - available space;
  - requirements of approval regulations, e.g. dust and noise emissions, groundwater protection;
- economics:
  - charges for accepting different types of C&D waste achievable on the market;
  - prices for selling different types of recycling products achievable on the market;
  - costs for C&D waste processing:
    - (i) fixed costs of different plant types;
    - (ii) operating costs of different plant types depending, in particular on energy and labour costs;
    - (iii) maintenance costs, depending mainly on wear and thus type of C&D waste processed and the plant type, in particular the type of crusher.

The complexity of the waste management plants varies between countries, with Germany generally having more complex plans than, for example, Brazil, Denmark, India and Taiwan (Nunes *et al.*, 2007). There are a variety of reasons for these differences, with legal and economic aspects playing a key role. Accordingly, several papers address economic aspects of C&D waste management recycling, for example Begum *et al.* (2006) in Malaysia, Böhmer *et al.* (2008) in Austria and Germany, Craighill and Powel (1999) in the UK, Duran *et al.* (2006) in Ireland, Huang *et al.* (2002) in Taiwan, Nunes *et al.* (2007) in Brazil, Robinson *et al.* (2004) in the USA, Symonds Group (1999) in several EU member countries, Tam (2008) and Tam *et al.* (2009) in Australia, and Zhao *et al.* (2010) in China.

Symonds Group (1999) analysed the framework conditions for C&D waste recycling in EU member countries and their effects on the level and quality of recycling. Three technical ambition levels for C&D waste management plants were identified:

1. **Level 1**: mobile crusher and sieving plant.
2. **Level 2**: same as level 1, plus metal collector and more complex sorting/sieving.
3. **Level 3**: same as level 2, plus hand-sorting, washing plant and appliances for other waste streams, other than aggregates such as wood.

The three levels were compared to the framework conditions affecting the financial scope for recycling, determined by revenues for accepting C&D wastes and also by revenues from selling recycled aggregate materials.

The level of the charges achievable for accepting C&D wastes depends on the available alternatives for disposing of that waste. If direct landfilling is allowed and landfill prices including taxes are low or fly tipping is common, achievable charges tend to be competitive. However, if there is a ban on direct landfilling or landfill taxes are at a level which makes direct disposal highly unattractive, such as in Denmark with more than 60 EUR per tonne (Fischer *et al.*, 2012), then achievable charges are determined by the competition between recyclers.

The demanded charges for specific C&D wastes depend on the necessary processing steps and thus on processing costs as well as the expected revenues from selling the recycled products. Accordingly, charges for C&D waste cited in Symonds Group (1999) for Belgian recycling plants from the 1990s are, for example, zero EUR per tonne for concrete without reinforcement (processing is simple and thus inexpensive and products can be sold at relatively high prices), 1.25 to 2.5 EUR per tonne for reinforced concrete requiring more processing (the increase in processing costs are apparently higher than the revenues from scrap) and 2.5 to 12.50 EUR per tonne for mixed C&D waste, including wood and plastics requiring a more intense processing, resulting in lower-quality recycling products with lower prices, as well as waste fractions for which disposal costs arise.

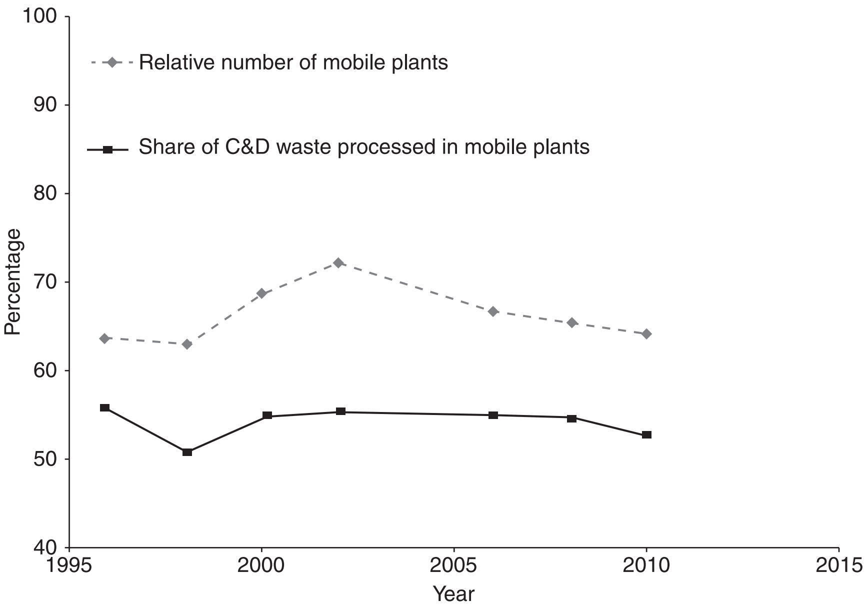
Regarding recycling products, an upper boundary for the achievable price for recycled aggregate materials is the price of virgin or primary ones with similar mechanical, physical and chemical properties. Though there are examples where recycled aggregate materials perform better than primary ones, for example, in road construction where the use of recycled aggregate materials allows in certain cases to reduce the thickness of the asphalt layer, there is generally a deduction in the price of recycled aggregate materials compared to primary materials. For example, Symonds Group (1999) found a prime in the order of about 1.5 EUR per tonne paid for primary aggregate materials used as sub-base materials in road construction in central-southern England.

The prices for competing primary aggregate materials have to cover production costs, transport costs, the profit of the aggregate producer and in some countries an aggregate extraction tax, for example in the UK about 3 EUR per tonne (EEA, 2008). Fixed and operating costs mainly depend on applied extraction techniques and thus on the locally available geological deposits, for example, ashore or marine gravel excavation, stone quarrying with rock crushing, etc., as well as labour and energy costs. With aggregates being a low-cost bulky product (average ex-works primary aggregate materials prices range in most European countries between 5 and 10 EUR per tonne (Böhmer *et al.*, 2008)), transport costs have, with around 13%, a high share of total costs in the extraction sector in the EU (EEA, 2012). Symonds Group (1999) give price formulas for transport costs in Germany, Spain and UK ranging from 0.075 to 0.09 EUR per t/km, plus a fixed cost term ranging from 1.25 to 1.35 EUR per ton. Accordingly, typical transport distances are below 20 to 25 km.

The high share of transport costs on total costs is even more decisive for C&D waste recycling. In contrast to primary aggregates, lower-quality recycling products with a lower value and thus an even higher share of transport costs on total costs need to be transported. In addition, if C&D waste is processed in a stationary plant, transport distances may be higher, as C&D wastes need to be transported to the plant and then the recycled aggregate materials to construction sites. With a typical annual recycling plant capacity of 100 000 tons per year (Hiete *et al.*, 2011b), a capacity utilisation of 80% and an assumed C&D waste generation rate of 2 tons per capita per year (see above), a population of 40 000 within a radius of 10 km is required (ignoring the difference between linear and road distance). This would mean a population density above 125 inhabitants per km^2, which is not met in many parts of the EU.

Furthermore, Duran *et al.* (2006) mention that processing costs per tonne of C&D waste strongly decrease with the capacity of stationary waste management plant (economies of scale), resulting in a trade-off between the reduction of processing costs in larger plants and the reduction of transport costs. One solution is the use of mobile or semi-mobile waste management plants. Though there is continuous progress in optimising mobile or semi-mobile plants technically, in Germany the share of such plants and their share of C&D waste processed varied relatively narrowly between 62 and 72% and 50 and 56%, respectively, in the period 1996 to 2010, without any observable trend (Statistisches Bundesamt, 2004, 2012, Fig. 4.5). However, in the last ten years, the number of stationary plants has steadily increased.

From these economic considerations, a number of conclusions can be drawn. First, C&D waste recycling is competing with direct landfilling (as far as legally allowed) and also with primary aggregate materials (as long as there are no requirements to use a certain share of recycled aggregate materials in construction activities), such that financial resources for C&D waste processing are externally controlled by landfilling and primary aggregate markets. If direct



4.5 Share of mobile C&D waste recycling plants in Germany (Statistisches Bundesamt, 2012).

landfilling is banned or landfilling taxes are high, the situation changes as higher charges can be demanded for accepting C&D wastes. Then the financial resources for C&D waste recycling are determined by competition between recyclers, such that charges for accepting C&D wastes remain reasonable and allow a more advanced recycling and overall higher recycling rates (Symonds Group, 1999).

Second, C&D waste recycling is most attractive in densely populated areas, where supply and demand are close together, resulting in shorter transport distances than for the supply of primary aggregate materials. Accordingly, in less densely populated regions, with a dense supply network of primary aggregate materials, C&D waste recycling often remains, without any policy instruments, economically unattractive.

4.3 Environmental and health aspects

When looking at environmental and health aspects, there are different perspectives: the workers' perspective regarding health and safety (dust, noise, vibrations, accidents, etc.); the neighbourhood perspective important for licensing and site selection, in order to minimize dust and noise nuisance; and the environmental life-cycle perspective for assessing the overall environmental profitability of C&D waste recycling.

Recently, the workers' perspective has gained considerable interest in Germany, as measurements near crushers of mobile plants revealed that limit values for alveolar particulate matter fraction (3 mg/m^3) and inhalable particulate matter fractions were exceeded (Müller, 2000). However, concentrations in the driver's cab were below the limit values. Further measurements by Kummer *et al.* (2010) indicate that 20 to 25% of the total suspended particles have an aerodynamic diameter of less than 10 μm (PM10) and are therefore of health concern for both workers and the neighbourhood.

Measures to reduce dust generation include the enclosure or dedusting of screening units, crushers and conveyor hand-over points, the reduction of drop heights for dumping, water spraying, and the use of electric machines or machines fulfilling recent emission standards, etc. (Department of Environment and Conservation, 2009).

Another important measure is the use of low emission construction engines (DELPHI n.d. for a worldwide overview over emission standards). Vibrations and noise for workers can be reduced by making use of automatic systems for crusher control, vibration decoupled and soundproof driver cabs and control places, as well as water cooled engines and screen cloths with plastics. Selection of an adequate site for a stationary plant with sufficient distance to residential areas may reduce nuisance to residents but may also increase transport if the site is located further away from the urban centre. The siting of C&D waste management plants in urban areas is therefore often a compromise between neighbourhood protection and resource efficiency.

The environmental life-cycle perspective of C&D waste recycling was analysed by a number of papers. As the results are valid only for the analysed system boundaries and settings of the studies, generalisation is difficult. Important influence factors include the composition of C&D waste depending on the demolished building or structure and applied demolition and deconstruction techniques, transport distances, recycling techniques applied and fate of recycled aggregate materials, etc. Environmental impacts arising during the use phase, such as from leaching, are addressed by a few studies only, for example LCA studies (Chowdhury *et al.*, 2010, Clement *et al.*, 2011; Hill *et al.*, 2001; Mroueh *et al.*, 2001) and studies on leaching (Jang and Townsend, 2001; Petkovic *et al.*, 2004). When compared with primary aggregate use, the type and origin of the aggregates are also important. Korre and Durucan (2009) report, for example, that LCA for aggregates from igneous rocks, sedimentary rocks, as well as from sand and gravel deposits of both land and marine origin, show strong differences in the environmental impacts between these products.

A general guidance of how to use LCA to identify the most environmentally sound C&D waste management options is provided by JRC-IES (2011). Lazarevic *et al.* (2012) analysed the difficulties of applying life-cycle thinking to decision making in waste policy. Several papers stress the high importance of environmental impacts from transport (Blengini and Garbarino, 2010; Chong and Hermreck, 2010; Chowdhury *et al.*, 2010; Craighill and Powell, 1999; Korre and Durucan, 2009; Marinkovic *et al.*, 2010; Mroueh *et al.*, 2001). However, Thormark (2001) found that energy demand for transport was less than 2% of total energy savings from material recycling and re-use under different recycling and re-use scenarios for Sweden. However, even more important than transport is aggregate washing. According to Korre and Durucan (2009) washing, which is seldom applied, could account for 60 to 99% of the impact in all impact categories. Among standard processing techniques, crushing has the highest environmental impact (Mroueh *et al.*, 2001).

In the following, results of a few LCA studies are analysed with respect to different recycling options. For road construction, the use of recycled aggregate materials in Germany was identified as less environmentally damaging than the use of primary limestone in all environmental impact categories, with materials processing and transport impacting the most (Gallenkemper *et al.*, 2004). Assumed transport distances ranging from 25 to 120 km used in this study are comparable to those in Chowdhury *et al.* (2010) (50 km for primary and 100 km for recycling aggregates) for the United States and in Marinkovic *et al.* (2010) (100 km for primary and two scenarios with 15 km and 100 km for recycled aggregates) but relatively high compared to Mroueh *et al.* (2001) for Finland. In the latter study, transport distances for crushed rock and crushed cement replacing the rock material were set equally to 10 km.

Mroueh *et al.* (2001) identified environmental benefits from using recycled aggregates, though a slight reduction in the thickness of the asphalt layer is thought to be more important, as impacts associated with the asphalt layer largely dominate total impact. However, Chowdhury *et al.* (2010) find that recycled aggregates have higher impact with respect to energy demand, acidification potential and greenhouse warming potential (GWP) than primary aggregates resulting mainly from production and notably primary crushing. Only if the ratio of transport distances for primary aggregates versus recycled aggregates is above four, calculated impacts are found to be lower for recycled aggregates.

For buildings, low-quality recycling (downcycling, used in foundations of a building, as base for an access road or a car park replacing primary material) is identified as environmentally advantageous over landfilling in most impact categories in UK case studies (Craighill and Powell, 1999), who also identified re-use as an even better option and argue that on-site recycling using mobile plants resulted in less waste compared to stationary recycling, as mixing of wastes is avoided. The total economic costs determined show that combined internal and external costs (the latter are costs for environmental damages or accidents covered by society) increase from re-use over recycling to landfilling by a factor of four. This is largely driven by changes in internal costs, whereas external costs have a minor share of 5 to 20% of total costs only.

High-quality recycling of concrete waste as recycled aggregate concrete is more common in Asia, in particular Japan, than in Europe (Tam, 2009; Tam *et al.*, 2010). Even this so-called high-quality recycling is in most cases still downcycling, as the produced cement has a lower strength, mainly used for non-structural indoor uses. In a simplified LCA study focusing on resource consumption, cumulative energy demand (CED) and GWP, Weil *et al.* (2003) compared recycled aggregate concrete with concrete made of primary aggregates. One of their results is that the higher cement content for the recycling concrete may considerably reduce the advantages in resource consumption and lead to higher CED and GWP values compared to conventional concrete. Similarly, Marinkovic *et al.* (2010) assumed that a higher cement content is necessary in recycled aggregate cement (an additional 5%) and identified recycled aggregate concrete only as environmentally beneficial if transport distances for recycled aggregates are low compared to those of primary aggregates, for example, below 20 km if primary aggregate transport distances are below 150 km.

4.4 Construction and demolition (C&D) waste management plants in the waste chain: a systems perspective

C&D waste management plants are one chain link within the waste chain ranging from waste generation to re-use, recycling, recovery and disposal, as well as further re-use or recycling loops. As shown above, when analysing the financial scope of C&D waste recycling, a system's perspective on the entire waste chain and the primary aggregate supply chain is necessary to better understand the system's behaviour and assess future developments and possible consequences of interventions.

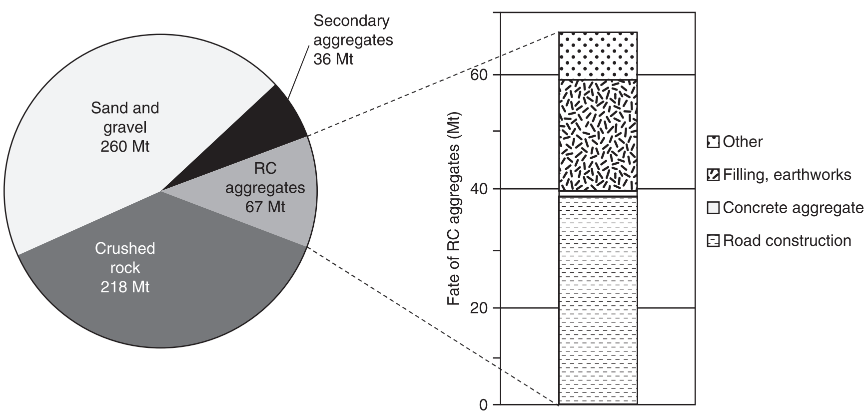
4.4.1 Supply and demand

The metaphor of a C&D waste management plant as a chain link within the waste chain has its strengths but also its limitations. In a chain there is little influence of what happens upstream. However, processing of C&D waste should not be viewed separately from the construction and deconstruction process, as the output of a waste management plant depends highly on the quality of the inputs. Sound planning of construction activities and related waste management activities on construction sites is a prerequisite for high recycling rates (Raess *et al.*, 2006).

On deconstruction sites, gutting including the removal of parts posing problems during recycling such as chimneys contaminated with polycyclic aromatic hydrocarbons (PAH), lead ducts and plaster, selective deconstruction and keeping different waste fractions separate, are essential for high-quality recycling products and can considerably reduce the concentrations of contaminants in the recycled products (Clement *et al.*, 2011; Schultmann, 2005; Seemann *et al.*, 2002). As explained above, the area-specific supply of C&D waste is important, as it influences the possible capacity and complexity of C&D waste management plants and transport distances. In sparsely populated regions, C&D waste recycling is more challenging.

Equally important is the demand for recycled aggregate materials. Consumer acceptance of and trust in recycled products needs further improvement. The prime of 2 EUR per tonne paid for functionally equivalent primary aggregate in the UK (Symonds Group, 1999) highlights this. To increase trust and avoid recycled aggregate materials not being used because of feared financial risks that could arise if the materials were of a low quality, different quality seals for recycled aggregate materials have been developed in Germany and also on a regional level (Böhmer *et al.*, 2008). Acceptance is even lower for structural engineering purposes.

Taking Germany as an example, 56% of recycled aggregates were used in 2008 in road construction (mainly as base course and sub-base) and 30% for earthworks, demonstrating the low demand for recycled aggregate materials in structural engineering (0.8%) (Kreislaufwirtschaft B, 2011, Fig. 4.6). This low demand is partly attributed to the fact, particularly in Germany, that cement producers also own the gravel pits and transport companies and regard recycled aggregate materials as competition to their own products (Schlupeck, 2010). However, with more constructors wanting to have a 'green' building, acceptance in structural engineering could rise if the use of recycled aggregate materials is successfully promoted as a contribution to environmental soundness. This is in contrast to the present day, where the use of recycled aggregate materials is mostly associated with some kind of waste disposal. Green or sustainable building rating systems, such as BREEAM, DGNB or LEED (Hegner, 2010; Hiete *et al.*, 2011a), could help change this attitude by requiring a certain share of recycled materials in new buildings where technically feasible.



4.6 Amount and fate of recycled aggregates in Germany in 2008 (Kreislaufwirtschaft Bau, 2011).

4.4.2 Environmental and resource policy

In many countries, C&D waste recycling is ranked high on the political agenda for various reasons. Sweden and Taiwan, for example, are running short of certain aggregate materials in some regions (EEA, 2008; Huang *et al.*, 2002). In Hong Kong, landfill area is limited (Tam and Tam, 2007). Recent activities of C&D waste recycling focus not only on mass raw materials but also on more valuable materials such as copper (urban mining). In EU member countries, C&D waste recycling gained momentum from the EU Waste Framework Directive (European Parliament and Council, 2008), which requires a minimum of 70% re-use, recycling or other material recovery for C&D waste by 2020 in EU member countries. Accordingly, there is widespread interest in identifying effective and efficient instruments for fostering C&D waste recycling. The instruments applied differ strongly between EU member countries (Böhmer *et al.*, 2008; Tojo and Fischer, 2011).

Landfill tax, which is raised in most EU member countries (Fischer *et al.*, 2012) is considered as a strong driver but on its own not a sufficient driver for recycling and bears some risks (EEA, 2008). Therefore, imposing a landfill tax needs thorough analysis and needs to be complemented by a package of policy instruments such as source separation mandate, a landfill ban or recycling targets. As the example of Germany shows, where there is only a source separation mandate, landfill taxes are not necessary to achieve high recycling rates. The Netherlands, with more than 98% is the country with the highest recycling rate in the EU, imposes a minimum of 10% recycled material in cement and asphalt.

There are several studies analysing different policy instruments (EEA, 2008; Hiete *et al.*, 2011b; Zhao *et al.*, 2011). Hiete *et al.* (2011b) developed a mixed integer linear optimization model, aiming at minimization of total costs in a C&D waste management network. The model takes costs for C&D waste transports, C&D waste processing and disposal, as well as revenues from sale of recycled products, into account. Recycled aggregates are modelled as being in economic competition with natural aggregates. C&D waste generation and demand are calculated for time steps 2010 and 2050 based on literature data (Schiller and Deilmann, 2010). Exemplary application of the model for a case-study region in southwest Germany reveals the high importance of matching C&D waste supply and recycled aggregate demand on a local scale, as transport is both economically and ecologically expensive. Disposal taxes proved to be a cost-effective political instrument to increase the recycling rates in the model, but apparently are not able to foster high-quality recycling.

Whereas C&D waste recycling is generally thought to be beneficial, both for resource conservation and the environment, discussions regarding the upcoming German Recycling Decree (Susset and Grathwohl, 2011) reveal a trade-off between a high C&D waste recycling rate and groundwater protection. Stricter limit concentrations for contaminants such as heavy metals, chloride, sulphate and selected organic pollutants in recycled aggregate materials and higher requirements in monitoring negatively influence the recycling possibilities in civil engineering (Weil *et al.*, 2006). However, stricter limit values might also stimulate activities to reduce contaminant concentrations, such as by a more selective deconstruction and more thorough processing, for example in stationary plants allowing enrichment of the contaminant load in the fines (Schultmann, 2005). However, the latter would be at the expense of additional environmental impacts for the additional processing (Weil *et al.*, 2006).

There is also a trade-off between lower contaminant concentrations and recycling rates. Clement *et al.* (2011) analysed different options for the demolition and recycling of a two-family dwelling in Austria, differing in the level of selective deconstruction with removal and disposal of contaminated fractions. They clearly demonstrate that a further reduction of contaminants in recycled aggregates is possible, but at higher costs and at the expense of a higher share of products arising in the waste management plant for disposal.

4.5 Conclusions and future trends

C&D waste represents a high share of total waste generated in many countries and has therefore gained increasing interest from both a waste management and a resource efficiency perspective. Recycling rates strongly vary between countries and in most countries downcycling prevails, mainly in the use of recycled aggregate materials as sub-base in road construction and for filling. To produce recycled aggregate materials of a defined and high quality, the C&D waste needs to be processed in waste management plants, which in the case of C&D waste are in direct competition with other waste management options such as landfilling and in the case of recycled aggregate products with primary aggregate production. Consequently, many countries have developed a mix of policy instruments such as landfill ban, landfill taxes, source separation mandate, etc. to increase recycling rates.

Acceptance of recycled aggregate materials by the consumer remains critical. The trend towards green buildings and the wish to receive a certificate for this might improve the situation in the future, if the use of recycled materials becomes a criterion in such building rating systems. Though C&D waste recycling is in most cases environmentally beneficial, environmental life-cycle analyses are necessary to analyse a given situation. Transport distances, type of processing and primary material replaced, and the environmental impacts for its production play key roles. Stricter limit values for contaminant concentrations in recycled aggregates may negatively affect recycling rates, showing a trade-off between groundwater and soil protection on one side and resource conservation on the other.

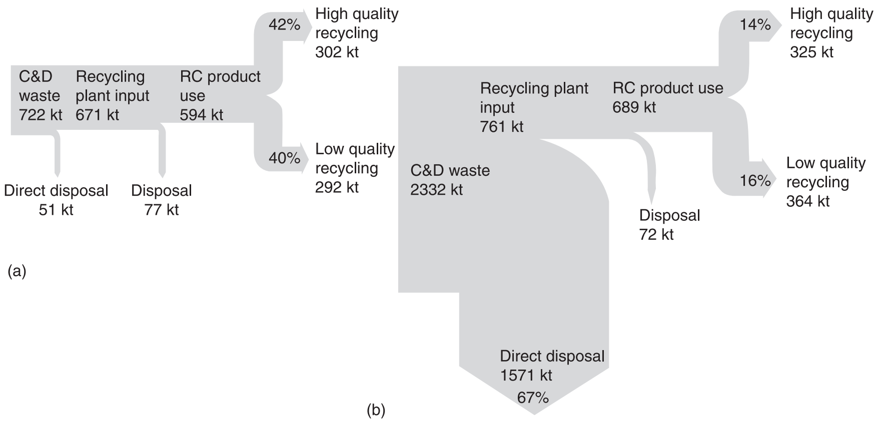
A more thorough deconstruction and a better waste processing aiming at reducing the contaminant load can reduce this problem to some extent and may clear the way for high-quality recycling. High-quality recycling, such as the use of aggregate in concrete is possible and practised in some countries, but might not in all cases be environmentally more sound than low-quality uses, as the often necessary additional processing results in additional environmental impacts as the same amount of primary aggregates is saved.

4.5.1 Future trends and challenges

Though C&D waste recycling rates are around 95%, and higher in some countries like the Netherlands (Tojo and Fischer, 2011), a number of challenges remain and further ones emerge. First, downcycling still prevails in C&D waste recycling as a high share of recycled aggregate materials is used in road construction and for filling and replaces primary aggregates there. As long as primary material is replaced, further processing of C&D waste to allow high-quality recycling is therefore not purposeful, as all processing is associated with additional costs and additional environmental impacts. More difficult to answer is the level at which contaminants should be removed during deconstruction and the recycling process, as this is done at the expense of a higher share of products from C&D waste management which have to be landfilled (Clement *et al.*, 2011). However, in many developed countries, such as Switzerland, infrastructure becomes saturated and populations start to shrink. This results in less new construction but more deconstruction and hence C&D waste generation, for example, for Germany (Deilmann *et al.*, 2009).

Consequently, the supply of recycling aggregate materials might exceed the demand in low-quality applications like road construction and filling in the future, resulting in a strong increase in the amount of C&D waste being landfilled (Hiete *et al.*, 2011b, Fig. 4.7). In such a scenario measures are necessary, allowing deviation of a higher share of recycled aggregate materials into structural engineering (Hiete *et al.*, 2011b; Spoerri *et al.*, 2009). A minimum share of recycled aggregates in concrete is one of those measures (Hiete *et al.*, 2011b), and additional efforts, such as selective deconstruction with re-use of building components, are considered necessary. The high dynamics in construction encountered in countries such as China might lead to similar effects in the future, highlighting the need for dynamic modelling of building stock and associated resource flows (Brattebø *et al.*, 2009; Müller, 2006; Rubli *et al.*, 2004; Zhao *et al.*, 2011).

Detached from this future problem of mismatch between supply and demand, there are still C&D waste fractions that lack a real recycling option, such as fines, porous concrete, masonry with bricks, sand lime bricks and gypsum products. As their shares in C&D waste are increasing, the problem becomes more pressing. Gypsum flows in C&D waste are expected to double by 2030 in Germany, in contrast to the Netherlands and Switzerland where no gypsum recycling is yet in place (Müller *et al.*, 2011a). Therefore, R&D of new recycling techniques (Müller *et al.*, 2010), such as sensor based sorting and R&D for removal of cement stone



4.7 Fate of C&D waste as modelled for the case-study area in SW Germany (modified after Hiete *et al.*, 2011b): (a) base scenario for 2010; (b) base scenario for 2050.

from cement fines (Müller and Sui, 2011), for new products, such as expanded clay such as granules from masonry waste (Müller *et al.*, 2011b) and new applications, such as the use of porous brick aggregates as vegetation substrate, are necessary.

4.6 Sources of further information and advice

The increasing interest in C&D waste management has resulted in a strong increase in related publications in recent years. International scientific journals covering this field include *inter alia* 'Building Research & Information' (Taylor & Francis), 'Construction and Building Materials' (Elsevier), 'Resources Conservation and Recycling' (Elsevier), 'Resources Policy' (Elsevier), 'Waste Management' (Elsevier) and 'Waste Management & Research' (SAGE). Practitioners will find 'Construction & Demolition Recycling' (Recycling Today Media Group (RTMG)) (USA) and 'AT INTERNATIONAL – Mineral Processing' (Bauverlag BV GmbH, in German and English) more interesting.

Books on C&D waste management are rare. Important contributions such as conference proceedings were and are published by RILEM Technical Committee 217-PRE 'Progress of recycling in the built environment' (2005–2012), CIB Task Group TG39 'Deconstruction' and CIB Working Commission W115 'Construction Materials Stewardship'. As a starter for LCA JRC-IES (2011) 'Supporting Environmentally Sound Decisions for Construction and Demolition (C&D) Waste Management' can be recommended.

A number of professional organisations are engaged in C&D waste management such as:

- European Union:
  - International Recycling Federation (www.fir-recycling.com);
  - European Quality Association for Recycling e.V. (www.eqar.info);
- Austria:
  - Österreichischer Baustoff-Recycling Verband e.V. (BRV) (www.brv.at);
- Flanders:
  - Federatie van Producenten van Recycling Granulaten zvw (www.fprg.be);
- Germany:
  - Bundesverband der Deutschen Recycling-Baustoff-Industrie e.V. (BRB) (www.recyclingbaustoffe.de);
  - Dachverband der Baustoff-Recycling-Verbände (www.recycling-bau.de);
- Italy:
  - ANPAR – Associazione Nazionale Produttori Aggregati Riciclati (www.anpar.org);
- The Netherlands:
  - BRBS Recycling (www.brbs.nl);
- Spain:
  - Gremio de Entidades del Reciclaje de Derribos (GERD) (www.gerd.es);
- Switzerland:
  - Abbruch Aushub Recycling Verband Schweiz (ARV) (www.arv.ch);
- USA:
  - Construction Materials Recycling Association (CMRA) (www.cdrecycling.org);
- UK:
  - WRAP (Waste and Resources Action Programme) (www.wrap.org.uk).

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5

Multi-criteria decision-making methods for the optimal location of construction and demolition waste (C&DW) recycling facilities

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Abstract: Multi-Criteria Analysis (MCA) is a powerful tool that has frequently been used to solve environmental decision-making problems, because it integrates different perspectives of the stakeholders involved. This chapter first includes an overview of different decision-making tools, to focus on MCA methods and their main characteristics. Some examples of the application of this tool for locating waste treatment facilities are included. In addition, the required steps to apply the of MCA-based methodology for the selection of adequate construction and demolition waste (C&DW) recycling facility locations are discussed. Finally, the applicability of this methodology is demonstrated with a case study in the northern Spanish region of Cantabria, in which the location of a recycling facility in the most populated area was evaluated: the steps followed in this case study are explained to increase understanding of this methodology.

Key words: construction and demolition waste (C&DW), decision making, facilities location, Multi-Criteria Analysis (MCA), strategic planning.

5.1 Introduction

The European Commission (EC) considers construction and demolition waste (C&DW) to be a priority waste stream, due to the large amounts generated and the high potential for re-use and recycling due to the composition of the waste.

Recycling rate of C&DW in Europe varies significantly among countries. Some countries have recycling rates of less than 10%, while others have recycling rates of greater than 90%. According to EC data on recycling rates in European countries in 2006, six countries reported recycling rates that already fulfil the target of the European Directive: Denmark, Estonia, Germany, Ireland, the United Kingdom and the Netherlands. In addition, three countries, Austria, Belgium and Lithuania, reported recycling rates between 60 and 70% and four countries, France, Latvia, Luxembourg and Slovenia, reported recycling rates between 40 and 60%. The remaining countries had recycling rates below 40%, although no data were available to estimate the recycling rates of Bulgaria, Italy, Malta, Romania, Slovak Republic and Sweden. The EC estimated a recycling rate average of 46% in

Europe, but it is a broad estimation with a high uncertainty, due to the lack of data for some countries (EC DG ENV, 2011).

Specific legislation for C&DW has been developed in Europe such as the Commission Directive 2008/98/EC on waste, which stresses the need to quantify the waste stream and to improve the material recovery efficiency of C&DW in the European Union. According to this Directive, a target of 70% re-use, recycling and material recovery of C&DW should be achieved by 2020. For this reason, a network of facilities dense enough for managing the waste produced should be properly located.

First, a preliminary evaluation of the economic feasibility of recycled C&DW is necessary, which can consist of a preliminary estimation of C&DW generation, a market analysis of recycled materials, and estimation of recycling facility costs, to finally develop an analysis of investments, in which the including payback period, internal rate of return and breakeven point must be included (Zhao *et al.*, 2010). Operating parameters of the recycling facility, such as the capacity of the plant, C&DW input gate fee, concrete aggregate selling price, rejected materials landfill price, percentage of mixed/separated C&DW input and C&DW input mass rate, would influence the return on investment period (Coelho and de Brito, 2013b).

Once the feasibility of recycling C&DW is ensured, the selection of appropriate locations for recycling facilities needs to be studied thoroughly and carefully, as this decision affects long-term profits and costs as well economic and ecological viability of recycled aggregate (Hiete *et al.*, 2011; Queiruga *et al.*, 2008). Coelho and de Brito (2013a) estimated that transportation costs, associated with having to send rejected material to suitable landfills, represents almost 70% of all annual operation costs, for a 350 tonne/h facility.

To begin with the location decision, potential alternatives need to be selected, based on the suitable land available, and appropriate capacity specifications of the facility must be determined (Hesse and Daskin, 1998). Apart from economic aspects, to reduce potential disagreements about the decisions made, decision making should begin with the identification of the stakeholder groups involved in that decision (Fülop, 2011; Robu and Macoveanu, 2009). For this specific case, the stakeholders involved are producers, recyclers, natural aggregates manufacturers or local communities, among others, and also the requirements of municipal, governmental and environmental regulations need to be included (Tuzkaya *et al.*, 2008). The issue of stakeholders is important, as some authors have stressed that most waste management models consider only environmental and economic aspects and very few consider social aspects (Morrisey and Browne, 2004).

In recent years, several decision-making tools have been developed in which environmental, social or economic aspects have been included in the decision-making process. Among the tools developed, Multi-Criteria Analysis (MCA) has frequently been used to solve environmental decision-making problems, because this tool is useful for evaluating different options or alternatives by considering different, often conflicting, criteria. By combining the decision criteria with the importance assigned to each one (weight), a single overall evaluation is required to solve the decision-making problem (Janssen, 2001).

5.2 Decision-making tools: site selection

Site selection is a strategic problem that is regularly encountered in management and marketing studies, as discussed in numerous published articles (Farahani *et al.*, 2010; Smith *et al.*, 2009; ReVelle *et al.*, 2008; ReVelle and Eiselt, 2005). Emphasising the importance of choosing a decision aid method for a site selection problem, the method applied may provide different results with the same data, which may lead to a non-objective identification of the best alternative (Lahdelma *et al.*, 2000).

Facility location is a critical aspect, because decision-making processes must consider more than one target or more than one factor or measure. In this field, it is of utmost importance to include aspects as varied as technical, social, economic, legal, ecological, political, and even cultural aspects in the final decision. To learn how to collect, summarise and facilitate the interpretation of data to make more effective decisions, many decision-making tools have been developed in recent years. Decision-making tools are usually classified into two domains, system engineering models and system assessment tools, although some may be intertwined with each other (Chang *et al.*, 2011; Pires *et al.*, 2011).

Pires *et al.* (2011) proposed a technology hub in which 14 decision models are connected; being considered the Cost–Benefit Analysis (CBA), as a common platform in support of decision making. CBA, Optimization Models (OM), Simulation Models (SM), Forecasting Models (FM) and integrated modelling systems (IMS) are defined as systems engineering models, commonly applied for siting facilities, selecting technologies and comparing management options. However, Scenario Development (SD), Material Flow Analysis (MFA), Life-Cycle Assessment (LCA), Life-Cycle Inventory (LCI), Risk Assessment (RA), Environmental Impact Assessment (EIA), Strategic Environmental Assessment (SEA), Socio-economic Assessment (SoEA) and Sustainable Assessment (SA) are defined as system assessment tools, usually applied to evaluate performance and consider how improvements could be made in a system. Communication among system assessment tools and systems engineering models can lead to defining others tools such as Management Information System (MIS), Decision Support System (DSS) and Expert System (ES).

While recognising the importance of characterising different tools, it is also important to note that most tools are flexible and not always well defined (Finnveden and Moberg, 2005). The type of tool selected also depends on the decision being made and on the decision-makers (EEA, 2003; Zopounidis and

Doumpos, 2002; Guitouni and Martel, 1998). A comprehensive summary of the models applied in decision making related to waste management are analytically presented in some developed reviews (Pires *et al.*, 2011; Morrissey and Browne, 2004). Among the wide variety of decision-making tools, some can be applied satisfactorily to site selection:

- **Cost–Benefit Analysis (CBA)** is a typical tool that enables decision-makers to assess the positive and negative effects of a set of scenarios by translating all impacts into a common measurement, usually monetary. The results are presented in a clear manner, with all impacts summed as one monetary figure; hence this tool provides a good overview of the result of different scenarios. However, there is uncertainty associated with the estimation of the monetary value of several environmental and/or social impacts, and it may not be possible to measure all impacts, direct as well as indirect, in physical units. The assumptions about prices may change during the lifetime of the infrastructure, changing the preferred outcome (Morrissey and Browne 2004; EEA, 2003).
- **Simulation Models (SM)** are used to store and elaborate environmental data to provide conclusions regarding future trends or the evaluation of alternative scenarios. The main characteristics of these models are that they can illustrate the current situation and even estimate the future situation. This tool can provide an integrated insight into a broad range of environmental, economic and socio-cultural aspects of sustainability and even represent a link between economy and environment. SM are also used to evaluate alternative scenarios and to allow the decision-maker to identify the best solution in terms of economic and environmental costs. However, data collection may be difficult and expensive, and this tool requires constant adjustments and modifications (EEA, 2003).
- **Environmental Impact Assessment (EIA)** identifies and assesses the effects of certain public and private projects on the environment; this assessment is then taken into account by the consenting authority during the decision-making process. The locations of the planned project and associated emissions are often known, and an EIA is often used to evaluate alternative locations (Finnveden and Moberg, 2005). The EIA enables the modification of projects to eliminate or mitigate the identified potential environmental impacts. However, EIA only includes environmental impacts, as social and economic impacts are not included in the evaluation.
- **Geographic Information Systems (GIS)** allow researchers to answer different queries and develop environmental impact SM, which can account for the interaction between various forms of environmental impact and geographic features. GIS is mainly a descriptive assessment tool that may be used in collaboration with other models to simulate environmental impacts and project future trends. Data collection is the most time-consuming and

expensive stage during the application of GIS, because these systems require a detailed and wide range of data and their manipulation is very complex (EEA, 2003).

- **Optimization Models (OM)** are applied to identify the best solution among numerous alternatives by considering one or several objectives. These models can address different types of issues, such as single network planning; dynamic, multi-period investment and can be used for selecting size and site facilities.
- **Multi-Criteria Analysis (MCA)** is useful to evaluate different options or alternatives by considering different criteria (Janssen, 2001), to compare scenarios with regards to contradictory objectives. MCA includes the potential for an equal assessment of economic as well as non-economic criteria, conferring a level of flexibility and inclusiveness that economic-based models tend to lack. A mixture of quantitative and qualitative information can be incorporated, but this information must be comparable. These methods produce a general ranking of all solutions (EEA, 2003).

5.3 Multi-Criteria Analysis (MCA): an overview

To select adequate locations for waste treatment facilities, the decision-maker needs to consider different aspects such as distance from waste sources, the competitiveness of the facility, and local municipal acceptability, among others. Also, a key feature of MCA is the involvement of stakeholders in the decision-making process, because it acts as an interactive learning process that allows stakeholders to consider the points of view of other stakeholders (Diakoulaki and Grafakos, 2004). Thus, the application of MCA is proposed.

MCA is a decision model that contains:

- a set of decision options that need to be ranked or scored by the decision-maker;
- a set of criteria, typically measured in different units; and
- a set of performance measures, which are the raw scores for each decision option against each criterion.

The MCA model is represented by an evaluation matrix X of n decision options and m criteria. The raw performance score for decision option i with respect to criterion j is denoted by xi,j. A minimum requirement for the MCA model is at least two criteria and two decision options (n≥2 and m≥2). The importance of each criterion is usually given in a one-dimensional weights vector W containing m weights, where wj denotes the weight assigned to the jth criterion. It is possible for X and W to contain both qualitative (ordinal) and quantitative (cardinal) data (Hajkowic and Collins, 2007).

MCA techniques commonly apply numerical analysis to a performance matrix in two stages, Scoring and Weighting. In the Scoring step, the expected consequences of each option are assigned a numerical score, on a strength of preference scale, for each option for each criterion. In the Weighting step, numerical weights are assigned to define, for each criterion, the relative valuations of a shift between the top and bottom of the chosen scale. Mathematical routines combine these two components to provide an overall assessment of each option being appraised (EEA, 2003).

5.3.1 MCA methods

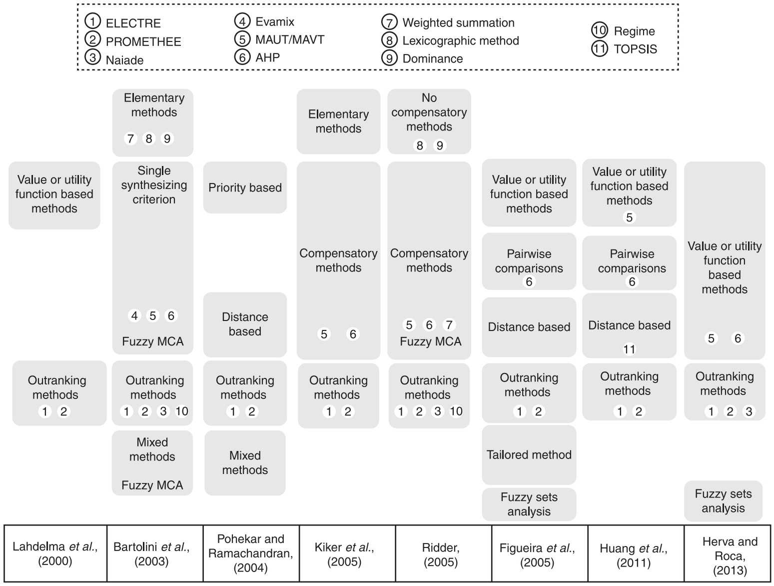
Decision-making processes, where multiple conflicting criteria are involved, can be classified into two types:

1. **Multiple objective problems**, which have an infinite number of feasible alternatives; and
2. **Multiple attribute problems**, which have a finite set of alternatives (Cheng *et al.*, 2002).

MCA methods typically require scores across several dimensions associated with different alternatives, and outcomes and weights relating to tradeoffs across these dimensions. The total value score for an alternative can be calculated in different forms, and thus a wide range of different techniques can be defined. Some techniques rank options, some identify a single optimal alternative, some provide an incomplete ranking, and others differentiate between acceptable and unacceptable alternatives (Kiker *et al.*, 2005). The most common MCA classification categories are priority based, outranking, distance-based and mixed methods, but MCA methods can also be classified as deterministic, stochastic and fuzzy. Depending on the number of decision-makers, the methods can also be classified as single or group decision-making methods (Pohekar and Ramachandran, 2004).

To combine scores for criteria and relevant weights between criteria, the most common method is to calculate a simple weighted average of scores. The use of such weighted averages depends on the assumption of mutual independence of preferences, that is, the judged strength of preference for an option based on one criterion will be independent of its judged strength of preference based on another criterion (EEA, 2003). Compensatory MCA techniques are those in which low scores for one criterion can be compensated by high scores for another criterion. If it is not acceptable to consider trade-offs between criteria, then there are a limited number of non-compensatory MCA techniques available. Review of different classifications of multi-criteria methods and their most relevant characteristics based on different authors are presented in Fig. 5.1.

Among the numerous alternatives proposed, some of the most commonly used MCA methods are Weighted Summation (WS), Electre, PROMETHEE, AHP, Regime (REG) analysis and Fuzzy techniques (Ananda and Herath, 2009; Wang



5.1 Review of different classifications of MCA methods based on different authors.

et al., 2009a; Bartolini *et al.*, 2005). The WS approach is the simplest MCA method, but the results may be affected by scale issues (if indicators are not normalised), and the aggregation is completely compensative among criteria. Consequently, this approach does not explicitly take into account the worst result as a choice criterion and does not account for possible threshold effects in comparing alternatives.

The Electre methods were developed and described by Maystre *et al.* (1994). These methods are based upon a comparison of alternatives through quantitative parameters, which are calculated mostly through concordance or discordance indexes and some type of distillation method.

PROMETHEE was created by Brans (1982) and developed by Brans and Vincke (1985) and Brans and Mareschal (1994). It uses an outranking principle, and the solution of the problem is based on providing the best compromise action to the decision-maker, basically through dominance concepts (Behzadian *et al.*, 2010).

The Analytic Hierarchy Process (Saaty, 1980) is a weighted procedure that is based on different levels of aggregation, allowing the decision-making process to be structured from the components of alternative actions to their final ranking through their effect on the relevant criteria.

The REG methods (Munda *et al.*, 1994; Hinloopen and Nijkamp, 1990; Nijkamp *et al.*, 1990) are designed to use qualitative information as the starting point. The methodology proceeds through the cardinalisation of information on a 1, 2, 3 scale, followed by aggregation by pairwise comparison.

The method selected must be well defined and easy to understand, able to support the necessary number of decision-makers, manage the necessary number of alternatives and criteria, and handle inaccurate or uncertain criteria information (Lahdelma *et al.*, 2000). Each method has strengths and weaknesses; while some methods are better grounded in mathematical theory, others may be easier to implement. The availability to evaluate or support a decision also may act as a constraint on the method of decision analysis (Kiker *et al.*, 2005).

A key method of assessing the robustness of the findings of an MCA study is based in the sensitivity analysis, which determines effects of small changes in the input of a model, by adjusting the weighting factors to represent different situations and by evaluating the sensitivity of the results obtained (Hanan *et al.*, 2012).

In addition, risk in decision-making is generally caused by the uncertainty of input data for the decision-making model and the uncertainty of the model itself. The uncertainty of input data includes uncertainty in data selection and measurement values, whereas uncertainty in the model itself includes model structure and parameters (Wood *et al.*, 2009; Oughton *et al.*, 2008; Refsgaard *et al.*, 2007; Walker *et al.*, 2003).

Decision-makers must select sites that not only perform well according to the current system state but will also continue to be profitable for the lifetime of the facility, even as environmental factors change, populations shift and market trends evolve (Owen and Daskin, 1998). Uncertainties in the estimated quantities of waste and in the assigned capacities of the waste management facilities needs to be assessed, because uncertainties have a large influence on planning decisions in the construction of facilities (Kumar and Nema, 2011). Different uncertainty analysis methods have been developed, such as probabilistic methods, indicator-based methods and fuzzy logic (Chen *et al.*, 2011; Ligmann-Zielinska and Jankowski, 2008; Lahdelma and Salminen, 2001; Zadeh, 1965).

5.3.2 Applications to site selection

Several studies of the application of MCA for the evaluation of the optimal locations for different waste treatment facilities can be found in the literature (Table 5.1). The table shows the most recent examples of the applicability of this tool, alone or in conjunction with Geographical Information System (GIS). In each example, the main objective of each paper, the type of waste involved and the MCA method applied is extracted.

When adequate locations of waste treatment facilities are evaluated by means GIS software, a MCA method is commonly incorporated. It can be observed from Table 5.1 that the most common application of that synergetic tool is for landfill location, mainly for Municipal Solid Waste (MSW). The AHP, Simple additive weighting, Ordered weighted average and WS are the main MCA methods

Table 5.1 Application of MCA in the location of waste treatment facilities

Tools	MCA method	Objective	Type of waste	References
GIS+MCA	Simple additive weighting	Locating treatment facilities and disposal sites	Hazardous waste	Sauri-Riancho *et al.* (2011)
	AHP	Select site for landfill	Hazardous waste	Sharifi *et al.* (2009)
	AHP	Location of an incinerator	MSW	Tavares *et al.* (2011)
	AHP	MSW facility sites	MSW	De Feo and De Gisi (2010)
	AHP	Location of MSW landfill	MSW	Moeinaddini *et al.* (2010); Sener *et al.* (2010); Guiqin *et al.* (2009); Kontos *et al.* (2005); Sumathi *et al.* (2008); Wang *et al.* (2009b); Ersoy and Bulut (2009)
	Simple additive weighting+AHP	Location of MSW landfill	MSW	Sener *et al.* (2006)
	AHP+ordered weighted average	Location of MSW landfill	MSW	Gorsevski *et al.* (2012)
	Weighted summation	Location of MSW landfill	MSW	Geneletti (2010)
	Fuzzy	Location of MSW landfill	MSW	Chang *et al.* (2008)
	Fuzzy+AHP	Location of MSW landfill	MSW	Gemitzi *et al.* (2007)
	Fuzzy AHP	Landfill site selection	–	Nazari *et al.* (2011)

MCA	Simple weighted addition, weighted product, cooperative game theory, TOPSIS and ELECTRE.	Selection of an optimal landfill site	MSW	Cheng *et al.* (2003)
	PROMETHEE and Gaia	To site waste management centres	MSW	Vego *et al.* (2008)
	SMAA-O	Locating a waste treatment facility	MSW	Lahdelma *et al.* (2002)
	Lexicographic minimax problem	To site waste management facilities	MSW	Erkut *et al.* (2008)
	ANP	To site waste management facilities	MSW	Tuzcaya *et al.* (2008)
	ANP	Locating waste treatment plant	MSW	Aragonés *et al.* (2010)
	Fuzzy TOPSIS	Location of disposal site	MSW	Ekmekcioglu *et al.* (2010)
	ELECTRE III	Waste to energy facility	MSW	Perkoulidis *et al.* (2010)
	ELECTRE III	Location of incinerator and landfill	MSW	Bobbio (2002)
	ELECTRE III	Location of incinerator and facility to store	Ashes and other wastes from MSW	Norese (2006)
	ELECTRE III	Location of waste treatment plants	Electrical and electronic waste	Achillas *et al.* (2010)
	PROMETHEE	Location of recycling plants	Electrical and electronic waste	Queiruga *et al.* (2008)
	Weighted summation, Evamix, ELECTRE II, and Regime	Location of recycling facilities	C&DW	Dosal *et al.* (2012)
	ELECTRE III	Optimal location of management facility	C&DW	Banias *et al.* 2010

applied, because they can be easily included in the GIS environment. Advantages obtained in the decision-making process, by including spatial analysis with a GIS tool, can be collapsed for the simplicity of the MCA method applied. When decision making is dealing with a broad area or the facility to be located involves major impacts on the landscape, as occurs with landfills, application of a GIS is advisable, in order to include aspects as groundwater. Also, GIS requires a detailed and wide range of data, and its manipulation is very complex; in addition, the data usually do not reflect the actual conditions of the site, and therefore site verification is mandatory.

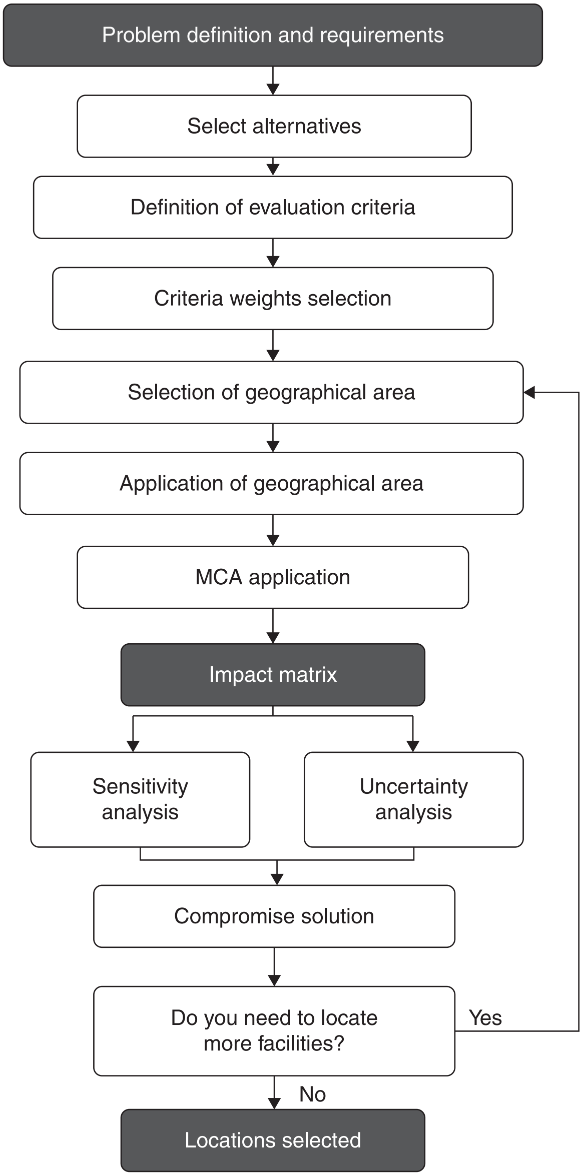
When a MCA method is applied alone, location alternatives must be previously defined. That disadvantage is softened by the fact that more complexity can be included in the model applied, leading to inclusion in the decision-making process of more criteria or even uncertainty. The most common MCA methods applied in this field are the outranking methods (Table 5.1), primarily Electre III and PROMETHEE. These methods encourage more interaction between the decision-maker and the model in the identification of good options and also recognise directly what is sometimes the political reality.

MCA has been widely used in location decisions of waste treatment facilities, as incinerators or recycling plants, more than landfills. In the literature, different examples for the application of MCA to locate recycling facilities have been found, in which different methods have been applied to locate MSW, electrical and electronic waste and C&DW recycling facilities. Banias *et al.* (2010) applied Electre III to locate a C&DW recycling facility, including a sensitivity analysis in the decision process. Also, Dosal *et al.* (2012) applied four different methods, WS, Evamix (EV), Electre II (E2) and REG, to evaluate the influence in the final ranking of the method applied. In this study, an uncertainty analysis is included apart from a sensitivity analysis, to evaluate the robustness of the final ranking obtained. Although studies apply different methods, the general methodology included in both are similar, and the criteria selected for the decision process is also similar.

5.4 MCA-based methodology for site selection of construction and demolition waste (C&DW) recycling facilities

For solving decision-making problems associated with facility location, the usual MCA methodology must be adapted. Figure 5.2 illustrates the main steps of this specific decision-making process.

First, the decision problem needs to be formulated, requirements must be defined, and the main goal of the analysis process must be identified. To establish objectives, it is necessary to establish who the decision-makers are, as well as who may be affected by the decision. A common component of this step is to refer to underlying policy statements (DCLG, 2009). This step involves the decision of



5.2 Methodology based on MCA to select optimal locations of recycling facilities.

the required number of facilities for an adequate management network, and whether it is necessary to divide the territory into management areas. This decision is important in order to diminish the transportation distance of the waste and to avoid impacts on the environment and disturbance to the population. Each geographical area defined has to manage its own waste, and for this reason a management infrastructure should be established in each area.

Once the problem and requirements are defined, location alternatives must be identified. In many situations, the number of potential alternatives is, in principle, infinite, but the decision-making process requires that a finite number of distinct alternatives be formed (Lahdelma *et al.*, 2000). For this reason, it is necessary to identify potential facility locations in each area based on different aspects, such as the existence of infrastructure as quarries, landfills or recycling centres, or even, through a survey on available sites, by including the stakeholder's philosophy in the decision process (Banias *et al.*, 2010).

The third step involves determining different decisional criteria to evaluate the alternatives defined. In the MCA approach, the criteria provide numerical measures for all relevant impacts of different alternatives. It is necessary to identify different impacts and evaluate their relevance. In addition to environmental impacts, there are impacts that relate to the economy, employment, attainability and valuation of different areas, use of energy, services, safety and health (Lahdelma *et al.*, 2000). Therefore, it is advisable to combine social criteria in the final decision to take into account the social dimension of the problem, as well as economic and environmental criteria (De Brucker *et al.*, 2013).

When the alternatives and criteria are defined, the next step is the assessment of the relative importance of each criterion. The use of a single set of criteria weights, elicited from expert opinion, might also be questioned as an option that incorporates a degree of subjectivity into the analysis. A common way to resolve such issues is to use several scenarios of weights with different distributions of the percentage of weight assigned to one criterion, such as waste transport, and the percentage given to the remaining criteria. In addition, varying weights can permit an assessment of the sensitivity of the selected option to these changing weights.

The next step can be ignored if only one recycling plant needs to be located. Otherwise, the evaluation order of the areas previously defined in the first step must be established. The location of the first facility has an important influence on the location of the next one, due to the competition that can occur between facilities. To ensure the competitiveness and feasibility of the proposed investment, a minimum level of incoming raw material needs to be achieved. This situation would be evaluated by estimating the quantity of waste generated in each area. In each one of the geographical areas defined, the feasibility of a new management infrastructure should be determined.

The sixth step involves the application of specific MCA methods to obtain and determine a reasonable rank-order of the alternatives. Numerous techniques for solving a MCA problem are available, leading to the development of a wide range of MCA software. Without relying on the software applied, it is advisable to use more than one method to reach the final decision, in order to avoid possible changes in the results due to the method applied.

In the seventh step, a sensitivity and uncertainty analysis are performed to corroborate the robustness of the solution obtained. The weights of the criteria and the scoring values of the alternatives could contain some uncertainties; the sensitivity of the final ranking of the alternatives to changes in some of the input parameters of the decision model is thus an important question (Bobancu, 2008).

Finally, an iterative process is performed. According to Achillas *et al.* (2010), with the aim of locating more than one recycling facility, the additional criterion of the distance from locations selected in the previous cycle must be evaluated. This criterion must be recalculated every time a new facility is located in a new area.

5.5 A case study: Cantabria, northern Spain

The case study was selected to demonstrate the applicability of the proposed MCA-based methodology to select optimal locations for C&DW recycling facilities (Desal *et al.*, 2012). This methodology could be easily adopted and modified to solve similar problems; additional criteria could be included, depending on the specific requirements of each problem. This case study was applied in Cantabria, a northern Spanish region with 500 000 inhabitants and an area of 5321 km^2, which has experienced a large increase in new construction over the last 10 years. Approximately 400 000 tonnes of C&DW was produced annually (Coronado *et al.*, 2011), most of which was dumped due to the absence of a network of collection and recycling facilities for the recyclable fractions of the waste. Regional regulations for C&DW have now been developed, including the Sectorial Plan for C&DW of Cantabria (BOC, 2010a), which establishes recycling targets for C&DW to be fulfilled in future years, and a regional Decree for C&DW (BOC, 2010b), which attempts to promote adequate management of this waste.

5.5.1 Problem definition and requirements

The Sectorial Plan of C&DW of Cantabria (BOC, 2010a) stresses the need for locating enough recycling facilities in order to meet proposed increasing recycling targets (recycling rate of 30% by 2011 and 65% by 2014) and to decrease transport costs. In a previous study, Coronado *et al.* (2011) evaluated the most sustainable management alternative for C&DW in Cantabria, taking into account environmental, economic and social criteria. The obtained results were a recycling target of 100% of the C&DW in Cantabria by means of four recycling facilities, which must be located, and one transfer station. To locate these facilities by considering economic, social and environmental criteria, a MCA based methodology is proposed.

5.5.2 Selected alternatives

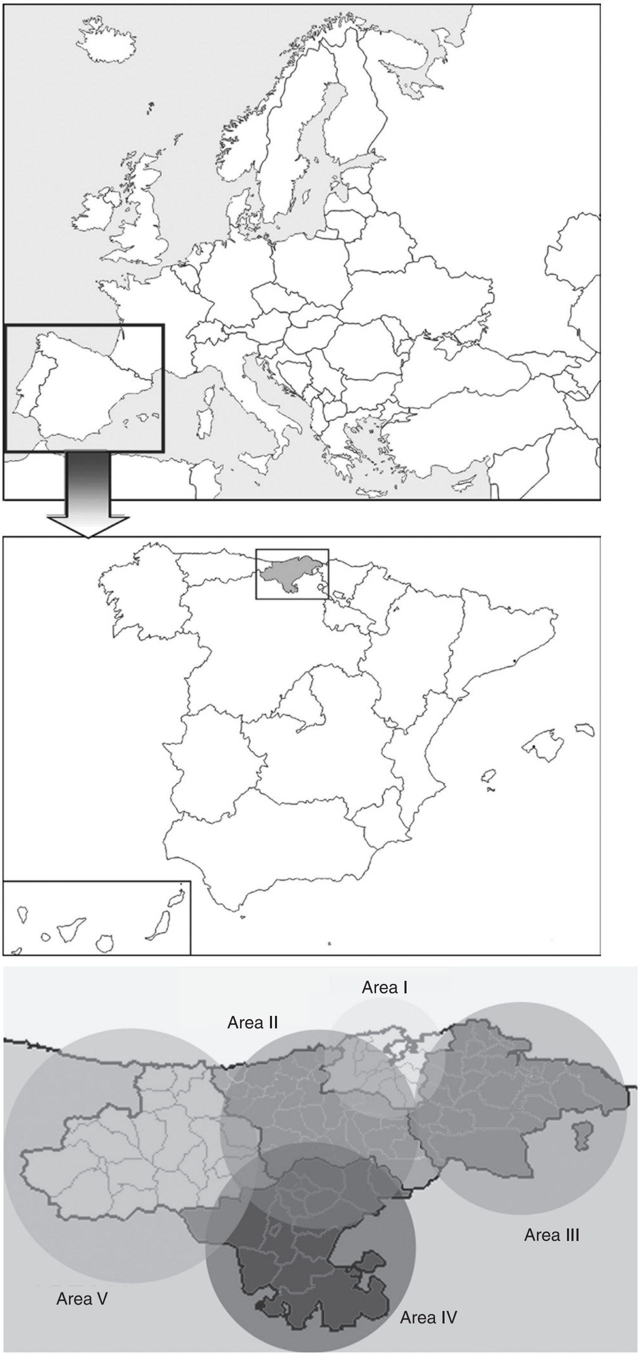
Five potential areas were identified in the case study (Fig. 5.3):

1. **Area I: Santander**, which includes the capital of the region being a densely populated area with a variety of infrastructures;
2. **Area II: Besaya**, which has a high level of industrial activity, contains the second most important city of the region;
3. **Area III: Eastern area**, with a high level of tourism and considerable landscape and environmental value;
4. **Area IV: Southern area**, with a low population density and surrounded by high mountains;
5. **Area V: Western area**, which is surrounded by high mountains, with a lower population and considerable landscape value.

Each zone has different socio-economic characteristics and different waste generation patterns. As mentioned previously, Coronado *et al.* (2011) concluded that, to achieve optimal waste management in Cantabria, four recycling facilities and one transfer station should be located to minimise waste transport and allow each area to manage its own waste. Due to the economic crisis, the generation of C&DW has decreased, and not all of the proposed facilities may now be viable; for this reason, in each of the five geographical areas, the viability of a new management infrastructure should be evaluated. After an exhaustive study of the territory and the land use plans, a set of potential plant locations was identified in each one of the potential areas previously defined, based on the existence of infrastructure in the area such as quarries, landfills or recycling centres. The final alternatives are shown in Table 5.2.

Table 5.2 Potential location alternatives in each geographical area

Area		Alternatives	Area		Alternatives
Area I:	Ia	Astillero	**Area III:**	IIIa	Ampuero
Santander	Ib	Camargo	Eastern	IIIb	Barcena de Cicero
	Ic	Medio Cudeyo	area	IIIc	Voto
	Id	Pielagos		IIId	Castro Urdiales
	Ie	Santa Cruz de Bezana		IIIe	Entrambasaguas
	If	Santander		IIIf	Ramales de la Victoria
	Ig	Villaescusa		IIIg	Santoña
Area II:	IIa	Cabezon de la Sal	**Area IV:**	Iva	Campoo de Enmedio
Besaya	IIb	Corrales de Buelna	Southern	IVb	Reinosa
	IIc	Polanco	area	IVc	San Pedro del Romeral
	IId	Puente Viesgo		IVd	Valdeolea
	IIe	Reocin	**Area V:**	Va	Potes
	IIf	San Felices de Buelna	Western	Vb	Val de San Vicente
	IIg	Torrelavega	area	Vc	Valdaliga



5.3 Geographical areas defined in the case study in Cantabria, northern Spain.

5.5.3 Selected criteria

In this case study, the criteria were selected by taking into account different aspects, which are organised on the basis of the stakeholders. Table 5.3 shows the stakeholder groups and the aspects and criteria considered:

- **(C1) Transport costs (€/ton)**: The distance from C&DW sources is related to the cost assumed by the C&DW generator to manage the waste. To determine the costs due to transport of waste, 'the price list for truck transport', from the monitoring centre of costs from road transport was used (Ministry of Infrastructure, 2010).
- **(C2) Industrialisation ratio (n° ind/km^2)**: Spanish companies can obtain advantages if they are within proximity to suppliers and other companies, and thus agglomeration effects are one of the most important location criteria (Queiruga *et al.*, 2008). This criterion is assessed through the number of industries per square kilometre.
- **(C3) Unemployed population (%)**: The unemployed population is an indicator of both the available workforce and the social acceptance of the development of an industrial facility (Achillas *et al.*, 2010). This criterion is evaluated by the percentage of unemployed population.

Table 5.3 Criteria for the application of MCA to obtain optimal locations for a C&DW recycling facility in Cantabria, northern Spain

Stakeholder Groups	Aspects	Criteria
Producers	Distance from C&DW sources	(C1) Transport costs (€/ton)
	Social acceptability	(C2) Industrialisation ratio (n° ind/km^2)
		(C3) Unemployed population (%)
Society	Local acceptability of municipalities	(C4) Ratio of affected population (inhab/km^2)
		(C5) Level of tourism activity (n° lodging places /km^2)
	Local ecosystem disturbance	(C6) Protected land (%)
National government	Emission due to waste transport	(C7) CO_2 emission (ton CO_2/km covered)
		(C8) C&DW quantities (ton)
	Competitiveness of the facility	(C9) Distance from existing facility (km)
Recyclers		(C10) Distance from inert landfill (km)
	Accessibility of the facility	(C11) Type of road network (km)
	Availability of land	(C12) Vacant land (%)

- **(C4) Ratio of affected population (inhab/km^2)**: Local acceptability of municipalities is assessed by the amount of population that would be affected in each area. This criterion is calculated by the population per square kilometre.
- **(C5) Level of tourism activity (n° lodging places/km^2)**: Recycling facilities have a high visual impact, emission and dust production, among other effects, which could impair the tourism sector. Thus, location alternatives in areas with high tourism activity must be avoided. For this reason, the level of tourism activity is evaluated by the number of lodging places per square kilometre.
- **(C6) Protected land (%)**: This criterion evaluates the ecological effects of the proposed location site. Establishing and operating a C&DW recycling facility has significant ecological effects on the local flora and fauna (Aragones-Beltran *et al.*, 2010). To determine the possible ecological effect, the percentage of protected natural areas is included in the evaluation process.
- **(C7) CO_2 emission due to transport (ton CO_2/km covered)**: The distance from the source of generation to the C&DW recycling facilities contributes to CO_2 emissions. Due to the environmental impacts of CO_2 emissions, those associated with each potential location are evaluated by means of an equation fitted to the available data for CO_2 emissions of various transport modes from the *Forum of Trade and Development*, which took place in Geneva in 2008. This equation considers emissions as a function of the tare weight and the distance covered by the vehicle (g CO_2/ton C&DW*km) (GTDF, 2008).
- **(C8) C&DW quantities (ton)**: To take into account the competiveness of the facility, the quantity of C&DW generated in each area is calculated. This is an indicator of the estimated incoming raw materials, which is the most critical component for the economic viability of the proposed investment (Banias *et al.*, 2010). This criterion evaluates the quantity of C&DW generated in each area by means of an equation fitted from a study by Coronado *et al.* (2011). If the quantity of C&DW generated by the area under study is lower than a certain quantity, the facility is not a valid alternative from an economic point of view, and a transfer station or a mobile plant rather than a fixed plant should be located.
- **(C9) Distance from existing facility**: The existence of a nearby competitor would decrease the quantity of C&DW entering the developed facility or reduce the prices charged for the services provided. Also, the distance from other existing C&DW management facilities plays an important role in the investment's viability (Banias *et al.*, 2010); therefore the distance to another facility is evaluated.
- **(C10) Distance from inert landfill**: The existence of a nearby landfill would increase the viability of the plant, because the C&DW recycling facility

would have a stable recipient for its output (Banias *et al.*, 2010). The distance between the recycling facility and the landfill is evaluated to assess the cost of transporting the non-recyclable waste from the recycling facility to the landfill.

- **(C11) Type of road network (km)**: The road network is likely the most crucial infrastructure for the smooth operation of the recycling facility (Banias *et al.*, 2010). The MCA assesses the availability of suitable roads for waste vehicles (Aragones-Beltran *et al.*, 2010). This criterion evaluates the accessibility of the facility based on the number of kilometres of adequate roads.
- **(C12) Vacant land (%)**: The availability of enough land to build a recycling facility is another aspect that must be considered. This aspect is assessed by the percentage of vacant land as an indicator of the availability of land.

5.5.4 Assignment of weights

Distribution of weights in each scenario was established to assess the influence of transport in the rank ordering. As can be observed from Table 5.4, nine different scenarios of weights were contemplated to assess the sensitivity of the obtained ranking using WS, EV and E2.

The REG method allows preference weights based solely on qualitative judgements so, for this reason, different scenarios was defined by this method:

- **Scenario A**, in which the criterion of CO_2 emission due to waste transport is more important than other criteria;
- **Scenario B**, in which an equal weight distribution is used for all criteria; and
- **Scenario C**, in which the criterion of CO_2 emission due to waste transport is less important than other criteria.

Table 5.4 Scenarios applied in the MCA to obtain optimal locations for a C&DW recycling facility

Scenarios	Weights
Scenario 1	The same weight is given to all the criteria
Scenario 2	100% CO_2–0% others
Scenario 3	90% CO_2–10% others
Scenario 4	75% CO_2–25% others
Scenario 5	60% CO_2–40% others
Scenario 6	50% CO_2–50% others
Scenario 7	40% CO_2–60% others
Scenario 8	25% CO_2–75% others
Scenario 9	0% CO_2–100% others

5.5.5 Selection of geographical area for the location of a recycling facility

Selection of the geographical area in this case study was based on the quantity of C&DW generated in each one, which represents an estimation of the incoming raw materials of the facility. Based on the regional plan of C&DW Management of Castilla-La Mancha (Coronado *et al.*, 2011; Nunes *et al.*, 2007; DOCM, 2005), recycling plants with capacities lower than 50 000 ton/year had both low productivity and selling cash flows. For this reason, areas with an estimated quantity of C&DW generated lower than this value would not be considered as valid alternatives. Figure 5.4 shows the estimated percentage of C&DW generated in each area.

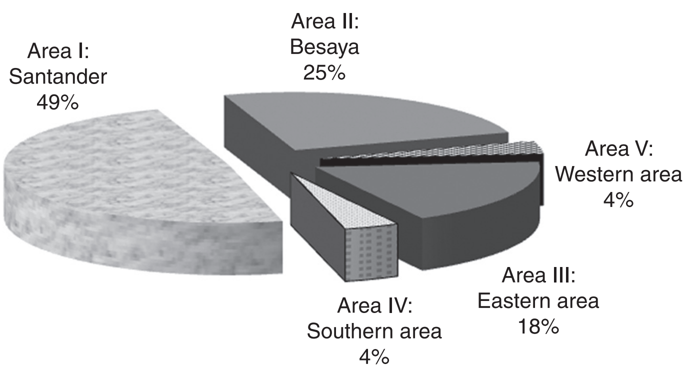
According to Fig. 5.4, Area I: Santander is the area with the highest C&DW generation, with more than 170 000 ton/year. With this quantity, enough incoming raw materials are estimated to be produced and therefore a recycling facility can be located there.

5.5.6 MCA methods application

To evaluate the alternatives selected based on the different calculated criteria, an *impact matrix* has been developed. Table 5.5 shows the estimated impact matrix of the proposed alternatives. This matrix was introduced in the software DEFINITE 3.0 (Janssen *et al.*, 2003), which contains four separate multi-criteria techniques: WS, EV, E2 and REG.

5.5.7 Interpretation of the results

Rankings obtained were calculated based on specific sets of weights, using different methods, as WS, EV and E2 (Fig. 5.5). The figure shows the results

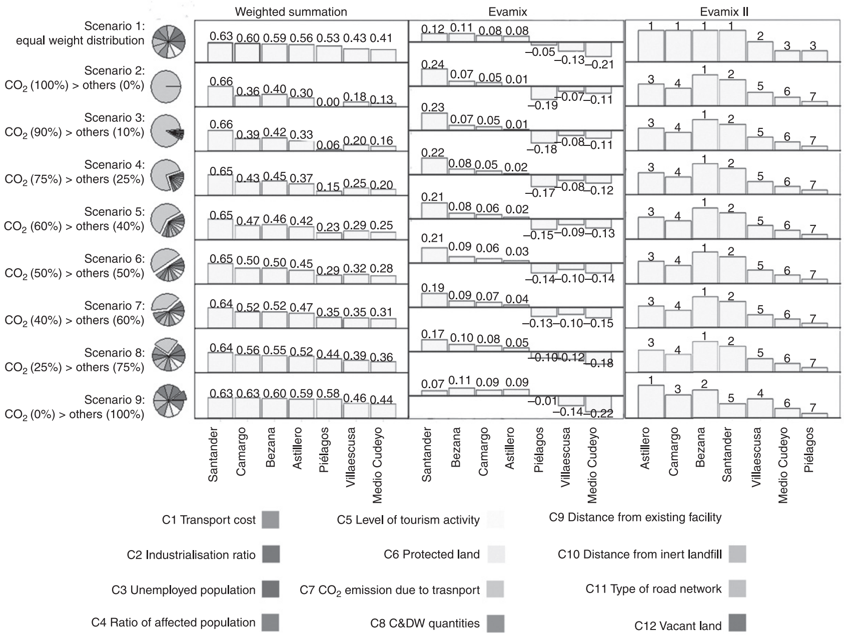


5.4 Estimated generation of C&DW in each of the five areas of Cantabria, northern Spain.

Table 5.5 Impact matrix obtained for the application of MCA to obtain the optimal location for a C&DW recycling facility in Cantabria, northern Spain

Alternatives / Criteria	Ia	Ib	Ic	Id	Ie	If	Ig
(C1) Transport costs (€/Ton)	2.60	2.37	3.26	3.62	2.22	1.27	3.07
(C2) Industrialisation ratio (n° ind/km^2)	31.92	12.66	4.37	1.20	5.62	36.39	1.39
(C3) Unemployed population (%)	10.31	9.46	8.31	8.07	7.02	10.64	8.98
(C4) Ratio of affected population (inhab/km^2)	2568.81	862.55	283.68	239.94	672.48	5224.08	128.52
(C5) Level of tourism activity (N° lodging places/km^2)	26.91	10.85	16.01	14.04	34.05	189.08	8.57
(C6) Protected land (%)	0.00	0.00	31.23	5.58	2.36	0.00	21.78
(C7) CO_2 emission (ton CO_2/km covered)	0.00130	0.00118	0.00162	0.00186	0.00111	0.00063	0.00153
(C8) C&DW quantities (ton)	10.369	18.646	4.490	12.569	6.859	107.313	2.128
(C9) Distance from existing facility (km)	0.00	0.00	0.00	0.00	0.00	0.00	0.00
(C10) Distance from inert landfill (km)	23.10	20.40	27.60	19.30	18.60	26.00	50.50
(C11) Type of road network (km)	13.30	73.90	31.30	88.60	20.30	88.60	19.40
(C12) Vacant land (%)	31.41	27.84	29.27	36.73	45.76	23.07	42.07

obtained with E2, although, according to Alvarez-Guerra *et al.* (2009), this method does not provide final evaluation scores. Moreover, Wang and Triantaphyllou (2008) demonstrated, by means of a computational test, the occurrence of changes in the indication of the best alternative when the E2 method was used, which they



5.5 Results of rankings with different MCA methods of the C&DW facility locations in Area I: Santander for the case study in Cantabria, northern Spain.

referred to as ‘ranking reversals’. Thus, there are compelling reasons to doubt the correctness of the rankings obtained with this method. For this reason, the results obtained with the E2 method must be analysed with care.

In *Scenario 1* of equal weights distribution, all methods indicated that Santander was the best location for the recycling facility. However, in some weight scenarios, more than one alternative was considered as the best solution. For example, in *Scenario 1* of equal weights distribution, E2 (which, as mentioned previously, usually cannot provide complete rankings) gave the best position in the final ranking to four alternatives: Santander, Bezana, Astillero and Camargo. A similar result was obtained with the *weighted summation* in *Scenario 9*, which excludes CO_2 emissions due to transport, in which Santander and Camargo were ranked as the first option.

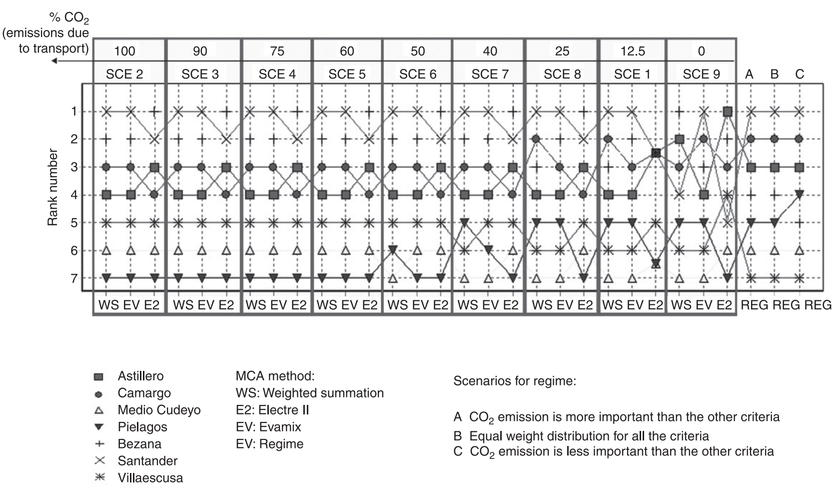
In *Scenario 2*, which only considers CO_2 emissions due to transport, and in the six intermediate scenarios: *Scenario 3*, *Scenario 4*, *Scenario 5*, *Scenario 6*, *Scenario 7* and *Scenario 8*, EV and WS gave the best ranking to Santander, but for E2, Bezana was ranked first in all of these scenarios.

Differences in the facility site alternative ranked first by the EV and WS methods were only observed for *Scenario 9* (which excludes CO_2 emissions due to transport), in which EV gave Bezana as the first alternative, in contrast to WS, which ranked Santander as the first option. In that scenario, E2 yielded differences in the ranking obtained compared with the other scenarios: Astillero was given the best ranking instead of Bezana.

In conclusion, the results indicate that the ranking of the alternatives obtained with EV, WS and REG is similar; all of these methods gave the highest ranking to Santander as the optimal location for the facility. The E2 method yielded a different ranking of alternatives compared with the rest of the MCA methods applied; with this method, Bezana was considered the best alternative.

Excluding *Scenario 9* and despite the differences in the final scores obtained by the E2 method compared to the other methods applied, the rankings derived from EV, WS and REG were considered relevant and yield the alternative of Santander as the best option. The rankings derived from E2 were not considered due to possible ranking irregularities.

The software DEFINITE 3.0 also includes sensitivity and uncertainty analysis. Sensitivity analysis assesses the influence of the weights assigned to each criterion, while uncertainty analysis assesses the effect of uncertainties in the criteria scores. The evolution of the ranking order of each management unit obtained with the different MCA methods and the distributions of weights is represented in Fig. 5.6. Thus, analysis of the sensitivity of the results compared to the criteria weights can be properly evaluated. Figure 5.6 demonstrates that for scenarios of weights in which transport was given greater importance (>75% of the weight), the alternative of Santander is the preferred option for the WS and EV methods, in contrast to E2, which gave the first position to Bezana.

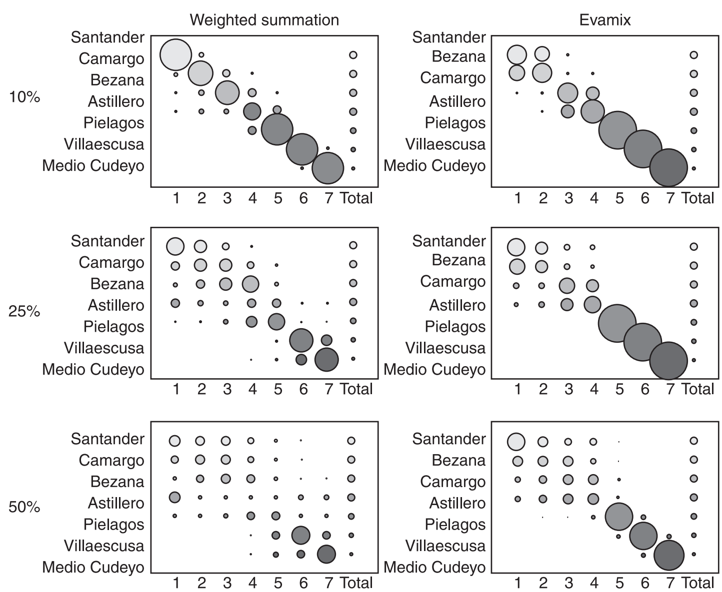


5.6 Sensitivity analysis of the ranking of C&DW facility location alternatives to the criteria weightings with different MCA methods.

For scenarios in which more than 50% of the importance is given to waste transport, the alternative of Pielagos is always the worst alternative. The results obtained with REG, a method that uses only ordinal information on weights, are shown in Fig. 5.6. Based on the analysis of final appraisal scores obtained with the application of the REG method, there are not significant differences among the ranking of alternatives by any of the three scenarios studied, and Santander was considered the best option.

To perform the uncertainty analysis, *Scenario 1*, in which weights were equally distributed among the criteria, was selected to assess possible variations in the results. The probability of an alternative obtaining a specific position in the final ranking is thus calculated. The results obtained, including uncertainties of 10, 25 and 50% for all criteria, are shown in Fig. 5.7, for EV and WS methods.

The size of the circles in Fig. 5.7 is proportional to the probability that each alternative location occupies a specific position in the rank order. The large-sized circles on the main diagonal of the graphs indicate that, despite score deviations from the values assigned up to 10%, the ranking of the areas exhibited little variation. However, this stability decreases when the uncertainty increases up to 25 and 50%, and the probabilities of obtaining different rankings increases. The robustness of the results obtained through MCA methods is confirmed, and



5.7 Influence of uncertainty on the ranking of C&DW facility location alternatives with different MCA methods.

therefore the most suitable site location option for ‘Area I: Santander’ is the alternative of ‘Santander’.

This process must be repeated until enough recycling facilities are properly located. In this case study it was estimated that the minimum quantity of C&DW generation in each area is 50 000 ton/year, to ensure the feasibility of the facility.

In conclusion, the applicability of the proposed MCA-based methodology to the selection of the optimal location of facilities has been demonstrated. In addition, the analyses helped to confirm the robustness of the ranking obtained and reiterated the decision of identifying the optimal location for a C&DW recycling facility, for the case study in Cantabria, northern Spain.

5.6 Conclusions

Environmental decisions are often complex and involve many different stakeholders with different priorities or objectives. Different tools have been developed to facilitate decision-making processes. Some of them have been applied to assess facility locations as CBA, SM, EIA, GIS, OM and MCA. Among these tools, MCA is considered one of the most useful for decision support in waste facilities location problems. That is because MCA integrates different points of view of the involved stakeholders, and in this type of decision problem, it is of utmost importance to include aspects as varied as technical, social, economic, legal, ecological, political, and even cultural aspects in the final decision.

This chapter included an overview of different decision-making tools, to focus on MCA methods and their main characteristics. In addition, the necessary steps in the application of the MCA-based methodology for the selection of adequate C&DW recycling facility locations were explained. Finally, the applicability of this methodology was demonstrated by the application of a real-life case in Cantabria, a northern Spanish region. In that case study, optimal location of a recycling facility among different alternatives in the most populated area of Cantabria was evaluated, with the aim of further understanding of the application of the methodology. In order to ensure the results independently of the method applied, four different MCA methods were included: EV, E2, WS and REG. The results obtained were tested with sensitivity and uncertainty analysis to corroborate the robustness of the solution obtained.

It is important to stress that some aspects based on the points of view of stakeholders needs to be included in the decision-making process, as are the perspectives of the producer of C&DW, the social communities, the regional government, and the recyclers, among others. The ‘Not in My Back Yard’ syndrome and local community acceptance or rejection are undoubtedly some of the most urgent local pressures for the effectiveness of any integrated waste management scheme. For this reason, criteria and weights applied in the MCA method must be adapted to each specific situation. Although the results obtained in this study would be considered a good approximation to evaluate adequate locations of recycling facilities, a thorough study that considers additional economic criteria would be advisable before applying this method in a real situation.

General aspects can be extracted from the results of the case study, one of which is obtained from the sensitivity analysis, which demonstrated that the criterion with the greatest influence on the results was emissions due to waste transport. If this criterion was not included in the evaluation, the ranking of alternatives changed drastically. Finally, application of more than one MCA method to verify the results is highly recommended, and the application of uncertainty analysis is an important step, because a C&DW recycling facility involves a large investment that must be insured for a long time.

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6

The economics of construction and demolition waste (C&DW) management facilities

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Abstract: This chapter gives an overview on key issues related to cost factors and economics of the construction and demolition waste (C&DW) recycling industry, in a broader context of sustainability and resource efficiency. It starts with a brief discussion on current and recent debate on resources and waste management at the European level and from the definition of Sustainable Supply Mix (SSM). The focus is on drivers and barriers to integration between primary production of aggregates (i.e. quarrying) and C&DW recycling towards SSM and to the achievement of the 70% recycling target set by the Europe 2020 strategy. Costs and economic aspects are discussed with emphasis on available technologies for recycling, to assure the technical quality of recycled aggregates. The last section is dedicated to cost factors that can be ascribed to the end-of-waste (EoW) criteria implementation.

Key words: construction and demolition waste (C&DW), recycling, aggregates, recycled aggregates, resource-efficiency.

6.1 Introduction

6.1.1 Economic and policy background of the construction aggregates sector: natural, recycled and secondary aggregates

Sustainability and integration are becoming keywords in many European Commission (EC) policies, including those directly or indirectly related to mineral resources and waste management. Moreover, life-cycle thinking (LCT) is core to many of these policies.

In 2011, the EC issued two Communications on 'A resource efficient Europe' and 'Roadmap to a Resource Efficient Europe', the stated goal of which is to reconsider the whole life cycle of resource use, so as to make the European Union (EU) a 'circular economy' based on recycling and the use of waste as a resource (EC, 2011a,b). There is a strong connection with the Directive 2008/98/EC, the so-called (WFD), which revised the legal framework for waste based on the entire life cycle, from generation to disposal, with emphasis on waste prevention, re-use, recycling and recovery (EU, 2008). Moreover, the EC responded to ongoing public debate about access to and supply of mineral resources with the

Communication entitled 'The Raw Materials Initiative – Meeting Our Critical Needs for Growth and Jobs in Europe', in short, the Raw Materials Initiative (EC, 2008). This integrated strategy has three pillars:

1. to ensure access to raw materials from international markets under the same conditions as other industrial competitors;
2. to set the right framework conditions within the EU to foster sustainable supply of raw materials from European sources; and
3. to boost overall resource efficiency and promote recycling to reduce the EU's consumption of primary raw materials and decrease the relative import dependence.

In the above described context, among other mineral resources, construction aggregates (i.e. gravel, sand and other granular inert materials used in the construction industry) are recognised as essential and valuable resources for the economic and social development of humankind, but they must be produced, used and disposed of according to Sustainable Development principles (Blengini *et al.*, 2012).

Construction and demolition waste (C&DW) is a possible source of unconventional aggregates. However, the current rate of recycling is still low in many EU Member States (MS) and the C&DW recycling goal of the WFD is still to be reached. Eco-efficiency, such as maximising environmental gains of recycling, while keeping under control the unwanted environmental impacts of recycling, is another key issue still to be reached, and for which LCT and life-cycle assessments (LCAs) could give remarkable support (EC *et al.*, 2011). There is an ongoing process at EU level to increase recycling, promote eco-efficiency and assist MS on the path towards a resource efficient Europe according to the third Raw Materials Initiative pillar. This chapter is also intended as a contribution to a better understanding of how the integration between mineral resources management and C&DW recycling can support a more resource efficient Europe.

A necessary background is the definition of Sustainable Supply Mix (SSM) of aggregates: a procurement of aggregates from multiple sources, according to criteria of economic, environmental and social efficiency (Blengini *et al.*, 2012). SSM can therefore be regarded as a blend of natural aggregates, quarry by-products and recycled waste, which together maximise net benefits of aggregate supply across generations. SSM goes beyond the need to ensure a secure supply of aggregates to the economy by adding the requirement that the selected blend must be produced and transported in an eco-efficient manner that minimises total negative impacts and maximises overall benefits to society (Blengini *et al.*, 2012). SSM therefore requires procurement from multiple sources, which must be selected based on comparison of each one's environmental and socio-economic impacts and benefits.

The promotion of recycling and the encouragement of SSM policies was one of the main challenges of the EU SARMa Project 'Sustainable Aggregates Resource Management' (www.sarmaproject.eu). In accordance with the 'The Raw Materials Initiative', SARMa was intended to create the right framework conditions within southeast Europe, in order to foster a sustainable supply of raw materials from European sources and to boost overall resource efficiency and promote recycling. Based on lessons learnt in the SARMa project, natural aggregates and other unconventional aggregates are not in competition but, rather, their joint utilisation is strategic.

A key issue is the correct evaluation of what can be the effective contribution, in qualitative and quantitative terms, that secondary/recycled aggregates can supply to satisfy the requirement for building aggregates. A careful evaluation, based on technical, economic, social and environmental criteria, must be carried out to better understand the role of aggregates from conventional and unconventional sources (Yuan, 2013a,b). Only when it is proved that the recovery/recycling process is both economically and environmentally sustainable, compared with the production of natural aggregates, can the contribution of recycled aggregates be considered net positive.

In such a context, it is important to access recent, complete and reliable statistics on quantities of unconventional aggregates sources and actual recycling rates, as well as on overall aggregates requirements for the construction industry. A further key issue is the discussion on drivers and barriers related to the integration between primary production of aggregates (i.e. quarrying) and other sources towards SSM, with emphasis on C&DW recycling.

6.1.2 Sources of aggregates and market trends

Aggregates can be defined as granular material used in the manufacture of construction products such as ready-mixed concrete (made of 80% aggregates), pre-cast products, asphalt (made of 95% aggregates), lime and cement. According to the European Aggregates Association (UEPG, 2012), global aggregates production is equal to 30 billion tons per year, of which 33% are produced in China, 17% in other Asian countries, 13% in India, 10% in EU + EFTA (Iceland, Norway and Switzerland), 2% in other European countries, 8% in Africa, 7% in the United States, 3% in other NA, 4% in South America and 3% in Oceania.

According to the source material, aggregates can be classified as:

- **natural aggregates**: produced from mineral sources (sand, gravel and crushed rock);
- **secondary aggregates**: sometimes referred as to as manufactured aggregates, produced from industrial processes; and
- **recycled aggregates**: produced from processing material previously used in construction.

However, in some cases, this classification could be seen as not fully comprehensive or even as potentially misleading. For instance, it might be unclear in which category excavated soil and stones would fall, as well as recycled mining waste and, finally, quarry/mine co-products. In the SARMa project, a classification of recycling activities was proposed instead, as such an approach seemed to be more straightforward and comprehensive. The following four types of recycling were therefore considered as potential sources of unconventional aggregates:

- **R1**: recycling of by-products, waste and residues from extractive activities;
- **R2**: recycling of C&DW;
- **R3**: recycling of excavated soil and stones from civil works;
- **R4**: recycling of industrial waste (e.g. slag from civil ferrous metal production, bottom ash from Municipal Solid Waste (MSW) incineration, ash from coal combustion).

Moreover, it should be stressed that some of the input materials that can be recycled into aggregates are classified as waste, while other input materials are not. Regardless of the origin and classification of unconventional aggregates, it is recommended to pay attention to the technical quality, which affects the potential end-uses. Accordingly, the following classification for recycled/secondary aggregates (RA) is proposed:

- **Type A**: high-quality RA for concrete and road construction (road sub-grade);
- **Type B**: medium-quality RA for road, airport and harbour construction;
- **Type C**: low-quality RA for environmental filling and rehabilitation of depleted quarries and landfill sites.

According to UEPG (2012), the European average production of aggregates in 2010 was 5.5 tonnes/per capita, decreasing from 7 tonnes/per capita in 2006. The overall production of aggregates in EU-27 for the year 2012 was 2784 million tonnes, of which 180 million tonnes were recycled aggregates, 55 million tonnes manufactured aggregates and the remaining amount natural aggregates (Table 6.1). Recycled and secondary aggregates account for about 8%, which is a relatively small contribution. However, it is expected that the contribution of unconventional aggregates to the SSM will likely increase by a large extent in the future. Also, one of the barriers is scepticism among conventional natural aggregate producers about the opportunity to extend the scope of their business by integrating recycling aggregates production. According to the UEPG, even in those countries where recycling has almost achieved the maximum target, the contribution of recycled aggregates is below 20%, thus recycled and natural aggregates are not competitors, but rather their joint use is key towards sustainability.

6.1.3 Achieving the 70% target of re-use, recycling and recovery of construction and demolition waste (C&DW) by 2020

C&DW has been identified by the EC as a priority stream because of the large amounts that are generated and the high potential for re-use and recycling embodied in these materials. Indeed, proper management would lead to an effective and efficient use of natural resources and the mitigation of the environmental impacts

Table 6.1 Best estimates of production data by country for 2010

	Total number of producers	Total number of extraction sites	Sand and gravel	Crushed rock	Marine aggregates (Mt)	Recycled aggregates (Mt)	Manufactured aggregates (Mt)	Total production (Mt)
Austria	1070	1362	61	31	0	4	2	97
Belgium	84	112	14	44	8	15	1	82
Bulgaria	190	280	11	14	0	0	0	24
Croatia	175	299	4	14	0	0	0	18
Cyprus	24	24	0	13	0	0	0	13
Czech Rep	202	378	19	37	0	0	0	56
Denmark	350	392	30	0	9	1	8	49
Estonia	31	291	7	0	0	0	0	7
Finland	400	2031	36	48	0	1	0	85
France	1347	2468	135	201	6	17	6	365
Germany	1400	2100	239	208	9	60	19	535
Greece	171	186	1	47	0	0	0	48
Hungary	305	589	30	18	0	3	0	51
Iceland	28	56	2	1	1	0	0	3
Ireland	130	500	10	40	0	0	0	50
Italy	1470	2200	180	120	0	0	0	300
Latvia	30	352	6	3	0	0	0	9
Lithuania	30	427	11	3	0	0	0	14
Luxembourg	7	10	1	1	0	0	0	2
Malta	15	16	1	0	0	0	0	1
Netherlands	145	250	40	0	17	20	0	76
Norway	726	1043	13	54	0	0	0	67
Poland	1542	2475	163	77	0	9	3	252
Portugal	288	362	8	59	0	0	0	67
Romania	430	735	34	15	0	0	0	49
Russia	1181	1485	163	234	0	0	25	422

Serbia	20	70	12	8	0	0	0	19
Slovakia	185	299	8	18	0	0	0	26
Slovenia	30	50	5	8	0	0	0	13
Spain	1475	1520	52	155	0	0	0	208
Sweden	985	1575	17	57	0	1	6	81
Switzerland	537	530	40	5	0	5	0	51
Turkey	770	770	25	290	0	0	0	315
UK	885	1393	51	106	10	49	10	226
34 Countries	**16658**	**26630**	**1426**	**1929**	**59**	**186**	**80**	**3680**
Like-for-Like	**15306**	**23943**	**1230**	**1680**	**58**	**185**	**55**	**3209**
EU-27+EFTA	**14512**	**24006**	**1223**	**1383**	**59**	**186**	**55**	**2906**
EU-27	**13221**	**22377**	**1168**	**1323**	**58**	**180**	**55**	**2784**

Note: The ‘Like-for-Like’ figures provide a comparison with the 26 countries included in the 2009 data published by UEPG. The EFTA countries include Iceland, Norway and Switzerland.

Source: UEPG, 2012.

to the planet (BIO Intelligence Service, 2011). For this reason, the WFD requires MS to take any necessary measures to achieve a minimum target of 70% (by weight) of C&DW by 2020 for preparation for re-use, recycling and other material recovery, including backfilling operations using non-hazardous C&DW to substitute other materials. The above target excludes naturally occurring material, defined in the EWC 17 05 04 list of waste as 'soil and stones'.

One of the obstacles for the achievement of the 70% target by 2020 is the incomplete and inconsistent information available in statistics dealing with both generation and recycling of C&DW. These data gaps and inconsistencies can partially be ascribed to the definition itself of C&DW. The nature-oriented definition is often misleading. The recommended approach to define C&DW is to take into account both its nature (materials used in buildings) and the activities that create it (construction and demolition activities), regardless of whom carries out these activities.

As quantities and nature of C&DW arising in the EU are concerned, available estimates are highly variable. Table 6.2 reports estimates from a recent survey carried out in the context of an EC contract (BIO Intelligence Service, 2011). It is important to note that in Table 6.2 recycling rates are intended as 'preparation for re-use, recycling and other forms of material recovery', as defined by the WFD. According to the data in the table, the average recycling rate for EU-27 is 46%. For six countries, no data is available for estimating the recycling rates (Bulgaria, Italy, Malta, Romania, Slovakia and Sweden). A major source of uncertainty in reporting is inclusion/exclusion of excavated soil and stones. Analysis of the table shows that the quantities reported in national statistics include high amounts of excavated material, which is not included in the definition of C&DW for the purpose of the 70% target set by the WFD. The inclusion of excavated material does not seem to be systematic in national reporting. This flow represents up to 80% (e.g. in France) of the total amount of construction, demolition and excavation waste.

The resulting ranges of quantities of C&DW arising in the EU are 0.63 to 1.42 tonnes per capita per year (excluding excavation material) or 2.3 to 5.9 tonnes per capita per year (C&DW plus excavation waste). Very low levels of generation reported in some MS probably reflect a lack of control by public authorities and therefore a incomplete reporting of C&DW quantities, which are therefore likely to range between a total of 310 and 700 million tonnes per year in the EU-27. Inclusion of excavation waste would significantly increase these amounts, ranging from a total of 1350 to 2900 million tonnes per year.

6.2 Drivers and constraints for the development of the recycling sector

As far as the 70% target is concerned, a main barrier to a higher level recycling of C&DW is that natural aggregates are often locally available and relatively inexpensive. Moreover, in some areas, landfilling of C&DW is still possible at

Table 6.2 C&D waste arising and recycling rates in the EU27

Country	C&D waste arising (Mt)	C&D waste arising (tonnes/capita)	%Re-used recycled
Austria	6.60	0.81	60
Belgium	11.02	1.06	68
Bulgaria	7.80	0.39	n.a.
Cyprus	0.73	0.58	1
Czech Republic	14.70	1.44	23
Denmark	5.27	3.99	94
Estonia	1.51	1.12	92
Finland	5.21	3.99	26
France	85.65*	5.50	45
Germany	72.40	2.33	86
Greece	11.04	0.37	5
Hungary	10.12	0.43	16
Ireland	2.54	2.74	80
Italy	46.31	0.80	n.a.
Latvia	2.32	0.04	46
Lithuania	3.45	0.10	60
Luxembourg	0.67	5.90	46
Malta	0.8	1.95	n.a.
Netherlands	23.9	1.47	98
Poland	38.19	0.11	28
Portugal	11.42	1.09	5
Romania	21.71	n.a.	n.a.
Slovakia	5.38	0.26	n.a.
Slovenia	2.00	n.a.	53
Spain	31.34	0.74	14
Sweden	10.23	1.14	n.a.
United Kingdom	99.10*	1.66	75
EU 27	**531.38**	**1.74**	**46**

* Corrected with the exclusion of excavated materials.

Source: BIOIS, 2011.

low costs. As a result, the economic attractiveness of unconventional aggregates from mineral C&DW can be low compared to natural aggregates. Recycling of C&DW has been mostly successful in urban areas, where virgin raw materials extracted from quarries are less abundant, or became more expensive due to long transport distances (Zhao *et al.*, 2010). A possible counter measure can be to make landfilling of waste unattractive, by introducing a ban or tax on landfilling, which could prove effective towards higher recycling rates. Alternatively, taxes on resource extraction could contribute to an increase in the price of primary raw materials and make recycling more competitive (Bio Intelligence Service, 2011).

The following are some of the main drivers that can boost C&DW recycling or constraints that might act in the opposite direction:

- Green Public Procurement (GPP);
- taxation on natural aggregates;
- taxation on landfill of C&DW;
- availability and cost of natural aggregates;
- quality certification of RA from C&DW;
- better public perception and increased consumer acceptance.

The above reported drivers/constraints are briefly discussed below.

6.2.1 Green Public Procurement

Given the significant share of public funded construction works, Green Public Procurement (GPP) can play a major role in promoting recycled aggregates and the use of recyclable materials. This voluntary instrument can help stimulate a critical mass of demand for more sustainable construction materials, which otherwise would be difficult to get onto the market. A number of regions and municipalities are applying GPP criteria, but further use should be encouraged. It is expected that GPP can help remove an important barrier, that is, the low market demand. However, often public authorities, who are the main purchaser of construction material (through public works), do not specifically ask for recycling material in their call for tenders. This can be due to negative experiences with non-certified, low-quality material.

6.2.2 Taxation on natural aggregates

The market for primary, secondary and recycled aggregates is influenced by a number of factors, such as taxation on primary aggregates, landfill taxation, availability and cost of natural aggregates and public perception or consumer acceptance. Furthermore, the aggregates market is influenced by the demand of building materials, which depends on the situation of the construction sector highly linked to the economic situation. The use of recycled and secondary aggregates differs from country to country, according to the management policies (landfill taxes) and restrictions on the use of natural resources (taxation on natural aggregates). Countries with taxes on landfill and primary aggregates extraction have the highest recycling rates.

Several MS have implemented taxation on primary aggregates (EC *et al.*, 2009):

- a tax on resource extraction;
- pollution taxes and/or taxes on mining waste.

Often prices for primary natural resources do not yet reflect the environmental disadvantages (use of resources) that occur during their production. Taxes on resource extraction may be used as a way of encouraging the substitution of secondary and recycled materials for natural materials. In general, natural materials are often associated with more negative externalities than recycled materials. One commonly cited reason is that the processing of secondary materials tends to be less energy intensive. In addition, recycling is one way of avoiding the disposal of solid waste. Taxes on virgin materials will change the relative price of natural and recycled materials, and in this way influence waste disposal behaviour. Theoretically, charges on waste disposal would be a good policy in this case, but several studies have also argued that direct charges on waste disposal can be ineffective because of the risk of illegal disposal.

For instance, Sweden introduced taxes on natural gravel in 1983. The tax was raised to SEK 10 per tonne in 2003. The tax is levied on extraction in Sweden and on extraction for export but not on imports, thus a possible disadvantage is that imports might become relatively cheaper. In 1990, Denmark set a tax of DKK 5 per m^3 for selected extracted raw materials, including sand, gravel, stones, clay and limestone. The Danish tax is levied on raw materials that are commercially extracted and consumed in Denmark or commercially imported, while no tax is levied on exports. The UK tax on aggregates came into effect in 2002. It is targeted at the extraction of sand, gravel and crushed rock and set at GBP 1.6 per tonne. The tax is levied on the extraction of minerals for the production of construction aggregates and imports to the UK (with the exception of recycled aggregates) but excludes exports.

Typically, the tax is intended to reduce demand for aggregates and encourage the use of alternative materials wherever possible. A tax on aggregates extraction also reduces the incentive to find new deposits, thereby limiting the economic availability of the resource. Taxing aggregates to promote the use of recycled materials is justified if the environmental net benefits increase as a result. Further restrictions on planning permission for new extraction sites will make recycling essential – the scarcity of virgin aggregate that will inevitably be created by dwindling reserves will push up aggregate prices, making re-use of existing materials vital (Böhmer *et al.*, 2008).

6.2.3 Landfill taxation

Low prices for disposal in landfills do not support recycling. The decision to go for recycling is strongly dependent on the cost of disposal. The low price of primary aggregates, including low transport costs, makes the substitution of primary materials for recycled and secondary aggregates difficult. This, together with a lack of rules to guarantee the quality of secondary aggregates, explains the low recycling rates. The purpose of landfill taxation is to make the landfill of waste more expensive than alternatives, forcing the separation or post-separation of waste streams into sub-streams suitable for recovery to become financially more attractive.

Table 6.3 Landfill taxes for different Member States

Member State	Tax €/t of waste	Brief description
Austria	8	Excavated materials and inert construction waste
	18	Inert residues
Belgium (Flanders Region)	0.32–7.73	Waste from mining and mineral industries and to recycling and soil sanitation residues
	10.83	Inert waste and inert asbestos
Czech Republic	15.85	Disposal of non-hazardous waste in landfill
Denmark	50.31	Landfilling tax
	28.4	Landfilling of residual waste (slag and fly ash)
Finland	30	Waste left at public landfill sites. Fly ash excluded
France	7.5–18.29	For non-hazardous and hazardous waste. Inert waste not taxed
Germany	–	No taxation
Italy	1.03–10.33	Industrial waste from mining extractive building and metalworking sector activities
Netherlands	13.98	Non-combustible waste. C&DW are banned from landfill since April 2000
Spain	3–10	C&DW
UK	3.36	Ceramic and concrete materials, furnace slags and ash
	43.06	Active waste

Source: Böhmer, 2008; EC *et al.*, 2009.

According to the Joint Research Centre of the EC (Böhmer *et al.*, 2008, EC *et al.*, 2009), landfill taxes are variable in different MS (Table 6.3). Landfilling costs differ substantially, the prices varying from 3 to 50 € per tonne of waste. Landfill taxes in other European countries are also increasing rapidly and are expected to reach more than 70 € per tonne in both the Netherlands and the UK in 2011.

According to Duran *et al.* (2006), the use of taxes has been seen as a tool to encourage recycling through the increase in the cost of either landfilling or using natural aggregates. Policy makers can also encourage recycling through the use of subsidies, which makes the creation of markets for recycled C&DW more economically viable. However, subsidies impose a cost on the public sector. A combination of taxes and subsidies could minimise this cost incurred by the public sector as the taxes could finance the subsidies. Anyway, Coelho and de Brito (2013b) underline that the economic viability for a C&DW recycling plant is likely to occur in pure open market conditions, that is, without subsidies or specific legislation in favour of C&DW recycling.

6.2.4 Availability and cost of natural aggregates

In present-day markets, secondary and recycled aggregates have to compete against natural aggregates, and the market for secondary and recycled aggregates is heavily influenced by the availability and quality of natural aggregates. Prices of natural aggregates can vary dramatically from country to country, depending on the availability of hard rock, limestone and sand and gravel resources, as well as quality. According to Böhmer *et al.* (2008) and EC *et al.* (2009), the average natural aggregates prices at the extraction site for 2007 vary from 2.5 to 12 €, for most MS, from 6 to 7 €. In 2007, the highest price rises in natural aggregates were seen in Eastern Europe, particularly in Russia, Hungary, Romania and Bulgaria (12 €). To compare aggregates prices across Europe, the analysis must be based on the extraction price and not the cost at the construction site, which will include transport costs that could distort the overall picture. The average price in European countries is not only influenced by market forces but also by the type of resources in a particular region, so that the cost structure for extracting hard rock is different than for sand and gravel extraction.

A key issue influencing the price of natural aggregates are transport costs. In normal cases, natural aggregates have to be mined outside of highly populated regions and transported long distances to the production areas or the areas where they are used. Recycled and secondary aggregates are first generated within production or construction processes taking place near highly populated regions. This gives recycled and secondary aggregates some cost advantages in terms of lower transport distances. For these reasons, in some MS, obligations for recycling activities are related to the transport distance:

- proximity and quantity of available natural aggregates;
- reliability of supply, quality and quantity of C&DW (availability of materials and capacity of recycling facility);
- government procurement incentives standards and regulations requiring different treatment for recycled aggregate compared to primary material;
- taxes and levies on natural aggregates and on landfill.

According to the World Business Council on Sustainable Development (UEPG, 2011), recycled concrete aggregates in Europe can sell for 3 to 12 € per tonne, with a production cost of 2.5 to 10 € per tonne. The higher selling prices are obtained on sites where all C&DW is reclaimed and maximum sorting is achieved, there is strong consumer demand, lack of natural alternatives and supportive regulatory regimes.

The above source reported that there is a remarkable geographic variation in the comparable profit margin for aggregates producers. As an example, in Paris there is a lack of natural aggregates making recycled aggregates an attractive alternative and the recycling market is driven mainly by civil works companies. In Rotterdam, higher production costs for recycled materials compared to virgin materials are compensated by high tipping fees. In Brussels, the lack of landfill space pushes C&D companies to drop market prices to find solutions for their waste, while in Lille the abundance of quarries makes the higher production costs a limiting factor (as raw materials have lower costs).

6.2.5 Quality certification of recycled/secondary aggregates from C&DW

The technical requirements for the use of C&DW in the production of aggregates are also addressed by the individual CEN Aggregate Product Standards, which set clear quality requirements for the different types of applications (e.g. aggregates for concrete, aggregates for mortar, aggregates for unbound and hydraulically bound materials for use in civil engineering work and road construction, etc.) and ensure that the end-products are durable and meet their technical specifications. These standards were developed in 2004 and clearly address 'aggregates from natural, recycled and manufactured materials'. Moreover, for natural and recycled aggregates, the 80/106/EEC Directive on construction materials requires the respect of technical specifications imposed by the CE marking harmonised standards (e.g. EN 12620 aggregates for concrete, EN 13242 aggregates for road construction, etc.), without consideration of the raw materials' sources. As such, the Directive represents an important tool to remove obstacles to the use of recycled aggregates as building products.

The key to the successful recycling of the mineral fraction of C&DW is that the waste collected is 'clean', that is, free of contaminants and other materials. The main recommendations to overcome this are the following:

- Encourage the sorting of C&DW 'at source': efforts should be made to sort out the different materials composing C&DW.
- Clear identification of materials and potential contaminants should be performed in order to avoid contamination of the inert fraction and ensure high quality of recycled material. This will also drive the separation and collection of 'smaller' but valuable fractions such as glass, metals, plastics, gypsum, etc., and the appropriate management of hazardous materials such as ODS containing foams.
- Selective demolition/controlled deconstruction: practices of 'controlled deconstruction', consisting in the systematic removal of contaminants prior to demolition, as well as the sorting of different building materials, should be encouraged and generalised.
- Further research on applications of recycled C&DW and, particularly on long-term behaviour, could contribute to reduce uncertainty linked to the use of recycled products.

6.2.6 Better public perception and increased consumer acceptance

The existence of national provisions and guidelines, which guarantee the quality of secondary and recycled aggregates, improves the user perception and the consumer confidence in the use of recycled and secondary aggregates. The acceptance of recycled and secondary aggregates by the final consumer is strongly linked with the waste status of the material. Even with new products meeting the same technical requirements, consumers may find it hard to trust them, especially when the products are made of waste. Proposing new products on the market requires active awareness raising and promotion, even if they are cheaper.

In such a context, the development of end-of-waste (EoW) criteria, as foreseen in the WFD, could also contribute to improve the image of the recycled aggregates and reduce uncertainty of the markets (EC *et al.*, 2009). Moreover, EoW criteria shall provide minimum quality requirements that would assure safe use of the material. This influences the consumer acceptance in a positive way. The CE mark associated with the fulfilment of technical requirements defined in the European standards, supports consumer acceptance and confidence in the recycled and secondary aggregates without, however, providing a totally secure guarantee of the environmental safety. CE marking will be a good legislative driver, but taxes or incentives to reduce dependence on virgin aggregates and make recycling a financially attractive alternative are also necessary in order to promote recycling. Communicating the benefits that secondary raw material can transfer to the end-products is another important tool. It is demonstrated that up to 20% of recycled aggregates can be used in structural concrete, without changing the quality of the product. Moreover, in road works, recycled aggregates present better properties than virgin aggregates.

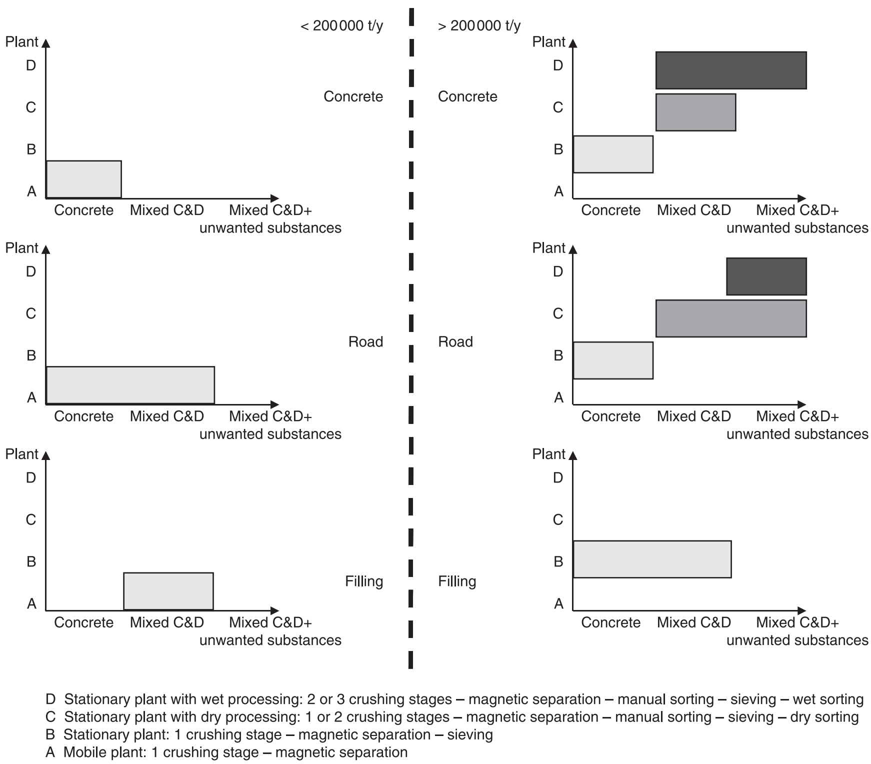
6.3 Cost factors of construction and demolition waste (C&DW) recycling

6.3.1 Technical and economic classification of C&DW treatment plants

As a general rule, it can be underlined that mineral dressing principles are the same for both natural and recycled aggregates. Furthermore, the equipment employed in the beneficiation of primary mineral materials can be successfully adapted or re-designed for C&DW recycling. Typically, C&DW treatment processes start with manual sorting, followed by comminution (mainly with impact or jaw crushers), magnetic separation and, eventually, sieving. In the case of mobile plants, the whole process ends at this stage, while in a stationary plant, one or more crushing, sieving and sorting units are added. With reference to sorting processes, unwanted components, such as organic materials or lightweight construction materials, have to be removed because they can adversely affect the quality of the aggregate due to issues of mechanical strength, grain shape, chemical reactivity, swelling, surface adhesiveness, magnetic susceptibility, colour, low melting point and other intrinsic properties. A dry process simply blows away lightweight materials, producing a recycled aggregate suitable mainly for road construction.

In order to separate both organic and lightweight construction materials, a wet processing section often has to be added to the processing plant, because it offers higher separation efficiency in comparison to dry processing. However, it requires an additional, and complex, processing stage for the treatment of process water and the disposal of slimes. These additional operations make wet processing more expensive than other sorting actions. The option of the wet process is probably unavoidable when recycled aggregates are used in concrete manufacturing. According to Garbarino (2010), the main washing machines are sink float machinery, up-current sorters, table belts (e.g. Aquamator) and jigs. According to Yuan (2013a), a sufficient on-site space for on-site waste collection, sorting and handling is also to be taken into consideration to maximize C&DW re-use and recycling.

As shown in Fig. 6.1, the appropriate choice of a processing plant depends on:



6.1 Plant and feed complexity versus possible employments classification scheme.

- the complexity of the feed material, variable from concrete monoliths to mixed rubble also containing unwanted substances;
- the treatment typology:

  a) **mobile plant**: only crushing and magnetic separation;
  b) **stationary plant**: crushing, magnetic separation and sieving units;
  c) **stationary plant with dry processing**: crushing, magnetic separation, sieving, hand picking and dry sorting; and
  d) **stationary plant with wet processing**: crushing, magnetic separation, sieving, hand picking and wet sorting;

- the plant capacity. According to the literature, the mobile vs. stationary plant capacity threshold averages 200 000 t/y;
- the end-use of the recycled aggregates, i.e. environmental filling, road construction and concrete manufacturing.

6.3.2 Cost factors of treatment equipment

Feed, pre-screening and crushing equipment are usually employed in a mobile plant. In Table 6.4, technical characteristics of different mobile plants and equipment are summarised in order to underline some common features.

With respect to stationary treatment plants, one plant could differ from another, especially with reference to the adapted separation systems. In Table 6.5, technical

Table 6.4 Technical characteristics of different mobile plants equipped with feed, pre-screening and crushing units

Producer	Model	Weight (t)	Inlet opening (mm*mm)	Power (kW)	Max throughput (t/h)	Min throughput (t/h)
HMH (Austria) http://www.rubblemaster.com/	RM70	18.5	760*650	103	120	
ERIN (Canada) http://www.erinsystems.com/	PC1055J	33.0	1000*550	186	200	
	PC1265J	44	1250*650	231	250	
	PC1380J	53	1300*800	291	350	
OM (Italy) http://www.omtrack.it	Mercurio	20.5	735*500	93	70	10
	Ulisse	34.5	900*700	168	200	30
	Apollo	39.5	1050*730	186.5	240	35
	Saturno	53	1070*750	205	360	45
	Marte	35	1100*750	205	224	30

Table 6.5 Technical characteristics of different stationary plants equipped with dry or wet separation units

Dry separation plant (Prüwasser, 2001)		Wet separation plant – simple washing (Verniéres and Carlier, 1998)		Wet separation plant – density separation (Schütze, 1987; Buntenbach, 1997)		Wet separation plant – density separation (Derks *et al.*, 1997)	
Capacity	400 000 t/y	Capacity	300 000 t/y	Capacity	500 000 t/y	Capacity	120 000 t/y
Feed	feeder, apron	Feed	feeder, apron	Feed	feeder, apron	Feed	feeder, apron
Pre-screening	feed roller grid	Pre-screening	feed roller grid 30 mm	Pre-screening	feed vibratory grid	Pre-screening	
Manual sorting	1 belt conveyor			Manual sorting	1 belt conveyor	Manual sorting	1 belt conveyor
Primary crushing	Impact crusher spread dimension 1000*1000 mm	Primary crushing	Impact crusher (inlet opening 1250*500 mm)	Primary crushing	Jaw crusher (throughput 150 t/h)	Primary crushing	Impact crusher
Magnetic separation	2 overband separators	Magnetic separation	2 overband separators	Magnetic separation	2 overband separators	Magnetic separation	2 overband separators
		Manual sorting					
Sieving	Vibratory screens 2 decks (32 and 12 mm)	Sieving	Vibratory screen 2 decks (60 and 20 mm)	Sieving	Vibratory screen 2 decks (50 and 10 mm)	Sieving	Vibratory screens 1 deck (20 mm) 1 deck (8 mm) 2 decks (100 and 40 mm)
		Secondary crushing	Impact crusher	Secondary crushing	Impact crusher (throughput 60 t/h)		

Sieving	Vibratory screen 2 decks – 12 and 4 mm	Sieving	Vibratory screen (2 decks – 20 and 6 or 10 mm)	Sieving	Vibratory screen 2 decks – 45 and 6 mm	Sieving	Vibratory screen 1 deck (20 mm) 1 deck (4 mm)
Dry separation	3 air separators					Dry separation	3 air classificators
		Wet separation	Density separator: screw hydro classifier	Wet separation	Aquamator (2.2 kW)	Wet separation	Pulsator jig (solid throughput 90 t/h and water throughput 240 m^3/h)
				Waste water process	Dewatering solids pump (22 kW) hydrocyclone (diameter 300 mm) sludge pump (30 kW) sludge and fresh water basins	Waste water process	dewatering

characteristics of existing plants are shown (Prüwasser, 2001; Verniéres and Carlier, 1998; Schuetze, 1987; Buntenbach *et al.*, 1997; Derks *et al.*, 1997). For each plant, the subsequent processing phases are shown in the left column, while in the right column, the employed equipment is described. It can be seen that some common operations can be identified, as feed, pre-screening, manual sifting, primary crushing, magnetic separation and sieving. Secondary crushing is not always performed and relevant differences occur in the separation units.

With reference to the main features highlighted in Tables 6.4 and 6.5, industry data, the Mine and Mill Equipment Cost Guide (InfoMine, 2010) and literature, (Coelho and de Brito, 2013a,b; Duran *et al.*, 2006) have been used to provide best estimates of the capital and operating costs associated with mobile and stationary plants for C&DW recycling (Table 6.6). The capital costs (CC) are list or budget prices for the described equipment item. Most of the item descriptions represent actual equipment models. In the guide, it is emphasised that large disparities can exist between prices listed and those actually charged to a specific buyer, because selling prices are commonly discounted to some degree from list prices. The discount offered by a manufacturer will depend on such factors as the number of units ordered and how well the model is selling at that particular moment.

Capital recovery costs (CR, euros per hour) provide an indication of the funds necessary to purchase or replace the machine, and they are obtained by subdividing the CC for the estimated replacement life (R, e.g. in the case of crushers, R varies from 26 000 to 52 000 h). The overhead costs (O) listed in the guide are indirect administrative costs associated with machine ownership. They include insurance, license, and maintenance and record keeping charges. They are determined by multiplying CR with the factor F, that is, an experience-based factor.

The hourly operating costs (HOC) are variable and applied directly to the estimated equipment use (hours). They are the sum of the following costs:

- **Overhaul parts costs**, which are those associated with scheduled reconstruction and/or replacement of major components such as engines and transmissions for mobile equipment.
- **Overhaul labour costs**, which are those associated with scheduled reconstruction and/or replacement of major components such as engines and transmissions for mobile equipment. For stationary processing equipment, the costs are for scheduled refurbishing or replacement of major wear components such as drives, support frames and vessels.
- **Maintenance parts costs**, which are associated with both unscheduled repairs and scheduled servicing of both minor and major components, excluding overhaul activities. These include all aspects of machine maintenance exclusive of fuelling, lubrication, tyre replacement, and maintenance and replacement of those parts used directly to impart energy.
- **Maintenance labour costs**, which represents a typical charge per hour of operation to cover mechanics' time to perform maintenance and repair functions, exclusive of overhaul work.

- **Fuel/power costs**.
- **Lube costs** of crankcase oil and other lubricants required to operate the equipment.
- **Tyre costs**.
- **Wear parts costs**, which typically refer to the costs of parts which directly engage the rock and impart some form of energy designed to change the condition of that rock (e.g. excavator teeth, crusher, grinding media, etc.).

The total HOC does not include depreciation, overhead, insurance, or cost of facilities capital. Nor does it include the cost of operators.

Table 6.6 Capital costs, capital recovery, overhead and hourly operating costs of different equipments in mobile and stationary plants

Processing unit	Equipment	Capital costs (CC)		Capital recovery (CR)		Overhead (O)		Hourly operating costs (HOC)	
		min k€	max k€	min €/h	max €/h	min €/h	max €/h	min €/h	max €/h
Feed	Feeder apron (steel)	140	350	5.3	13.5	0.19	0.47	9.0	23.0
	Feeder apron (manganese)	35	260	1.3	10.1	0.05	0.35	2.2	17.2
Pre-screening	Feeder, vibrating	10	25	0.3	0.9	0.02	0.03	0.6	1.6
Crushing	Jaw crusher	165	320	6.4	12.2	0.23	0.43	24.2	40.3
	Impact crusher	115	150	4.4	5.9	0.15	0.20	13.7	17.8
Magnetic separation	Overband	20	75	0.4	1.4	0.02	0.05	1.5	3.4
Sieving	Screen (1-2-3 decks)	20	40	1.0	1.3	0.03	0.05	1.5	2.0
Dry separation	Air separator	30	95	1.7	3.6	0.06	0.13	1.9	3.9
Wet separation	Jig	380	425	7.3	8.2	0.26	0.29	17.4	19.4
	Aquamator	150	300						
	Density separator	80	105	3.1	4.1	0.11	0.14	3.3	4.3
Water water process	Filterpress	435	485	8.4	9.3	0.29	0.32	18.3	20.2
	Dewatering, desliming	30	35	1.1	1.4	0.04	0.05	1.7	2.3
Mobile plant (feed, pre-screening and crushing)		400	450	24.5	25.0	0.85	0.90	43.5	47.9

Source: from industrial data and from the Mine and Mill Equipment Cost Guide (Infomine, 2010).

According to Zhao *et al.* (2010), the lifetime of main equipment has to exceed the payback period of investment on disposal facilities as a necessary precondition of the investment. Furthermore, estimated profit from internal rate of return, such that the discount rate for which the total present value of future cash flows equals the cost of the investment, has to exceed the opportunity cost of capital investment with reference to interest of saving.

6.3.3 Cost factors of processing plants

Costs factors of C&DW processing generally differ from plant to plant and some local conditions also have to be considered. In order to highlight some common features, some examples are presented in Table 6.7, where the analysis of the total costs for a stationary plant and a mobile treatment plant in Germany are shown (Maurer, 2002). In Table 6.8, total costs, total income and total profit for two Italian stationary plants are presented (personal communications and data reported in Garbarino, 2005). These data can be compared with the cost factors presented by UEPG (2011) (Table 6.9).

Coelho and de Brito (2013a) analyse the feasibility of a C&DW recycling plant with a 350 t/h treatment capacity, to be installed in the Lisbon Metropolitan area. Initial and operating costs are considered, including the set-up costs of real estate purchase and engineering design costs, as well as life-cycle maintenance/ replacement costs within 60 years of plant lifetime. The authors highlight that

Table 6.7 Total costs of two treatment plants in Germany

Plant typology		Stationary		Mobile	
Capacity	(t/y)		360 000		120 000
Capital costs (CC)	(€)		1 040 640	Crushing unit (75%) + sieving unit (25%)	515 000
Financial costs	(€/y)	Depreciation of 10% in 10 years	104 064	Depreciation of 16% in 6 years	82 400
	(€/y)	Interest ratio 8% of CC/2	41 626	Interest ratio 8% of CC/2	20 600
	(€/y)	Insurance	39 542	Insurance	19 570
Production costs					
Labour cost	(€/y)		153 600		50 700
Energy costs	(€/y)		65 249		25 412
Maintenance costs	(€/y)		110 592		42 000
Storage costs	(€/y)				72 000
Total costs	(€/y)		**514 673**		**312 682**
Total costs per tonne	(€/t)		**1.43**		**2.61**

Source: Maurer, 2002.

Table 6.8 Total costs, income and profit of two treatment plants in Italy

Plant		Ferrara dry stationary plant			Torino dry stationary plant		
Capacity	(t/y)		90 000			150 000	
Capital costs CC	(€)		1 350 000			1 549 371	
					Forgivable loan	433 824	28%
Financial costs	(€/y)	Depreciation of 6.7% in 15 years	90 000		Depreciation of 6.7% in 15 years	74 370	
Production costs	(€/y)	Landfilling of organic materials (60€/t of hazardous organic waste or 10€/t of not hazardous organic waste)	8100	3%	Landfilling of organic materials	10 404	2%
	(€/y)	Labour costs	90 000	34%	Labour costs	88 437	20%
	(€/y)	Energy costs			Energy costs	15 606	59%
	(€/y)	Maitenance costs	54 400	20%	Maintenance costs	260 110	4%
	(€/y)	Fixed costs	114 500	43%	Fixed costs	67 629	15%
Total costs	(€/y)		**447 000**			**516 566**	
Total costs per tonne	(€/t)		**5.0**			**3.4**	
Total income	(€/y)	Reception fees (1.55€/t of incoming waste)	139 500	15%	Reception fees	380 062	39.5%
	(€/y)	Iron waste sale (30€/t of iron waste)	16 200	2%	Iron waste sale	4811	0.5%
	(€/y)	Sale of recycled aggregates (9€/t of produced aggregates)	801 900	84%	Sale of recycled aggregates	577 309	60%
	(€/y)	Total	**957 600**		Total	**962 182**	
	(€/t)	Total	**10.6**		Total	**6.4**	
Total profit	(€/y)		**600 600**			**445 626**	
	(€/t)		**6.7**			**3.0**	

Source: Personal communication plus data in Garbarino, 2005.

Table 6.9 Total costs, income and profit

Production costs	Costs for reception and storage	(€/t)	1
	Treatment costs	(€/t)	5
	Total		**6**
Fixed costs		(€/t)	**1**
Total costs		(€/t)	**7**
Total income	Reception fees	(€/t)	10
	Sales of recycled aggregates	(€/t)	3
	Total	(€/t)	**13**
Total profit		(€/t)	**6**

Source: Summarised by UEPG (2011)

investment in a large-scale C&DW recycling centre is a multi-million euro enterprise, but has a high profit potential, even in the absence of specific subsidies. In the case study under review, the reception fee of incoming waste (48 €/t for mixed C&DW and 8 €/tonne for separated C&DW streams) is the largest share, providing around 86% of all benefits.

In terms of costs, fees for transport and landfilling of rejected input waste account for almost 70% of all yearly equipment operating costs and about 9% of the total 60 year costs, which puts into perspective the need to minimise non-treated material. Furthermore, Coelho and de Brito (2013b) develop a simplified sensitivity analysis to see how the variation of operating parameters (such as plant's capacity, C&DW input waste reception fee, recycled aggregate selling price, rejected input waste landfill price, percentage of mixed/separated C&DW input and C&DW input mass rate) would influence the return on investment period. They evaluate a return on an investment period of under 8 years, 1 or 2 years under favourable conditions, for a wide range of parameter variations.

6.3.4 Emerging C&DW management practices and future trends

One of the key aspects of C&DW, next to prevention, is the high end re-use and valorisation of recycled waste. As pointed out above, possible bottlenecks in the recycling chain are assigned to problems with:

1. purity or homogeneity of input wastes;
2. quality of recycled materials;
3. installation and management of collection system; and
4. technical and economic feasibility of the recycling process.

Possible ways of overcoming these barriers are:

- **Sorting at source**: For recycled aggregates derived from C&DW, the generation of the material is one of the most relevant steps. The removal of hazardous and non-hazardous materials before the demolition is the most effective way to ensure that unwanted materials are not present in the input material, and consequently in the recycled product. Asking for a demolition project during the design of a building and integrating specific requirements for selective demolition in the building permit (for works of a certain size) can improve the whole chain. Selective demolition is extremely relevant for the identification and removal of hazardous materials present in the building shell, such as asbestos. This could also allow a better monitoring of the waste streams. However, that kind of demolition entails higher costs than traditional ones (according to EC *et al.* (2009), costs associated with selective demolition could be 17–25% higher), because more time, special machinery and more space are needed.

- **Closed loop material**: The waste management plan in the construction sector has to reach the high end valorisation of recycled waste. This is valid both for the largest mineral fraction and for the smaller fractions (gypsum, metal, glass, etc.). Focus on these fractions can increase the quality of the mineral fraction and create opportunities for recycling.
- **Gypsum waste management**: Recycling is a sound measure for decreasing the exploitation of gypsum as raw material. According to BIO Intelligence Service (2011), an increase in price of raw gypsum material has been observed in some plants, where it has gone up by more than 50% since 2008. Another economic aspect linked to gypsum waste management is the cost associated with the increasing landfill taxes for plasterboard waste. For example, in the UK between 2004 and 2008, the total landfilling cost of gypsum increased from 67 to 146 € per landfilled tonne (i.e. an increase of 117%). The effect of the increasing landfill costs is assumed to be the development of recycling practices among gypsum waste producers and manufacturers. More specifically, a clean gypsum powder can be recycled and re-used in the production of new gypsum board. The collection and recycling of gypsum is a key example of global chain management to close the material loop for gypsum. Producers have to design their products to be more environmentally friendly and easily recyclable. Dismantling and demolition companies commit to sort at source, while collection and sorting companies increase their efforts to separate the gypsum from C&DW. For example, in 2010, the Flemish region targets to recycle 25 000 tonnes of gypsum waste (40% of the total gypsum waste) and it is foreseen to reach 80% by 2015 (BIO Intelligence Service, 2011).
- **Ozone depleting substances (ODS)**: The recovery of ODS from insulation foam prior to appropriate treatment of other materials is mandatory 'if technically and economically feasible'. Given the limited field experience to date at recovering construction foam, it is difficult to assess the economic feasibility with certainty. According to BIO Intelligence Service (2011), the cost of destruction of ODS with prior recovery from sandwich panels (possibly the most feasible type of construction foam to recover) is estimated at €83/kg

of ODS, around 8€ per kg of foam, with the assumption that the ODS content is 10%. Costs without prior recovery of the blowing agent are likely to be significantly lower (at least by €20/kg of ODS). Costs for recovery of ODS from board stocks, spray foam or XPS foam are likely to be much higher, as this material is not easily identified and sorted out during demolition.

- **Rock wool**: Another smaller fraction is represented by rock wool that is collected through a system of payable bags or containers (with large quantities of waste). The collection system is not free but competitive with traditional ways to dispose of the waste. In the latter case, the rock wool waste is likely to end up in landfills. It is possible to cut residues and used rock wool qualifies for recycling, if the waste is dry, chemically clean and free of other wastes such as packaging material, plaster, etc. The recycled material can be transformed into new products through heat treatment in furnaces.
- **Quality of secondary raw material**: As highlighted earlier, recycled aggregates must have the same quality of natural aggregates and must fulfil the same technical specification and is also related to the following considerations.
- **On-site or off-site treatments**: The choice as to whether a treatment should be done on- or off-site is complex and depends on many factors. The main advantages of on-site treatment are lower materials handling and transport costs and lower machinery CC, while the disadvantages are related to space demands for materials and machinery in the civil yard, more local noise and dust nuisance and less flexibility for the employment of recycled materials. However, advantages of off-site treatment are related to an easier reduction of adverse environmental impacts on surrounding areas of the plant, more practical use of a wider range of higher capacity equipment, lower machinery operating costs per treated tonne, better success in the separation of organic materials (paper, wood, plastic, etc.) and lightweight materials, an easier control of the quality of recycled aggregates and better products marketing. In the latter case, disadvantages are proper control of the demolition process that is essential to guarantee C&DW quality, higher materials handling and transport costs, higher machinery CC and fixed costs. In order to choose between an on- or off-site treatment, it is suggested to focus on the required quality of recycled aggregates for the fulfilment of the requirements of the intended use. If selective practices are introduced during the demolition phase and if a very homogeneous stream of C&DW is treated (i.e. concrete stream, bricks and tiles stream), an on-site treatment could be a good choice.
- **Recycling technology**: The production of high-grade recycled aggregates employable in high demanding uses is almost achievable by means of wet separation processing sections. This separation process is nowadays performed in stationary plants. Only the aim of producing recycled aggregates for high

demand employment, such as concrete, could pay the costs related to the waste water treatment. Also the opening of the aggregates market to recycled materials could be beneficial. Some studies are under way in Italy (Garbarino *et al.*, 2007), the Netherlands (within the EU project 'C2CA Advanced Technologies for the Production of Cement and Clean Aggregates from Construction and Demolition Waste') and in Flanders (BIO Intelligence Service, 2011), in order to design mobile or semi-mobile wet processing plants, following similar techniques employed in soil recycling equipment.

6.4 Cost factors of the end-of-waste criteria implementation

6.4.1 Conditions for end-of-waste criteria and costs associated with the removal of the waste status

The 'Thematic strategy on the prevention and recycling of waste' (EC, 2005) proposed to clarify when a waste might cease to be a waste and can be considered as a recovered material and freely traded on the open market. In this respect, the revised WFD contains provisions that could enable the EC to propose implementing measures to set EoW criteria for some specific waste streams. These conditions concern the use of substances for a specific purpose, the existence of a market, the respect of technical requirements and standards and the protection of human health and the environment. One of the streams under study is represented by recycled aggregates from C&DW. In detail, C&DW are commonly used as input material in the production of recycled aggregates and a market already exists.

The European standards for aggregates differentiate primary, recycled and manufactured aggregates, and have to fulfil the same technical requirements in order to be used as aggregates in the European common market. The European standards cover not only principles for guaranteeing a safe use of the construction material, but also the release of dangerous substances from the material to the environment and indoor air. According to the overall adverse environmental or human health impacts, in general, recycled aggregates present little risk to the environment.

However, the most relevant issue from the environmental point of view is the release of substances from the recycled materials to the environment due to the dissolution of substances and their transport to the soil and water. In this case, the definition of the EoW leaching requirements to be met by recycled aggregates would provide a guarantee that no additional environmental impact will occur when recycled aggregates cease to be waste. The definition EoW leaching requirements is one of the key aspects and is a task that JRC is carrying out in order to clarify and define EoW pollutant limit values for recycled aggregates to cease to be waste. This is a complex issue due to the fact that there are different test methods and approaches to the same problem among MS.

The rationale for setting EoW criteria can be summarised in the following steps:

- **Harmonisation and clarification of the legal status**: In some MS, recycled and secondary aggregates retain their waste status, whilst in other countries these aggregates are not wastes. In addition, recovery rules differ between MS and this hinders the marketing of the recycled materials between countries. The legal uncertainty associated with waste definition also inhibits the investment in waste management facilities. A clear definition of rules for the recovery of waste would create a solid base for the development of more recycling centres and for the increase of recycling rates.
- **User perception**: EoW criteria would help to improve confidence in the recycled products by ensuring that these products fulfil technical and environmental requirements that guarantee safe use. The removal of the waste status and trading the materials as a product would improve public perception and consumer acceptance of recycled aggregates.
- **Unnecessary burdens associated with the waste status**: Associated with the waste status are all the administrative procedures needed to ensure proper control of the material. Typically, the use of recycled aggregates is done on a case-by-case basis, which makes a quick response to the market demand difficult. These procedures increase the final cost of the recycled products that compete with natural aggregates, thus creating a potential barrier to the material recycling.
- **Environmental benefits**: The establishment of EoW criteria, which do not entail an environmental risk, would overcome these ambiguities, promoting the reuse and recycling of C&DW. With the removal of the waste status, this scenario is maintained. Recycled aggregates that cease to be waste would have to meet these extra requirements, in addition to the environmental requirements imposed by the criteria.

For recycled materials that do not meet EoW requirements, finding a market will be more difficult. The competition with natural aggregates plus recycled and secondary aggregates that are products, together with the controls due to their waste status, will make it harder for these materials to enter into the aggregates market. This could lead to efforts to improve the product quality, the processing and the source separation of the input material, in order to obtain a product that meets EoW requirements. Recycled aggregates are often used in lower-grade types of applications such as engineering fill and road sub-base. In some countries, with well established recycling practices, the use of recycled aggregates in more demanding applications already exists, because the market is saturated with lower-grade types of material.

As an overall assessment, it can be said that a significant positive economic impact will be associated with the removal of the waste status. In the short term, the investment is substantial, but it needs to be evaluated in the long term. Costs associated with the fulfilment of EoW criteria are related to the costs for the selective demolition and the general control of each stream. Moreover, EoW requirements establish that the production of the recycled aggregates must be covered by a quality assurance system and in many MS, quality assurance systems are not so well implemented or need to be upgraded.

According to EC *et al.* (2009), the costs associated with the administrative procedures related to the waste status could reach 1% of the turnover of the recycling sector. The costs associated with these procedures will be reduced once the recycled material fulfils EoW criteria. The transport and use of the recycled material is done as a product, with no waste controls.

6.5 Future trends

The promotion of recycling and the encouragement of SSM policies are two of the main challenges that need to be addressed in the EU-27 in the near future. SSM would benefit from an increased recycling rate and an integrated management of quarrying, processing and recycling. However, recycling rates are still low in several EU MS, and there is still scepticism among conventional natural aggregate producers about the opportunity to extend their business by integrating recycled aggregates production. C&DW has been identified by the EC as a priority stream because of the large amounts that are generated and the high potential for re-use and recycling embodied in these materials. Indeed, proper management would lead to an effective and efficient use of natural resources and the mitigation of the environmental impacts to the planet.

The 70% target of C&DW recycling by 2020 imposed by the WFD can be a tremendous driver towards SSM. However, one of the most important barriers for the achievement of the 70% target by 2020 is the incomplete and inconsistent information available in statistics dealing with both generation and recycling of C&DW. Based on highly uncertain and incomplete statistics on potential sources of unconventional aggregates, it can be argued that, in the medium term, the contribution of unconventional aggregates in SSM will dramatically increase.

The new waste regulation restrictions, the reduced land use availability for extractive activities, and the reduction of the available natural sources of aggregates are some factors that will dramatically change aggregate supply by the year 2020. This change will subsequently require major modification in the entire construction sector that has to adapt and ensure supply of aggregates. Currently, about 6% of the total aggregate demand (out of the overall 6 t per capita demand) in the EU-27, is covered by C&DW recycling. The experience gained from countries that recycle more than 70% of their C&DW, reveals that these recycled aggregates are sufficient enough to cover about one-quarter of the market's demands. That means that the necessity for new aggregates will still be valid;

however, it is clear that the C&DW recycling industry will grow significantly in the next decade (likely from 6 to 25% demand for C&DW recycled products), especially in countries that, at present, report low C&DW recycling rates.

Therefore the following questions need to be answered:

- Is the construction sector ready and resilient enough to adapt to this new business environment?
- Are European regions prepared to assist regional industry to confront this challenge?
- Are European regions ready to exploit this opportunity and develop smart specialisation strategies and ensure their sustainable and smart growth?

A key issue is the correct evaluation of what can be the effective contribution, in qualitative and quantitative terms, that secondary/recycled aggregates can supply in order to satisfy the requirement for building aggregates. A careful evaluation, based on technical, economic and environmental criteria, must be carried out in order to better understand the role of aggregates from conventional and unconventional sources. Only when it will be proved that the recovery/recycling process is both economically and environmentally sustainable, in comparison with the production of natural aggregates, can the contribution of recycled aggregates be considered as net positive.

6.6 Acknowledgement

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7
Conventional demolition versus deconstruction techniques in managing construction and demolition waste (CDW)

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Abstract: Technological aspects of conventional demolition/deconstruction are discussed, and an economic analysis is performed on a case study, directly comparing these two options. This comparison illustrates that although deconstruction is not yet competitive with conventional demolition, within the conditions established in the study, some deconstruction scenarios do present economic advantages. In environmental terms, from a simplified life-cycle analysis (LCA) approach, it can be concluded that, with current techniques and transportation methods, only significant separation efforts that actually result in re-use or recycling of bulk aggregate materials may lead to sizable environmental impact reductions, compared to a conventional demolition scenario.

Key words: conventional demolition, deconstruction, cost analysis, environmental analysis.

7.1 Introduction

Within the building demolition market, deconstruction as a process of pulling buildings down is gaining momentum, partly for environmental reasons (materials re-use, recycling), but mainly for economic reasons. An important driver is regulation. In 2008, for example, Regulation DL46/2008 took effect in Portugal. It requires separation of construction and demolition waste (CDW) at source or, when such an operation is not possible, the routing of such waste to registered operators who are, in principle, capable of performing the necessary separation tasks. These waste operators set their prices according to waste contamination and density, varying widely within the various types of CDW entering the operator facility. Delivering CDW to these waste operators in Portugal (excluding its islands) costs from 5.5 €/ton for clean aggregates up to 129 €/ton for contaminated CDW (not necessarily with hazardous materials), assuming bulk loose densities below 200 kg/m^3. Given these costs, it becomes more attractive to deliver this waste to the operator after some sort of initial source separation (and with as few contaminants as possible), which provides an incentive to the use of deconstruction techniques.

Deconstruction in Portugal has been studied (Lourenço, 2007; Santos and de Brito, 2007; Sousa *et al.*, 2004) and some companies have initiated activity in this area. Economic conditions exist in Portugal for deconstruction to be a profitable activity, although not all materials taken from demolished buildings are recycled, much less reused. Deconstruction in other countries/regions has also been studied, covering its technical, environmental and economic implications (Dantana *et al.*, 2005; Guy, 2003, 2005; ITEC, 1995; EPA, 2008; Southworth, 2009; Roussat *et al.*, 2009). However, deconstruction is seldom practiced in Portugal, mainly because of lack of information concerning implemented examples, as well as unattractive economic conditions.

Demolition in Portugal represents a niche within the wider construction industry. To change procedures and methods for most demolition contractors implies significant changes within what is a highly conservative industry. However, partial or total deconstruction is starting to be recognized by these companies as beneficial, in economic and social terms (Chini and Nguyen, 2003).

Local economic conditions can vary significantly, directly affecting deconstruction viability. As a consequence, different quantities of CDW are kept out of landfill. Local conditions are usually related to labour costs, tipping fees and market value of the recovered materials. Several studies have analysed the economic viability of these activities (Lourenço, 2007; Dantana *et al.*, 2005; Guy, 2005; EPA, 2008; CIB, 2005; WasteMatch, 2004) and have drawn different conclusions. However, in environmental terms, there is a clear advantage in pursuing deconstruction methods (Southworth, 2009; Roussat *et al.*, 2009), mainly due the lower mass of materials sent to landfill. A complete LCA of such systems, analysing the environmental impacts of different building end-of-life options, has rarely been performed. Those studies (Lasvaux *et al.*, 2009; Seppo, 2004a,b) generally have not focused on the particular demolition method applied, although they have considered environmental impact quantification concerning the building end-of-life cycle stage. Still, deconstruction operations have been the object of a certification proposal (Guy, 2003), in an attempt to reduce construction environmental life-cycle impacts, while bringing benefits to local economies and societies.

Deconstruction techniques can vary widely, depending, for instance, on construction type, available labour, equipment and work deadlines (Guy *et al.*, 2003; ITEC, 1995; CIB, 2005). These techniques will affect the quantity and quality of recovered materials, as well as the economic viability of the whole operation. Safety issues are also specific to deconstruction, which do not occur in traditional demolition, such as working in high places, protection from falling objects, and fatigue (Guy *et al.*, 2003). Work duration and schedule must be carefully taken into account in deconstruction operations when compared to traditional demolitions.

Generally, deconstruction operations are unique cases, each job displaying its own idiosyncrasies, though several common aspects must always be taken into account (Chini and Nguyen, 2003):

- obtaining a license (which is compulsory for both traditional demolition and deconstruction activities);
- estimate of noise and dust production during working time;
- hazardous materials handling;
- temporary structures allowance (e.g. scaffolding);
- neighbourhood (of particular relevance in deconstruction, when the recovered materials are to be sold in the local economy).

Buildings projects and urban policies can significantly help deconstruction activities, the challenges of which are:

- Buildings are not usually designed to be deconstructed;
- Building elements are generally not deconstructable;
- Specific deconstruction tools are not always available;
- CDW landfill fees are usually low (which usually favours traditional demolition practices);
- Deconstruction takes longer to execute than traditional demolition;
- Building and construction materials regulations seldom consider the possibility of re-using recovered materials in new constructions;
- Cost items in deconstruction can be hard to estimate;
- There is not enough accumulated experience of deconstruction techniques;
- Economic, social and environmental benefits of deconstruction, with its higher material recovery and subsequent re-use, are not well understood.

7.2 Technological aspects of demolition

Obtaining a licence to pursue building demolition activities is compulsory in many regions. In Portugal, it is usually necessary to deliver to the municipality in charge the labour accident insurance policy, demolition contactor certification form, jobsite data log, worksite coordination responsibility statement, and the health and security plan. These are administrative documents which do not specify the nature of the demolition work nor the specifics of the method applied; clarification of the latter is the purpose of the design project, which is not currently compulsory in Portugal (da Costa, 2009).

Demolishing structures can depend on many factors such as the type of structure, its condition, construction materials present, building height, building base plant area, surrounding available area, job deadlines, weather conditions and CDW management (Fueyo, 2003; AEDED, 2008). Demolition methods are chosen as a function of cost and the availability of equipment to the demolition contractor (da Costa, 2009).

Demolition methods can be divided in the following categories (de Brito, 1999):

- **Mechanical processes**: mechanical equipment is used to crush or bring down constructions.

- **Thermal processes**: materials are fused and separated from the structure.
- **Abrasive processes**: constructed elements are cut into smaller pieces using abrasive saws.
- **Explosives**: structures are collapsed using explosive charges at strategic points of the structure.
- **Electrical processes**: electrical discharges are conducted through the elements causing breakage.
- **Chemical processes**: use of highly expansive chemicals, which cause fragmentation of constructed elements.

7.2.1 Mechanical processes

There are a wide range of breaking and cutting devices available to demolition contractors, which can be held (smaller and/or lighter jobs), or fitted onto reaching arm excavators (larger and/or heavier jobs). Such devices include (da Costa, 2009):

- manual electrical hammer;
- manual pneumatic hammer;
- manual hydraulic hammer;
- manual (liquid) fuel hammer;
- excavator fitted hydraulic hammer;
- pounder hammer (fitted in excavators or special purpose cranes);
- smashing ball;
- hydraulic scissors (with independent hydraulic system);
- mechanical scissors (fitted onto an excavator hydraulic system);
- steel cutters;
- multi-processing scissors;
- hydraulic forceps;
- demolition jaws.

There is also the possibility of using hydraulic wedges which, through hydraulic pressure, generate traction forces within concrete masses, breaking them up.

7.2.2 Thermal processes

Applications of thermal demolition processes are somewhat specialized, since very high temperatures are not usually necessary for practical demolition purposes (e.g. general steel fusion temperature is 1600 °C). They therefore tend to be applied in special structures, such as nuclear power plants, and at jobsites which are difficult to access. According to Manning (1991), thermal demolition processes can be divided into three main categories:

1. drilling and thermal cutting using torches, plasma or lasers;
2. concrete removal through heating of reinforcement steel bars;
3. surface concrete removal by direct heat application.

7.2.3 Abrasive processes

Demolition by abrasion is usually done with very hard materials such as diamond or carborundum, or with highly pressurized water. Normally these methods are used in partial demolitions, as total building demolition could become cost prohibitive. Cutting with diamond can be performed with discs, strings or by coring. Diamond discs can cut through concrete and steel (reinforced or pre-stressed) with a 40 m^2 cutting surface yield. These cutting discs are usually water refrigerated, although there are also waterless diamond disc cutting machines on the market (da Costa, 2009).

Diamond strings are mostly used with granite, marble and concrete. String cutting is generally more efficient than using discs. According to Fueyo (2003), cutting yields using diamond strings can be between 3 and 5 m^2/h, while the string velocity can be as high as 40 m/s.

Finally, coring can be used to obtain samples, slabs or beams, drilling for crossing pipes, or to complement other demolition methods.

Demolition with highly pressurized water (Hilmersson, 1999) is a high yielding technique. It does not damage the overall structure, does not produce dust, vapour or slag, has no induced vibrations, has small reaction forces, and has a vast application range. However, it cannot generally be used to cut through reinforcement, and cracks can slow down progression.

7.2.4 Explosives

There are two main categories of explosives for demolitions, military or commercial. Military explosives have generally higher detonation speeds, between 6000 and 9000 m/s, and include the following substances: TNT, RDX and PETN. Commercial explosives, mostly dynamite based, have detonation speeds between 3000 and 7000 m/s. Explosive demolition demands expertise and detailed structural knowledge (Sánchez, 2009). Three main types of structural collapse can be expected: implosion (also called telescope), push and gradual collapse.

7.2.5 Electrical processes

Although not commonly used in demolition, there are some niche applications in which electrical process methods have advantages. One such form uses shape memory alloys, materials which regain their original shape when heated (in this case, by electrification). Another method works by heating a ferromagnetic material which will then expand and, in doing so, cause cracking. A final electrical process uses water as a medium for propagating electricity, producing shock waves. This phenomenon breaks the interfaces between aggregates and the cement paste and is particularly applicable for underwater structures (Linß and Mueller, 2003).

7.2.6 Chemical processes

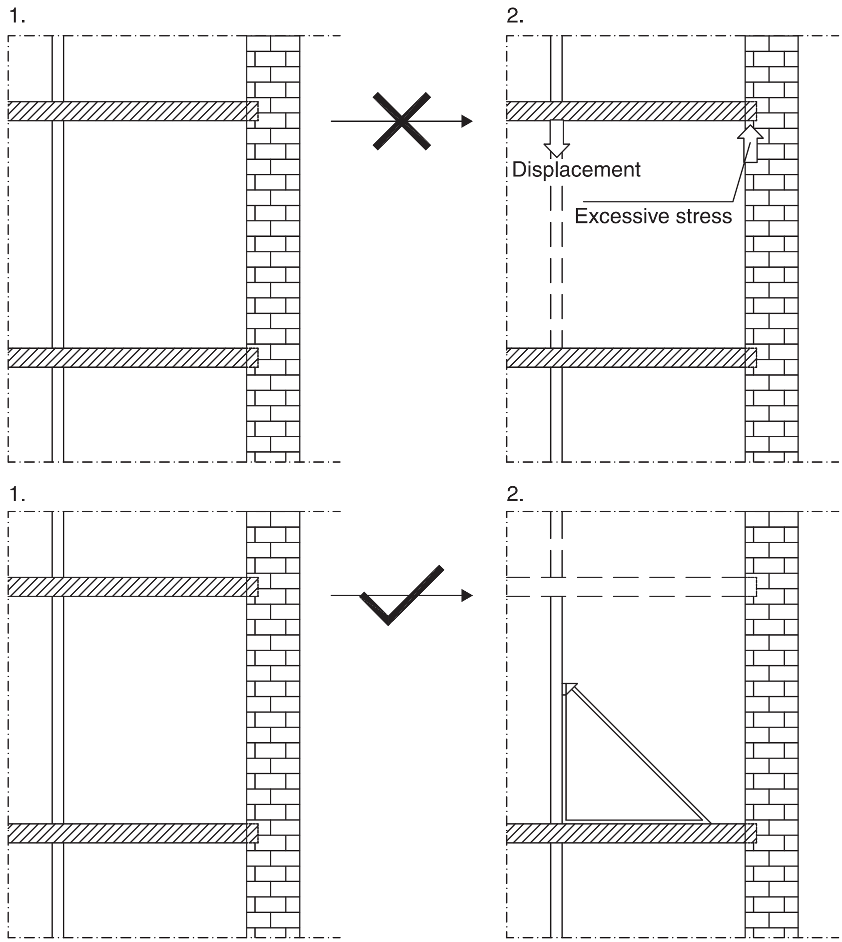
Chemical demolition uses the expansive properties of certain chemical reactions which can, under specific conditions, generate sufficiently high tensions to crack concrete elements. One such method uses pressured gas – normally CO_2 – which can burst concrete elements from within (although it is less efficient with reinforced concrete structures). Another method uses quicklime in a similar manner. However, it is a slow process in which it is not possible to control the cracking pattern.

7.3 Technological aspects of deconstruction

Deconstruction is basically the construction process in reverse. It is usually applied when re-use and recycling of construction elements is important for environmental, economic or social reasons. Basic aspects to consider include (ITEC, 1995):

- formal communication to all entities that may be affected by or have jurisdiction over the deconstruction activity;
- deconstruction area setup;
- disconnection of all services still active in the building, such as water, electricity, gas supply, as well as the collection or flushing of all fuel tanks still present in the area;
- bracing construction elements that may collapse if their internal stress state changes significantly;
- erection of scaffolding;
- preparation and execution of personnel safety measures;
- routing and separate storage for recovered materials;
- workers' individual protection measures.

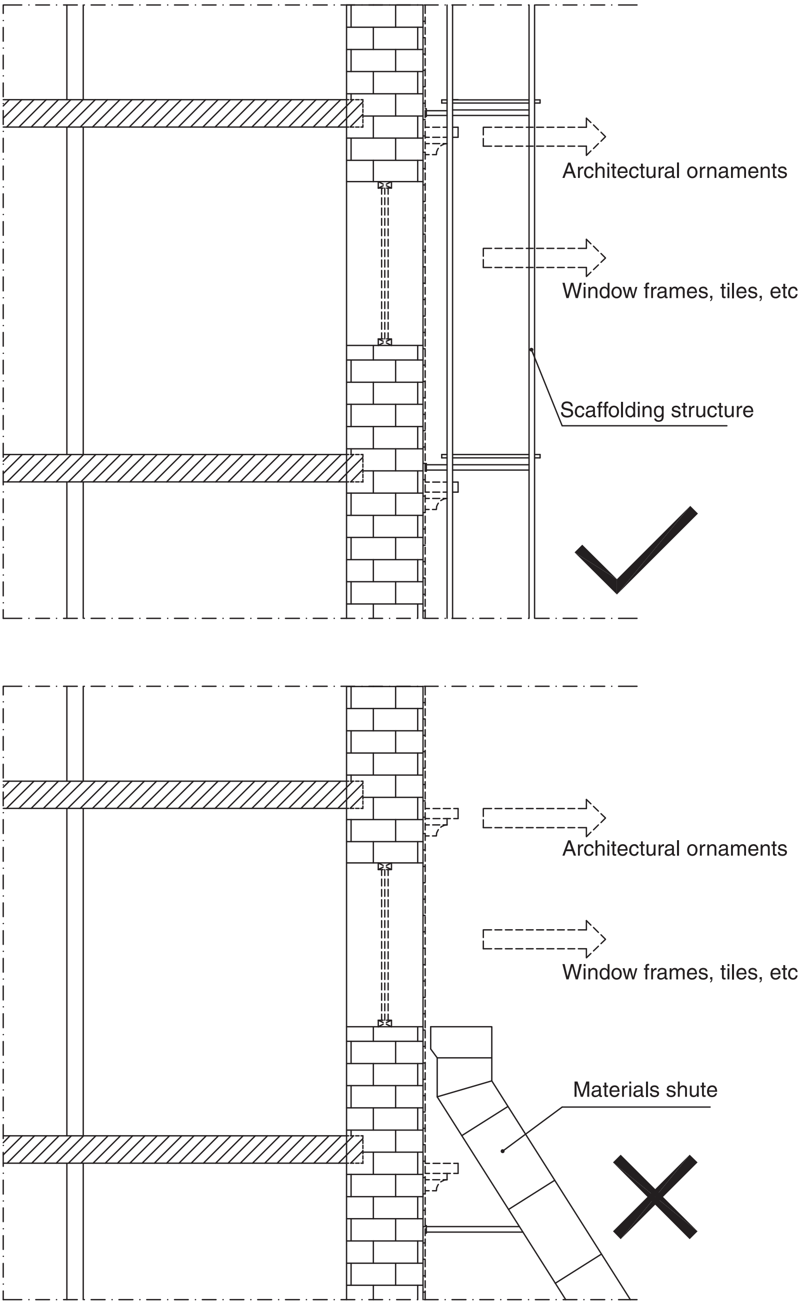
The first three aspects are common to all demolition works. However, bracing systems in deconstruction jobs are usually related to sudden stress changes in structural elements, and may be necessary to avoid unexpected structural failure (Fig. 7.1). Scaffolding is necessary to withdraw façade elements or if certain elements must be sent directly out of the building (Fig. 7.2). To ensure maximum material recovery, both for re-use and recycling, job site routing is important. The least possible damage to recovered elements will ensure a maximum resale value. Following a reversed construction logic, all covering elements shall be taken out



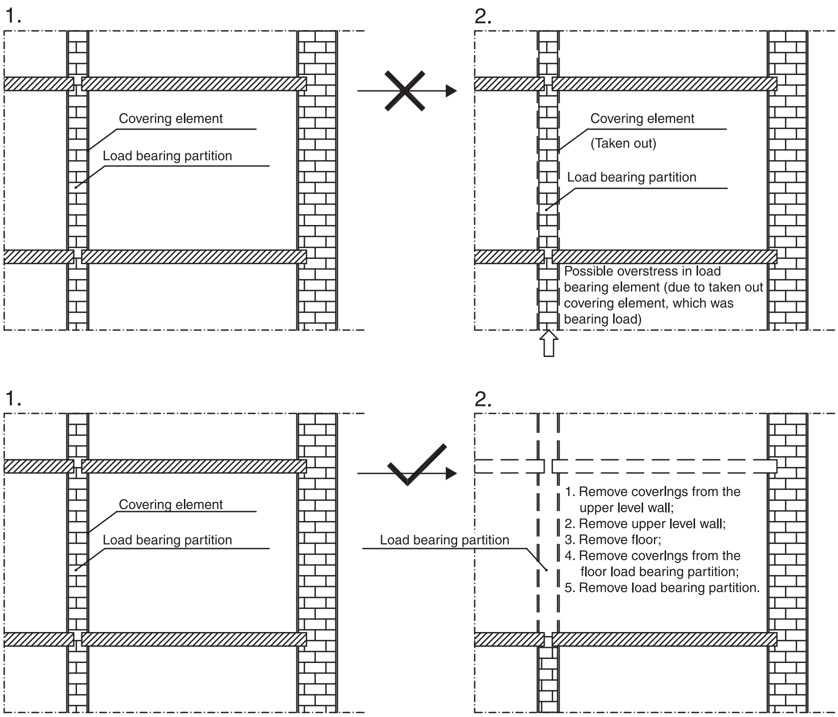
7.1 Deconstruction sequence of possibly loaded partitions.

from the outer layers down to support elements; however, if certain envelope elements are loaded or if there are doubts about their stress state, then they must not be taken out before the upper levels have been entirely deconstructed (Fig. 7.3).

Some specific techniques have been developed for deconstruction work, particularly relating to wooden buildings or building elements, since these are in some regions (e.g. USA) the structures most often targeted for demolition/ deconstruction. One such technique is called panelization, and consists of cutting entire building element sections, while suspending them, and then transporting them to a lower, more convenient plane (usually the ground, although it may be an intermediate floor). This requires use of heavy mechanical equipment, such as cranes or excavators equipped with hooks, but considerably speeds up the



7.2 Deconstruction of façade elements, for re-use (with scaffolding).



7.3 Deconstruction sequence of load bearing partitions, whose covering elements might be loaded.

deconstruction process (compared to a full hand in-place deconstruction) and enhances general safety conditions, since subsequent work can be done at ground level without risk of materials or personnel falling.

Also, controlled collapse can be a convenient way to speed up the process, a technique called dropping, since manual deconstruction can again be performed at a convenient levelled surface for workers, although there is a possible loss of quality in the recovered materials (Guy, 2005). These techniques can also be applied to wall sections if they are light and stiff enough to be suspended by a mechanical fork or crane; again the purpose is to bring whole constructed sections down to ground level where manual deconstruction can be faster and safer, while continuing building panelization with mechanical means (US ACE, 2007).

Other methods, such as punching, can be used to withdraw inner wall coverings from the outside, or from the opposite wall surface. Punching can also be performed with mechanical means, for instance with an excavator equipped with a fork or with a backhoe. This technique's advantage lies in the quick release of these inner covering elements (e.g. wood boards, gypsum plasterboards), especially if these elements are not targeted for re-use (only recycling). It can also

7.4 Punching out of windows, using heavy mechanical equipment.

be applied to floor coverings with some adaptation. A punching variation was used in Fig. 7.4, where all the windows were extracted from the outside using an excavator arm equipped with a claw. The latter represents an example of crude separation of materials to obtain glass and wood for recycling.

Tables 7.1 to 7.3 list and briefly describe hand tools, mechanical or electrical, as well as heavy equipment associated with deconstruction work.

7.4 Demolition versus deconstruction: economic analysis

During recent years, deconstruction has been studied, generating reliable economic, physical and environmental information. However, particularly concerning economic data, conclusions differ significantly between regions as parameters such, as salaries, tipping fees and/or material resale prices are highly variable. In a free market social environment, the decision to undergo demolition or deconstruction is an economic one, within the legal constraints specific to a given region. As a consequence, this study focuses on the economic evaluation of both demolition methods, using a case study located in Portugal. Several scenarios are evaluated, based on several possible options for waste management applicable to this case, some of which have overall costs lower than those incurred with traditional demolition.

Table 7.1 General and specific manual deconstruction tools

Manual tools (purely mechanical)		
Professional designation	Example image	Function/notes
Wrench adjustable		Bolt loosening, metal elements separation.
Bow saw		Precise cut of small wooden elements.
Cats paw		Crow bar type of tool, usually in small sizes, to complete small chores or to work in tight places.
Fiberglass ladder		Lightness, stability and flexibility, all essential properties for a swift and safe deconstruction work.
Gorilla bar		Crow bar type of tool, usually in large sizes, to complete heavy tasks, as lifting metal plates or stone elements. It can reach 90 cm in length.
Bow saw with traction		Basically a bow saw but able to receive force from both arms. A faster, more intense sawing.
Hammer		One of the most basic tools, both in construction and deconstruction, and also one of the most necessary. Specifically, in deconstruction, it is used to separate elements, pluck nails or remove obstacles out of the way.
Hand saw		Flexible cutting in several wooden element types and shapes.

(*Continued*)

Table 7.1 Continued

Manual tools (purely mechanical)		
Professional designation	Example image	Function/notes
Pick axe		Basic tool for breaking stone or brick sections, or dig ditches for infrastructure.
Pliers		For cutting cables, dismantling of metallic and plastic elements.
Crow bar		The traditional lever, used for prying. Particularly useful for lifting wooden floors, several types of coverings, plasterboard, among other construction elements.
Rakes		Cleaning and levelling of the deconstruction surrounding area.
Sledgehammer		Heavy demolition (at the human scale), for instance brick or stone walls. If not a tool for material recovery for re-use, it can still be useful when recycling is an attainable goal.
Tin snips		Easily cuts metal coverings.
Vise grips		Allows a tight grip onto several objects. Very useful to unscrew bolts (it holds the fixed end).

Table 7.2 General and specific light mechanical/electrical deconstruction tools

Manual tools (electrical)		
Professional designation	**Example image**	**Function / notes**
De-nailing gun		A specific deconstruction tool. From the de-nailing gun barrel a small ram is fired, which punches the nail out. Each shot moves the nail 5 cm (long nails might require more than one shot).
Drill, cordless with batteries, and battery changer		A step further from the traditional wired drill, plugged into the socket. It enhances movement flexibility and the safety of its use. In deconstruction jobs, it can be used to place safety barriers and to prepare removed elements for re-use.
Metal detector		If re-fabrication of wooden elements is intended, a metal detector is precious. Nails or other metal fasteners which inadvertently have been left on wooden elements targeted for re-fabrication will inflict great wear on wood cutting and grinding machines, contributing to their quick deterioration.
Pneumatic or electrical hammer with chisels		Extremely useful for punching and cutting through stone or concrete elements.
Post-hole digger		For a quick and safe fastening of fencing posts around the deconstruction area.

(*Continued*)

Table 7.2 Continued

Manual tools (electrical)		
Professional designation	**Example image**	**Function / notes**
Chain saw		Gross cutting of wooden elements (small, medium or large sizes).
Sawz-alls with bi-metal blades	 	Equipped with wood, metal, stone, ceramics or concrete cutting blades, there is virtually no material which cannot be cut. Both in fixed table or hand versions, it is extremely useful to weaken elements (in order to recover them more easily), correct their shape and cut them into appropriate dimensions for storage and sale.
Hydraulic scissors		A precision demolition unit, for metal, plastic, wood, brick and concrete.

7.4.1 Case study approach: characterization

The case study is located in Cacém in the outskirts of Lisbon, Portugal, and refers to approximately 13 430 m^2 of building area to be removed, with construction dates within the first half of the 20th century. Subject to an urban regeneration program, buildings in this area, of medium to low construction quality and around 100 m^2 in size, are to be removed using deconstruction methods. The company responsible for executing this contract is well known in the demolition industry and has applied a mixed method of deconstruction (for recycling purposes) and demolition, in order to minimize costs and maximize work speed, while separating most of the construction materials. The deconstruction part of the work can be seen as a soft stripping activity, where most of the indoor covering materials were removed separately, such as floor coverings (carpets, cork tiles), gypsum cardboard, wood floors, doors, windows, furniture, sanitary appliances and wooden stairs. The

Table 7.3 General and specific heavy mechanical/electrical deconstruction tools

Mechanical equipment		
Professional designation	Example image	Function/notes
Telescoping boom hi-lift		Allows the transportation of panelized roof or wall sections onto ground level, enhancing deconstruction efficiency. It can also be used to perform deconstruction tasks at high places, as in the removal of glass façades or wall covering surfaces.
Excavator accessory – hydraulic metal scissors		For cutting large metal sections, aiming for recycling (not to be used if re-use is an aim). Particularly useful to separate welded elements.
Excavator accessory – hydraulic multi-purpose cutter		Possible use for cutting large wooden elements, also adaptable for steel and concrete sections. Cutting precision is obviously low, but it allows great element recovery speed for recycling, or partial re-use. It can be used for panelization.
Fork lift		A mobile lifting fork is of particular use to pull up roof sections, floors and even walls.

(*Continued*)

Table 7.3 Continued

Mechanical equipment		
Professional designation	Example image	Function/notes
Mini excavator		From levelling terrain down to lifting wooden floors, plus helping in piling recovered materials (i.e. with the fork accessory), this is definitely one piece of equipment which can really contribute to operation efficiency and reduction of working time and costs.
Excavator – carrier		When the mini excavator capacity is exceeded, or by other reasons desirable, a larger carrier could be used. It will represent higher efficiency in material transportation, mainly in medium- to large-size deconstruction contracts.

demolished materials were the heaviest and most difficult to separate, including stone, ceramic tiles, concrete mixed with small amounts of ceramic, lead plumbing, and copper and plastic electricity cables. However, from the demolished piles, efforts were made to recover some materials for recycling, namely wood and metals.

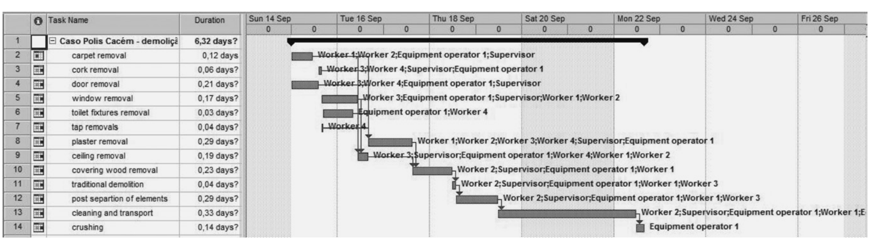
Labour organization for this operation was of a traditional nature in construction contracts, with non-specialized workers performing the soft stripping and scavenging for wood and metals in the demolished piles, equipment operators for loader and excavator manoeuvres, and supervision for technical control and personnel organization.

7.4.2 Execution time

Execution time was calculated for the 100 m^2 house unit, both for conventional demolition and deconstruction. This was calculated considering the number of workers, work performance data supplied by the contractor, and other data collected on-site. For deconstruction, calculation was made using a standard project management tool, using compiled data (Tables 7.4 and 7.5). From this data, using the task management software, a task flow map is presented (Fig. 7.5). This flow map shows that the necessary deconstruction completion time is around 6.5 days. Dismantling the same 100 m^2 standard housing unit using a traditional demolition technique would take only 1 day (Table 7.6) and as confirmed by the contractor. The following information was taken into account in these calculations:

Deconstruction:

- carpet removal refers to 40% of floor area;
- plaster removal refers to ceilings (100%);
- cork tiles removal refers to 15% of floor area;
- window removal implies wood and glass separation;
- within the present deconstruction process, traditional demolition activities refer to:
  - stone and wood walls;
  - concrete, stone or wood floor coverings;
  - roofs (wood and ceramic shingle elements);
  - water supply and drainage networks (ceramics and lead);
- traditional demolition sub-task takes 1 h (for a 100 m^2 standard size housing unit);
- traditional demolition sub-task labour structure (5 workers):
  - one excavator operator;
  - one loader operator (to remove still recoverable metals and wood);
  - one hose operator;
  - two unskilled workers to scavenge the rubble and recover hand-size wood and metal elements;



7.5 Task flow management chart for deconstruction activities (for a 100 m^2 standard size unit).

Table 7.4 Data for calculating the case study deconstruction cost and duration (per standard housing unit m^2)

Activity									Productivity					Operation order
	Element area	Labour						Units	Calculated			Contractor		
	m^2	Unskilled worker	Hours	Equipment operator	Hours	Supervisor	Hours	Number	Man.h/m^2	Man.h/un	Man.h	Man.h/m^2	Man.h/un	
Carpet removal	40	2	3	1	0.4	1	0.1		0.163		6.5	0.16		1
Plaster removal	100	4	7	1	1	1	1		0.300		30	0.30		2
Covering wood removal	76.6	2	5.5	1	0.65	1	0.6		0.160		12.3	0.16		3
Cork removal	15.0	2	1.5	1	0.1	1	0.1		0.213		3.2	0.20		1
Door removal	22.6	2	5	1	0.5	1	0.5		0.486		11	0.50		1
Window removal	19.3	3	4	1	0.5	1	0.5		0.673		13	0.66		1
Ceiling removal	100	4	4.5	1	1	1	1		0.200		20	0.20		2
Bathroom fixtures removal		1	1	1	0.3			12.6		0.103	1.3		0.1	1
Taps removal		1	1					9.7		0.103	1.0		0.1	1
Conventional demolition	100	3	1	1	1	1	1		0.050		5.0			4
Post separation of elements	100	3	7	1	7	1	2		0.300		30			5
Cleaning and transport	100	2	8	2	8	1	2		0.340		34			6
Aggregate crushing	100			1	3.27				0.033		3.3			7

Table 7.5 Average zone use areas and total material weight (per standard housing unit m^2)

Kitchen + bathroom	0.23	m^2/m^2
Doors	0.23	m^2/m^2
Windows	0.19	m^2/m^2
Bathroom fixtures	0.13	un/m^2
Tap units	0.097	un/m^2
Total material weight	1.96	ton/m^2
Total demolished weight (for a standard 100 m^2 house)	196	ton

Note: These values were obtained from case studies, located in Lisbon. More details in Coelho and de Brito (2010).

- cleaning and transportation sub-task labour structure (5 workers):
  - one excavator operator (to crush large chunks into smaller pieces, allowing loading and transportation);
  - one loader operator (to place separated and mixed materials in the containers);
  - one truck driver;
  - two unskilled workers to scavenge the rubble and recover hand-size wood and metal elements.

Traditional demolition

- demolition sub-task takes 1 h (for a 100 m^2 standard size housing unit);
- labour structure:
  - one excavator operator;
  - one hose operator;
- cleaning and transportation sub-task labour structure (2 workers):
  - one excavator operator (to crush large chunks into smaller pieces, allowing loading and transportation);
  - one truck driver;
- without materials separation after the demolition sub-task, the cleaning and transportation stage is assumed to take half the time the same activity would take in the deconstruction context.

7.4.3 Partial costs

Five main cost categories were considered in the analysis: yard, direct and indirect labour, equipment, transportation and disposal. Cost scenarios were considered for the deconstruction option, but only in the transportation and disposal categories,

Table 7.6 Data for calculating the case study traditional demolition cost and duration (for a 100 m^2 standard size unit)

Task	Element area	Man labour						Productivity (calculated)		Operation order
	m^2	Unskilled worker	Hours	Equipment operator	Hours	Supervisor	Hours	Man.h/m^2	Man.h	
Traditional demolition	100	1	1	1	1	1	1	0.03	3	1
cleaning and transport	100	2	4	2	4	1	1	0.17	17	2
aggregate crushing	100			1	3.27			0.033	3.3	3

since the other categories are fixed. Waste operators are located at different distances from the worksite (which affects transportation costs) and charge different rates for accepting mixed or separated CDW loads (which affects disposal costs). Table 7.7 shows these differences for the several regional waste operators.

Yard

The yard cost was considered the same for the traditional demolition and deconstruction options, since its physical and cost arrangement is similar. From contractor data, and averaged over a 100 m^2 standard housing unit, a 154.9 € yard cost was considered.

Direct and indirect labour

Direct labour refers to skilled (e.g. sanitary pieces removal, heavy equipment manoeuvring) or unskilled work (e.g. scavenging, floor coverings removal), whereas indirect labour concerns supervision work regarding technical, worksite management or health and security related issues. For deconstruction activities, direct and indirect labour costs were calculated using the task management software (Table 7.4 and Fig. 7.5), resulting in 1212 € per 100 m^2 standard housing unit. For the same unit, traditional demolition would entail a 202.3 € cost (based on data given in Table 7.6), accounting for both direct and indirect shares. These values were based on current regional labour costs, as given by the contractor:

- **unskilled worker**: 6 €/h;

Table 7.7 Waste operators distance to the case study location and charges for receiving waste loads

Waste operator	Distance from the worksite, km	Waste load charges for different materials, €/ton	
		Separated aggregate	Mixed CDW
1	2	6.0	20.0
2	61	15.0	59.0
3	64	5.5	20.5
4	21.5	12.5	30.0
5	30	0	75.0
6	219		
7	30		
8	253		
9	8		

Note: Operators 6 through 9 do not accept aggregates (separated or mixed with other materials), or are specialized recyclers.

- **equipment operators**: 10 €/h;
- **supervisors**: 12.8 €/h.

Equipment

In this case study, equipment costs consist of excavator, loader and crusher (located away from the worksite) operation. According to the contractor, excavator and loader costs are, respectively, 65 €/ton and 30 €/ton. Crusher cost was derived from a fuel consumption estimate, based on a reference power capacity of such machines (in this case, around 100 kW), which amounts to 32 € per hour of operation. Total equipment costs are given in Table 7.8, for both traditional demolition and deconstruction processes considering a standard 100 m^2 housing unit.

Transportation

Transportation costs were calculated according to known distances between the worksite and the destination locations of the materials (separated or mixed CDW). The number of trips of a standard heavy truck (with a 19.3 m^3 loading capacity) was calculated according to material quantities and their loose densities (Table 7.9). Costs per transported kilometre are listed in Table 7.10, according to two data sources. For the deconstruction case, and considering the several possible destinations for removed materials, Table 7.11 shows total transportation costs from 1429 € up to 1865 € per standard 100 m^2 housing unit, dependant on the waste operators to which aggregates are routed. As for transportation with the traditional demolition case, routing mixed CDW can cost from 16 € up to 506 € per 100 m^2 standard house (Table 7.12).

Disposal

Disposal costs were based on waste operators information, as well as market current values (Table 7.13). From these market values (year 2010), global disposal

Table 7.8 Equipment costs for deconstruction and traditional demolition operations

	Deconstruction		Traditional demolition	
Equipment	Operation hours	Cost, €	Operation hours	Cost, €
Excavator	9	585	5	325
Loader	15	450		
Mobile crusher	3.3	64.8	3.3	64.8
Total	27.3	1100	8.3	390

Table 7.9 Materials loose densities

Material	Loose density kg/m^3
Wood	300
Gypsum plasterboard	350
Gypsum plaster	1000
Masonry brick	1400
Mixed CDW	1400
Ceramic aggregates	1464
Paper and cardboard	100
Plastics	13
Metals	900
Green wastes	150
Equivalent to municipal solid waste	150

Note: Values according to Construction Materials Recycler (2009) and Victorian Government (2009).

Table 7.10 Transportation unitary costs (€/km) for CDW

Material	Contractor	Lourenço (2007)
Separated aggregate (ceramics)	2.76	1.83
Mixed CDW		
Hazardous waste Non-contaminated gypsum materials	2.95	7.1
Wood Tyres	4.00	
Other waste		2.19

costs were calculated for both deconstruction and traditional demolition cases (Tables 7.14 and 7.15).

7.4.4 Total costs

Summing up calculated costs for each category, it is possible to provide global costs for the deconstruction and traditional demolition cases. However, global cost variations only reflect differences in transportation, disposal distances and market circumstances (operator charged fees) (Table 7.16). Table 7.17 presents average category cost percentages and respective standard deviations.

Global average costs amount to 5062 € for traditional demolition and 5818 € for deconstruction, per standard 100 m^2 housing unit (Fig. 7.6).

Table 7.11 Deconstruction transportation costs

Average material composition	%	kg	m^3	Destinations	Distance, km	Truck trips	Cost	
							€/km	€(/100 m^2)
Uncontaminated soil and rocks	51.1	13 463 531		Used locally				
Concrete, masonry, tiles, shingles and other ceramics (clean aggregates)	37.2	9 810 840	6701	Waste operator 5	30	347	2.76	213.9
				Waste operator 1	2			14.3
				Waste operator 2	61			434.8
				Waste operator 3	64			456.2
				Waste operator 4	21.5			153.3
Uncontaminated gypsum materials	4.47	1 178 883	1179	Waste operator 2	61	62	2.95	83.1
Wood	3.33	878 228	2927	Waste operator 6	219	152	4.00	991.5
Potentially hazardous materials	2.57	677 792	484	Waste operator 2	61	26	2.95	34.8
Municipal solid waste	0.71	187 250	1248	Waste operator 2	61	65	2.95	87.1
Metals (except lead)	0.49	129 229	144	Waste operator 7	30	8	2.19	3.91
Glass	0.1	26 373	76	Waste operator 8	253	4	2.19	16.5
Green waste	0.05	13 187	88	Waste operator 9	8	5	2.19	0.65
Plastics	0.02	5275	406	Waste operator 2	61	22	2.19	21.9
Total	100	26 370 588	13 253	Clean aggregate: waste operator 5				1453
				Clean aggregate: waste operator 1				1254
				Clean aggregate: waste operator 2		691		1674
				Clean aggregate: waste operator 3				1696
				Clean aggregate: waste operator 4				1393

Note: Distances are measured from real roadmaps, between the worksite and each waste destination.

Table 7.12 Traditional demolition transportation costs

Average material composition	%	kg	m^3	Destinations	Distance, km	Truck trips	€/km	€(/100m^2)
Uncontaminated soil and rocks	51.1	13 463 531		Used locally				
Mixed CDW	48.9	12 907 056	9219	Waste operator 5	30	478	2.19	233.9
				Waste operator 1	2			15.6
				Waste operator 2	61			475.6
				Waste operator 3	64			499.0
				Waste operator 4	21.5			167.6
Total	100	26 370 588	9219	Clean aggregate: waste operator 5				233.9
				Clean aggregate: waste operator 1				15.6
				Clean aggregate: waste operator 2		478		475.6
				Clean aggregate: waste operator 3				499.0
				Clean aggregate: waste operator 4				167.6

Table 7.13 Material waste flows disposal costs

Average material composition		Unit disposal costs €/ton
Uncontaminated soil and rocks		used locally
Concrete, masonry, tiles, shingle and other ceramics (clean aggregate)	Waste operator 1	6
	Waste operator 2	15
	Waste operator 3	5.5
	Waste operator 4	12.5
	Waste operator 5	0
Uncontaminated gypsum materials		55
Wood		0
Potentially hazardous materials		100.5
Municipal Solid Waste		60.5
Metals (except lead)		–150
Glass		0
Plant waste		3.5
Plastic		–50
Mixed CDW	Waste operator 1	20
	Waste operator 2	59
	Waste operator 3	20.5
	Waste operator 4	30
	Waste operator 5	75

Note: Negative values represent income.

7.4.5 Analysis of economic results

Figure 7.6 shows that, despite the fact that traditional demolition is still on average less costly than deconstruction, there are some circumstances in which this trend is reversed. As expected, the scenarios where deconstruction is less costly than traditional demolition are those where routing mixed CDW is more expensive. However, the case is reversed regarding separated aggregate disposal costs, being respectively the lowest (0 €/ton for operator 5) and the highest (15 €/ton for operator 2).

Table 7.17 also shows that the traditional demolition global cost is more sensitive to disposal costs than deconstruction, since the former is more sensitive to this cost category (75%) than deconstruction (28%), on average. However, the latter entails around six times greater labour costs, largely a result of the extra time needed to perform deconstruction activities as compared to traditional demolition (Table 7.16).

Traditional demolition is also more sensitive to transportation costs, since it usually depends on single waste material destinations, while in deconstruction a

Table 7.14 Deconstruction disposal costs

					Cost (per 100 m^2)					
					Waste operators		Deconstruction contractor (waste operator 5)		Revenue	
Average material composition	%	kg	Waste management option		€/kg	€	€/kg	€	€/kg	€
Uncontaminated soil and rocks	51.1	100 250		Reuse						
Concrete, masonry, tiles, shingles and other ceramics (clean aggregate)	37.2	73 052	Recycling	Waste operator 1	0.006	438.3	0	0		
				Waste operator 2	0.015	1096				
				Waste operator 3	0.0055	401.8				
				Waste operator 4	0.0125	913.1				
Uncontaminated gypsum materials	4.47	8778		Landfill	0.055	482.8	0.09	790.0		
Wood	3.33	6539		Recycling	0.00	0.00	0.015	98.1		
Potentially hazardous materials	2.57	5047		Landfill	0.101	507.2	0.075	378.5		
Municipal solid waste	0.71	1394		Landfill	0.061	84.4	0.075	104.6		
Metals (except lead)	0.49	962.2		Recycling					0.150	144.3
Glass	0.1	196.4		Recycling	0	0.00	0	0.0		
Plant waste	0.05	98.19		Landfill	0.0035	0.34	0.0035	0.3		
Plastic	0.02	39.28		Landfill			0	0.0	0.05	2.0
Total	100	196 356					0.259	1372	0.200	146.3
					Global cost, €					
					Waste operator 1		1367			
					Waste operator 2		2024			
					Waste operator 3		1330			
					Waste operator 4		1842			
					Waste operator 5		1227			

Table 7.15 Traditional demolition disposal costs

					Cost		Revenue	
Average material composition	%	kg	Waste management option		€/kg	€(/100 m^2)	€/kg	€(/100 m^2)
Uncontaminated soil and rocks	50.3	98 748		Reused locally				
Mixed CDW	49.7	97 613	Landfill	Waste operator 5	0.075	7321		
				Waste operator 1	0.02	1950		
				Waste operator 2	0.059	5759		
				Waste operator 3	0.027	2587		
				Waste operator 4	0.03	2928		
Total	100	196 361			0.041	4109	0	0

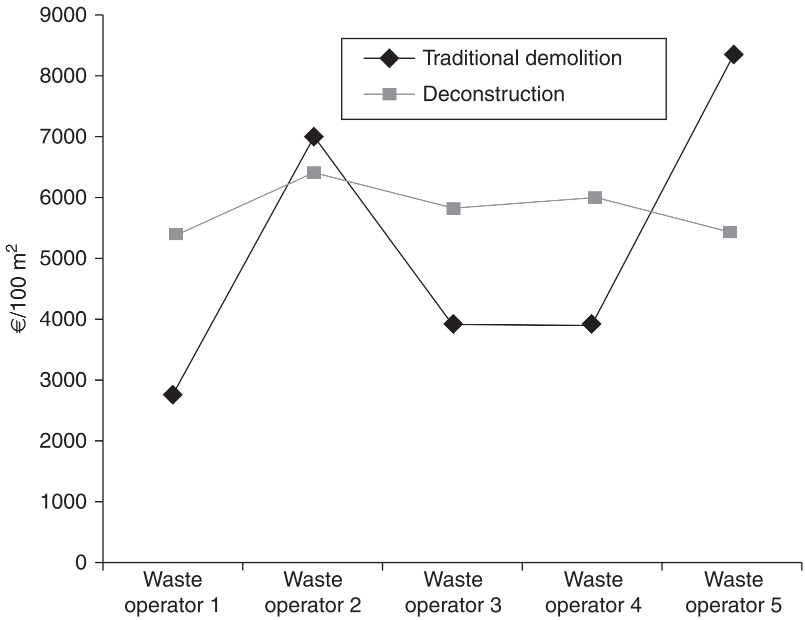
Note: Total costs are average values.

Table 7.16 Deconstruction and traditional demolition global costs (per standard housing unit m^2) for different scenarios

Cost category	Separated aggregate (deconstruction) and mixed CDW (traditional demolition) managed by waste operator									
	Waste operator 1		Waste operator 2		Waste operator 3		Waste operator 4		Waste operator 5	
	Traditional demolition	Deconstruction	Traditional demolition	Deconstruction	Traditional demolition	Deconstruction	Traditional demolition	Deconstruction	Traditional demolition	Deconstruction
Yard	155	155	155	155	155	155	155	155	155	155
Labour	202	1212	202	1212	202	1212	202	1212	202	1212
Equipment	390	1100	390	1100	390	1100	390	1100	390	1100
Transportation	156	1254	476	1674	499	1696	168	1393	234	1453
Disposal	1920	1367	5670	2024	1969	1330	2883	1842	7208	1227
Total	2682	5087	6893	6165	3215	5493	3798	5701	8189	5147

Table 7.17 Average proportion and standard deviation of cost categories over total cost

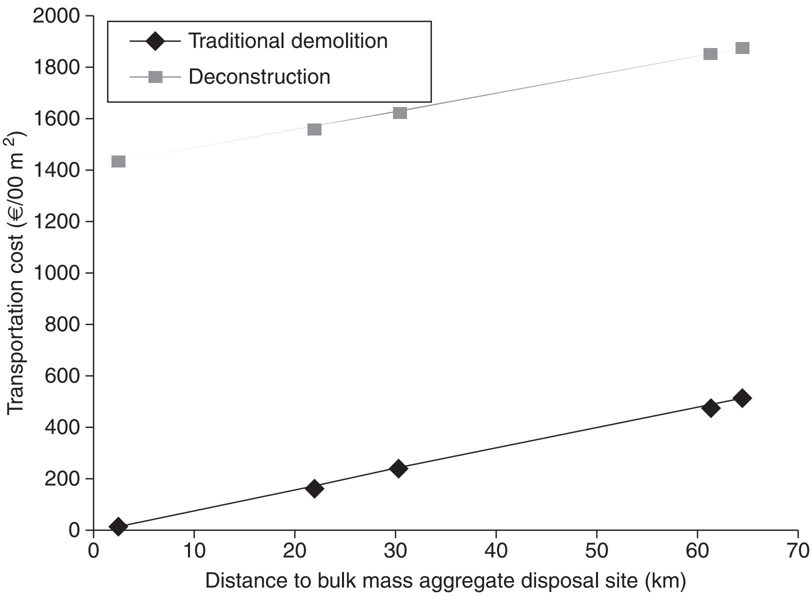
Cost category	Traditional demolition		Deconstruction	
	Average (%)	Standard deviation (%)	Average (%)	Standard deviation (%)
Yard	3.67	1.44	2.67	0.17
Labour	4.80	1.89	20.9	1.36
Equipment	10.2	4.01	19.7	1.28
Transportation	6.01	5.11	28.6	2.23
Disposal	75.3	9.28	28.1	3.60



7.6 Global costs for traditional demolition and deconstruction, per standard 100 m^2 housing unit, for different waste routing scenarios.

wider variety of destinations are expected, to which the several separated materials are routed. While separate aggregate and mixed CDW weight fractions are similar in both situations, their cost difference is significant, especially with increasing routing distance (Fig. 7.7).

It is also relevant to point out the more levelled cost structure of deconstruction than of traditional demolition, with the former presenting 21% in labour costs, 20% in equipment, 29% in transportation and 28% in disposal costs (Table 7.17).



7.7 Separated aggregate or mixed CDW transportation costs, with disposal distance.

This levelling is mainly due to both lower disposal costs and higher transportation costs. Less variability is also to be expected in deconstruction activities, when compared to traditional demolition, as shown through standard deviation values assigned to each activity (Table 7.17). The global average standard deviation for deconstruction is around 60% less than for traditional demolition.

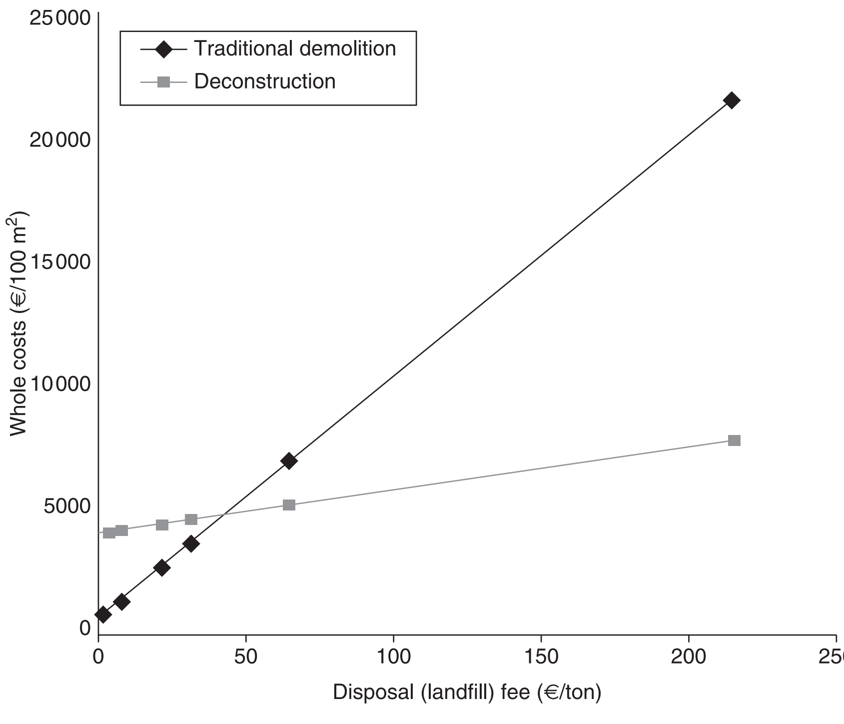
Given these results, it can be seen that marketing recovered materials has the potential to turn uneconomical deconstruction efforts into profitable ones, even at low price ranges (as compared to buying new equivalent materials). Re-using large amounts of separated aggregates in the same area or nearby locations can also save the owner or contractor considerable costs. However, marketing recovered materials implies communication and advertizing costs.

Further cost reductions in deconstruction activities are possible if execution time is reduced. However, this is usually linked with more intense heavy equipment use, as established by other studies (Guy, 2005).

7.4.6 Sensitivity analysis – disposal fees

Since disposal costs are the highest individual cost items in both traditional demolition and deconstruction, it is appropriate to consider this more closely. For this purpose, variations in disposal costs were introduced into the calculations in order to find the minimum disposal fee which results in deconstruction being globally less costly than traditional demolition. Maintaining the same calculation procedures presented above, but now considering landfill distances equal to waste operators, different disposal costs were considered (maintaining all other transport and activity costs), based on published information (Table 7.18). From this information, whole costs are recalculated for each scenario. Considering, as an example, the case of routing wastes to waste operator 1, Fig. 7.8 shows that deconstruction will be economically competitive with traditional demolition as long as disposal fees are above 38 €/ton. Repeating the procedure for all other cases, threshold disposal fees will vary between 38 €/ton and 50 €/ton, which amounts to approximately 90 and 150%, respectively, compared to common disposal fees paid by contractors in the Lisbon area (20 €/ton).

Considering the diminishing reserves of landfill space and the increasingly demanding legislation concerning waste management, this extra cost in CDW disposal is not only a necessity for increasing the profitability of deconstruction over traditional demolition, but also a natural consequence of these conditions. This expected rise in disposal costs converges towards the average landfill cost



7.8 Variation of total cost with different disposal fees (waste operator 1).

Table 7.18 Published information on landfill fees, for several world regions

Country	€/ton	Source
Portugal	20	Major Portuguese contractor direct information, 2010
Germany	213	Weisleder and Nasseri, 2006
USA (Washington)	18.7	Metropolitan Washington Council of Governments, 2006
USA (Florida)	6.6	Peng *et al.*, 1997
Kuwait	0.6	Kartam *et al.*, 2004
Canada	63.5	Chandrakanthi *et al.*, 2002
Brazil	13.2	Nunes *et al.*, 2007
Sweden	30.1	VII Congreso Nacional del Medio Ambiente (Grupo de Trabajo 14) 2004
Spain	6.4	
France	6.0	
Italy	1.0	

already observed in some EU countries (including Italy, Spain, Sweden and Germany), of around 46 €/ton.

7.5 Demolition versus deconstruction: environmental analysis

An environmental analysis was conducted, considering several possible end-of-life waste management scenarios, accounting for all building life-cycle stages and including construction materials input (therefore closing the cycle). Evaluation of environmental impacts from recycling and re-using activities is linked to the concept of avoided impacts, although all direct recycling and re-use impacts must be accounted for. If recycling and re-use transform output CDW materials into (construction materials) industry inputs, then these materials are effectively replacing virgin ones, and so the corresponding environmental impacts are avoided.

However, for the present study, and in order not to undergo a full bottom-up LCA which was considered too lengthy and detailed for this purpose, a top-down simplified approach was chosen. This choice therefore implies the use of published buildings life-cycle results as from Seppo (2004a). This allows for a considerable reduction in calculation efforts, but may only be used for comparison purposes.

7.5.1 Environmental impact assessment methodology

Buildings results analysed in Seppo (2004a) were used as base data for establishing a general life-cycle analysis building case (Table 7.19). The environmental impact factors were considered adequate to this study, since these aggregate many different substance emissions and are generally accepted as a good

Table 7.19 Seppo's results (2004a), per environmental impact factor and analysed building

Impact category	Unit	Building A	Building B	Building C	Average
Climate change	kg CO_2eq/m^2	4700	3100	3300	3700
Acidification	kg SO_2eq/m^2	15.1	8.5	9.8	11.1
Summer smog	kg C_2H_4eq/m^2	2.1	1.6	2.3	2.00
Nitrification	kg PO_4eq/m^2	1.6	1	1.3	1.30
Heavy metals	kg Pbeq/m^2	0.0021	0.001	0.001	0.0014

Notes:
All life-cycle environmental impact stages are included. From materials (extraction, production and transport) to end-of-life (demolition and waste management operations).
Considered a building life span of 50 years.
Underlying climate conditions: Finland.

representation of environmental impact. Averages were calculated for each environmental impact factor considered, using the three buildings analysed in Seppo (2004a), setting up a general reference building case (Table 7.20).

Five waste management scenarios were defined (§0), for each of the materials and end-of-life cycle stages, depending on the demolition strategy – traditional demolition or deconstruction – the materials quantity sent to landfill and different options in managing recovered materials, such as recycling and re-use and reintroduction into new constructions. All scenario environmental impacts, for the demolition/end-of-life stage, were calculated from the average values presented in Table 7.20. Since activities at this stage are essentially related to demolition efforts and transportation, these two parts were quantified separately. However,

Table 7.20 Reference building case environmental impacts, per impact factor and life-cycle stage (materials and demolition/end-of-life)

Impact factor	Unit	Life cycle stages – base case					
		Materials		Demolition/ end-of-life		Total	
		Quantity	%	Quantity	%	Quantity	%
Climate change	kg CO_2eq/m^2	271	7.3	37.0	1.0	308	8.3
Acidification	kg SO_2eq/m^2	1.26	11.3	0.37	3.3	1.63	14.7
Summer smog	kg C_2H_4eq/m^2	0.51	25.3	0.05	2.7	0.56	28.0
Nitrification	kg PO_4eq/m^2	0.13	10.0	0.06	4.7	0.19	14.7
Heavy metals	kg Pbeq/m^2	0.00057	42.0	0.00003	2.3	0.0006	44.3

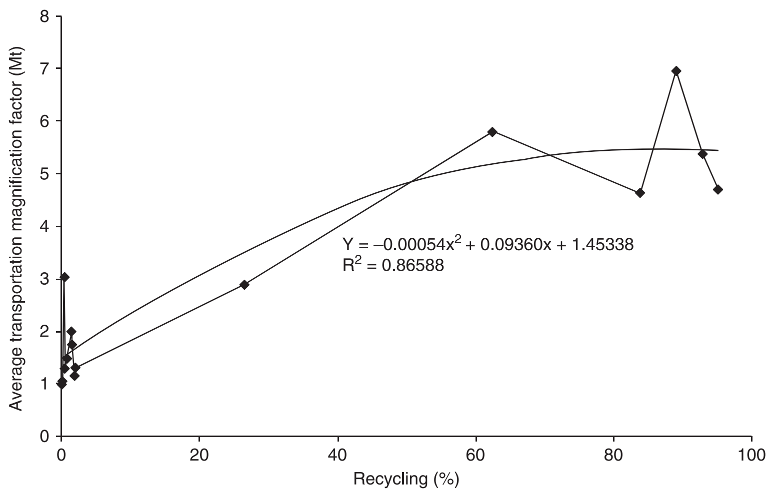
Note: Percentages refer to life-cycle stages of total life-cycle environmental impact, per impact factor.

demolition has been considered constant between scenarios, which is acceptable as demolition accounts for only about 7% of all end-of-life environmental impacts (Blengini, 2006), the rest being due to transportation. Therefore only transportation impact differences were considered, derived from different percentages of re-use and recycling within each scenario.

Transportation impact differences between scenarios were calculated considering 15 different real demolition jobs, in which recycling amounts were given by the contractor and distances estimated from known materials destinations. For each case, a total equivalent mixed CDW disposal was also calculated, in order to derive the transportation magnification factor (M_t), which is the total transportation distance measured for each case divided by the equivalent distance if all material were to be disposed of in a landfill. This equivalent distance has to be determined for each case, and depends on the mixed CDW final destination, which has been considered to comply with the following criteria:

1. The receiver/operator has to be located within 100 km from the worksite;
2. The receiver/operator has to accept several CDW fluxes;
3. After criteria 1 and 2, the chosen receiver/operator is the one to which greater distance has been travelled (in the real situation).

The results of applying this procedure to all 15 real demolition jobs are shown in Table 7.21. From this table it is clear that recycling amounts (as a percentage of total managed waste) affects transported distance, even though these are related in a non-linear way. This curve can be found in Fig. 7.9, which presents a quadratic approximation curve with a good fit ($R^2 = 0.86$).



7.9 Relationship between recycling percentage and transportation magnification factor.

Table 7.21 Transportation magnification factors and masses of recycled materials

Case/site	Generated CDW, kg	Materials destination, %		Total transportation distance, with recycling options, km			Total transportation distance, without recycling options, km	Average transportation magnification factor (M_t)	Percentage of recycled material available to be incorporated in new construction, %
		Landfill	Recycling	Landfill	Recycling	Total	Landfill		
1	50 005	7.0	93.0	153	2041	2194	408	5.4	100
2	179 200	16.1	83.9	552	5131	5683	1225	4.6	100
3	214 250	37.5	62.5	3533	4256	7789	1344	5.8	98.8
4	496 992	73.5	26.5	560	1567	2127	735	2.9	100
5	1 535 710	4.8	95.2	930	7200	8130	1729	4.7	100
6	2 138 652	98.0	2.0	2337	147	2484	1886	1.3	100
7	3 411 760	99.6	0.4	6194	720	6914	5312	1.3	100
8	4 101 440	98.6	1.4	5854	1886	7740	3861	2.0	100
9	4 135 354	99.6	0.4	8647	1148	9795	3225	3.0	100
10	4 483 375	99.2	0.8	5434	495.6	5930	3977	1.5	100
11	4 554 410	98.5	1.5	5048	2524	7572	4176	1.8	100
12	4 934 126	98.1	1.9	72 083	1766	73 849	63 455	1.2	100
13	7 213 303	10.9	89.1	33 694	31601	65 295	9390	7.0	31.6
14	18 435 606	97.3	2.7	264 144	4774	268 918	125 712	2.1	100
15	61 174 652	99.9	0.1	70 272	1001	71 273	67 146	1.1	100
Simple average		69.25	30.75	31 962	4417	36 379	19 572	3.05	95.4
Weighted average (on CDW generated kg)		92.1	7.9	2 145 929	3662	88 261	58 926	1.8	95.8
Standard deviation		35.09							

Besides demolition/deconstruction activities environmental impacts, recycling impacts must also be taken into account, with the latter depending on how much material is actually sent for recycling. Although re-use also entails environmental impacts, it has been assumed to be small compared to recycling impacts, and is therefore neglected. However, transportation in re-use activities has been taken into account. These (recycling) impacts have been quantified using published results by Blengini (2008), which refer to recycling impacts of managing mixed ceramic and concrete aggregates and steel (ferrous metals in general). From these results, average extra impacts from recycling were determined, which amount to about 17% for aggregates and 39% for steel. From these averages it is possible to estimate, knowing the quantities of recycled aggregates and ferrous metals (for each scenario), the overall extra environmental impacts due to recycling (Table 7.22).

Finally, environmental credit for recycling is also being considered, since recycled materials eventually replace virgin materials in the industrial cycle. In the present calculation procedure, this is achieved by applying a reduction factor (for all impact factors) in the materials part of the life cycle, proportional to the recycled/reused mass.

7.5.2 Analyses scenarios

Analyses scenarios are based on previous analyses of existing buildings (Coelho and de Brito, 2010):

- **Scenario 1**: Traditional demolition of all materials, routed to landfill (comparison case). Environmental impacts are considered equal to the general reference building case. Transportation impacts are calculated using a unitary M_t.
- **Scenario 2**: Soft stripping of non-structural elements, followed by traditional demolition of structural elements. Soft stripping (for recycling) includes recoverable coverings, sanitary equipment and piping, electric cables, doors and windows. All other materials are traditionally demolished, such as masonry walls, concrete, steel and wood, and are sent for landfill. Demolition

Table 7.22 Extra environmental impacts due to recycling, for the selected scenarios

Scenario	Aggregates sent to recycling/total CDW, %	Ferrous metals sent to recycling/total CDW, %	Extra environmental impact weighted average, %
2	0.66	0.10	0.15
3	4.8	2.05	1.62
4	93.7	0.64	16.3
5	53.9	0.63	9.48

impacts are considered equal to those of the general reference building case. Transportation impacts are calculated using the M_t corresponding to the scenario recycling percentage. Extra recycling environmental impacts are calculated from those of Scenario 1, using the corresponding figure in Table 7.22 (same procedure for Scenarios 3, 4 and 5).

- **Scenario 3**: Deconstruction (for re-use) of non-structural elements, followed by traditional demolition of structural elements. Even though structural elements are taken down in a traditional way, these are sent to recycling whenever possible. Demolition impacts are considered equal to those listed for the general reference building case. Whenever concrete aggregates are sent for recycling, its use in new construction is limited to 10% of recovered mass (accounts for general fills and pavements).
- **Scenario 4**: Complete deconstruction, sending all materials for recycling. Only hazardous materials are sent to a controlled landfill. Separately recovered concrete and ceramic aggregates are sent for recycling (100%). All recovered materials are potentially recyclable, and therefore able to be reintroduced into new construction elements.
- **Scenario 5**: Complete deconstruction, preparing for re-use whenever possible. All other materials are sent for recycling. Only hazardous materials are sent to a controlled landfill (including gypsum based materials). Extra re-using activities environmental impacts are ignored. Re-used materials transportation impacts are taken into account, employing the same transportation distance as if these were routed to landfill. From all materials targeted for re-use, only a part is actually re-usable (Guy (2000, 2005); Southworth, 2009), amounting to about 50%.

7.5.3 Environmental impact calculation

As far as environmental impacts are concerned, the described scenarios are only different in the demolition/end-of-life stage due to differences in transportation distances, and in the materials stage due to recycling and re-using in new construction. In the latter stage, environmental impacts can be calculated using the following equation, for Scenario 2:

$$I_i^{C2} = I_i^{C1}\left(1 - \frac{P_r^{C2}\, u^{C2} / 100}{100}\right)\left(1 + \frac{A^{C2}}{100}\right) \qquad [7.1]$$

where:

- I_i^{C2} is the environmental impact value, for category i, in Scenario 2;
- I_i^{C1} is the environmental impact value, for category i, in Scenario 1;
- P_r^{C2} accounts for the material percentage sent for recycling, in Scenario 2;

- u^{C2} equals the weighted average, over waste mass, of possible recycled material used in new construction; and
- A^{C2} is the recycling operations averaged extra environmental impact, expressed as a percentage.

This equation applies to Scenario 3 by using a different quantity of materials sent to recycling, which affects the M_t factor, and therefore the environmental impacts due to transportation. In Scenario 4, as all materials are recycled and reintroduced in new construction products, the factor u^c is 100%, which simplifies equation [7.1]. Finally, for Scenario 5, the equation used for Scenario 4 is still valid, but the factor P_r is divided into recycled and re-used percentages of managed material mass.

7.5.4 Comparison between scenarios

Materials and end-of-life environmental impacts for the scenarios considered are presented in Table 7.23. Table 7.24 summarizes measured variations between the scenarios, taking Scenario 1 as the benchmark. If all environmental life-cycle stages are taken into account however, the differences are not so significant (Table 7.25).

It is clear from Table 7.23 that the materials stage accounts for more than 75% of any environmental impact factors for Scenarios 1 through 3. In these cases, the replacement of virgin materials by recycled ones is low (<9%), and so impacts from producing and transporting virgin materials remains predominant. For Scenarios 4 and 5, this life-cycle stage always remains under 35% for any environmental factor. Due to the high levels of recycling and re-use in these scenarios (>95%), major impact reductions are possible in the materials stage, so that this stage's environmental impacts are lower than those from the end-of-life stage.

Table 7.23 also shows that the highest reductions occur in the heavy metals (−88%), summer smog (−81%) and climate change (−77%) categories, when comparing Scenarios 1 and 5. Scenarios 2 and 3, with their modest percentage of material mass sent for recycling (and assumed re-insertion in the construction material's industry), can actually generate more environmental impacts than the traditional demolition/landfill situation (Scenario 1), which comes from the extra transportation distances necessary.

Scenarios 4 and 5, as expected, present considerable environmental reductions as compared to Scenario 1, which essentially derives from recycling and re-use mass percentages of over 95% and in ensuring this mass re-enters the construction production cycle. The fact that Scenario 5, in which some re-use of materials occurs, does not imply a significant environmental impact reduction compared to Scenario 4, is the result of the modest amount of re-usable mass (~20%), as well as the small increment in transportation distance (due to re-use activities). The

Table 7.23 All scenarios environmental impacts, for materials and end-of-life life-cycle stages

Impact factor	Units	Scenarios – life cycle stages									
		Scenario 1		Scenario 2		Scenario 3		Scenario 4		Scenario 5	
		Materials	End-of-life	Materials	End-of-life	Materials	End-of-life	Materials	End-of-life	Materials	End-of-life
Climate change	kg CO_2eq/m^2	271	12	267	18.5	253	24.3	11.6	54.4	10.9	54.5
Acidification	kg SO_2eq/m^2	1.26	0.12	1.24	0.19	1.17	0.24	0.05	0.55	0.05	0.55
Summer smog	kg C_2H_4eq/m^2	0.51	0.017	0.50	0.027	0.47	0.035	0.022	0.078	0.020	0.078
Nitrification	kg PO_4eq/m^2	0.13	0.020	0.13	0.030	0.12	0.040	0.006	0.089	0.005	0.089
Heavy metals	kg Pbeq/m^2	5.74E-04	1.05E-05	5.64E-04	1.60E-05	5.34E-04	2.09E-05	2.45E-05	4.69E-05	2.30E-05	4.69E-05

Table 7.24 Materials and end-of-life global environmental impact differences (%) between scenarios 2 and 5, compared to scenario 1

Impact factor	Scenario			
	2	3	4	5
Climate change	0.57	−2.34	−76.7	−76.9
Acidification	3.02	2.49	−56.7	−56.9
Summer smog	0.06	−3.35	−80.9	−81.1
Nitrification	5.45	7.28	−36.8	−37.0
Heavy metals	−0.78	−5.01	−87.8	−88.0

Table 7.25 All life cycle stages environmental impact differences (%) between scenarios 2 and 5, compared to scenario 1

Impact factor	Scenario			
	2	3	4	5
Climate change	0.045	−0.19	−6.08	−6.10
Acidification	0.39	0.32	−7.30	−7.33
Summer smog	0.016	−0.19	−22.0	−22.0
Nitrification	0.65	0.86	−4.36	−4.38
Heavy metals	−0.34	−2.20	−38.5	−38.6

modest re-used mass in Scenario 5, free from direct environmental impacts, is not enough to allow significant environmental impact reductions.

Even though in global environmental terms (at all stages of the life cycle) reductions in impacts are less significant (Table 7.25), it is still important to guarantee high levels of recycling/re-use and replacing of virgin materials in construction (Scenarios 4 and 5). Nevertheless, reductions in environmental impacts are clear, especially in smog and heavy metals impact factors, at 22 and 38.5% respectively, when comparing Scenarios 4 and 5 with Scenario 1 (benchmark). Climate change and acidification impact factors are potentially reduced, within the complete building life-cycle perspective, at 6.1 and 7.3% respectively, which is considered significant.

7.6 Conclusions

A bottom-up economic analysis of a deconstruction case study has been undertaken and compared with a demolition situation. A top-down environmental analysis was also conducted, comparing several deconstruction and demolition scenarios. From an economic point of view, the following conclusions are possible:

- Traditional demolition is still economically advantageous over deconstruction, within present regional economic constraints, although in some circumstances, namely when disposal costs are considerable (above 30 €/ton), deconstruction can be competitive.
- Cost structure is more distributed between labour, equipment, transport and final disposal costs for deconstruction, whilst traditional demolition is dependent on disposal costs.
- Deconstruction is more time and labour intensive than traditional demolition, which may require as much as 6 times more labour and take 6.5 times longer to execute.
- Landfill costs should be raised by up to 150%, for deconstruction to gain competitiveness when compared to traditional demolition.
- Other options for enhancing the profitability of deconstruction are linked with the possibility of marketing recovered materials, even at low prices, and introducing further mechanization efforts in conducing deconstructions, which also benefit from more accurate planning and optimized procedures.

Environmentally, the top-down analysis of different building demolition scenarios has made possible the following conclusions:

- Only removing surface non-structural materials from buildings will not imply significant environmental impact reductions when compared to the traditional demolition scenario. Extra transportation needs will offset avoided impacts, which result from applying recycled content materials in new construction products, although this effect is slight (generally <5%).
- Routing materials for recycling and re-use in large quantities, >95% of demolished mass, does bring environmental benefits, which can reach 88% (in the heavy metals impact factor) at the materials and end-of-life cycle stages, and 39% across the whole environmental life cycle (for the same impact factor). For other impact factors, reductions are smaller but still considerable, even when accounting for complete life-cycle impacts.
- What really embodies environmental impact reductions is replacement of new construction materials, whether these substitute materials are from recycling or re-use activities. Since the construction industry in Portugal is not prepared to undergo considerable change in its re-use of materials (>25%), the focus should be on recovering as much mass as possible from building sites for recycling purposes. This remains true, even when considering that re-using entails less direct impact than recycling, confirmed by the small differences between environmental impact reductions of Scenarios 4 and 5 (compared with Scenario 1).

7.7 Future trends

This study yields a first picture of how deconstruction activities might impact economic and environmental performance, when compared to more traditional approaches to building removal represented by simple demolition and landfill of resulting materials. Regional studies would broaden the scope of the conclusions, and would better inform regional waste management policy and relevant economic and environmental impacts. Social impact factors could also be considered to include the importance of social development in the context of waste management activities.

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8
Demolition techniques and production of construction and demolition waste (CDW) for recycling

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Abstract: Demolition techniques and the management of construction and demolition waste (CDW) of buildings are key issues in the development of sustainable construction. Prevention, re-use and recycling are the basic approaches to waste management. Demolition techniques, which have hitherto been unsafe, dangerous and uncontrolled, are today developing into an engineering discipline. Thus demolition of buildings is now a well-planned part of civil engineering, with many aspects to ensure a safe demolition phase.

Key words: demolition, end-of-lifetime, recycling, life-cycle engineering.

8.1 Introduction

Buildings now reaching their end-of-life were not constructed in the past with thoughts of later re-use of materials or structures. Even today, demolition as one possible strategy at the end-of-life of a building is not deeply investigated. While buildings from the latter half of the last century reach their end-of-life, and rebuilding at the same places becomes ever more important, demolition techniques also become increasingly important. The boundary conditions are a dense building development and high traffic rates, so that any uncontrolled methods for demolition are no longer appropriate (Kamrath and Hechler, 2011).

Another issue concerns waste management. Especially in highly populated areas of Europe, the amount of recycled concrete or masonry generated by demolition projects is about twice as much as the need for recycled material, for example as an alternative to natural gravel. In general, the need for crushed concrete or masonry depends on the availability of natural resources. Thus, if natural gravel exists locally only in small quantities, willingness to use recycled products is higher and recycling becomes more important as an alternative resource.

Modern demolition projects have to respect two important rules:

1. Planners must consider waste management, as well as contamination and recycling possibilities.

2. The demolition must be safe and undertaken within technical guidelines, which reduce risks to an absolute minimum (Kamrath, 2012).

The main aspect of this chapter is a general introduction to the demolition techniques applicable today.

8.2 End-of-life scenarios for buildings

Any building has a limited lifetime. Although there are a number of buildings which last for more than 100 years, more usual lifetimes are given below. At the end of life, there exist typically three possibilities, the most appropriate option depending on costs, environmental conditions and other local issues such as preservation orders (Dorsthorst and Kowalczyk, 2002):

- **Deconstruction**: A clearance could extend the lifetime and could be an alternative to demolition. During a building clearance, any non-load bearing parts of the building will be deconstructed. The rebuild process starts with the old skeleton.
- **Reuse of structure**: Deconstruction and re-use of the structure itself could be an alternative for some structures, especially those made of steel. This method helps to generate a second life for the load bearing structure, e.g. for bridges or halls at another place.
- **Demolition**: Complete demolition is the typical end-of-life scenario. To avoid waste and landfilling, re-use and recycling of materials should be taken into account.

The lowest impact on natural resources is achieved by re-use of whole structures or even a whole building. If a whole building can be deconstructed and rebuilt elsewhere, no waste is produced. If this possibility is considered prior to build, this is called 'design for deconstruction' (Hechler *et al.*, 2012).

If whole structural elements cannot be re-used, recycling of materials is the best choice with still little waste. Re-using and recycling is divided into three stages (BIS, 2011):

1. **First-order** recycling is possible with all kinds of metals and glass. Steel is the world's most recycled material. After melting the sorted materials, a new product of the same quality can be made.
2. **Second-order recycling** can be done with all non-polluted mineral materials such as concrete, brick-stone and general masonry. Recycling of concrete is the process of producing gravel, which can be used instead of natural gravel (Fig. 8.1). However, concrete, unlike a first-order material, while made from cement cannot be recycled into more cement. Once concrete has been made, there is no practical way to decompose it into the basic elements of sand, water, aggregate and cement that went into its formation. Nevertheless, processing waste concrete to produce recycled concrete aggregate (RCA) or

8.1 States of concrete during recycling progress: Concrete as given after deconstruction by machines (left), separated core wires (middle) and gravel of concrete (right).

recycled crushed concrete (RCC) has the potential to greatly reduce the quantity sent to landfill each year, and complements the government's sustainable development and waste minimization policies (Kirby and Gaimster, 2008). The potential of RCC/RCA differs according to the local market. If natural gravel is cheap, the demand for recycled concrete is low. If no natural resources exist (e.g. the Netherlands) the potential is high and the possibilities for recycling are also high (Blengini, 2009).

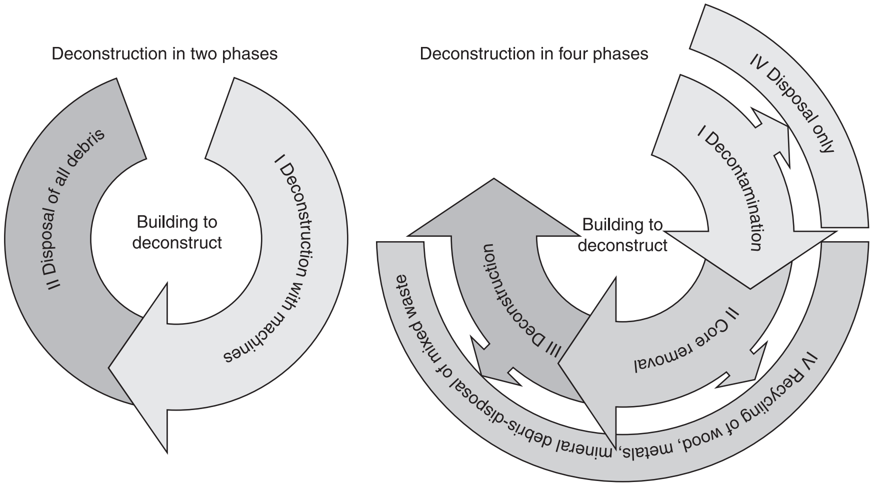
3. **Third-order recycling** concerns thermal use. Wooden materials and plastics are possible sources of energy for power plants. If no thermal use is possible (e.g. due to pollution or contamination), landfilling is the only possibility.

8.3 Planning demolition

The demolition of buildings began to be considered by the construction industry in the second half of the 20th century, when buildings and industrial sites dating from the industrial revolution reached their end of life. At that time, the required machinery, such as excavators and trailer trucks, became available to medium- and small-sized demolition companies. The scope of duty was mainly the complete demolition of a building. Up to the 1970s, demolition rubble was deposited on dumpsites. Thus, the waste was not re-used or recycled. Even contaminations were not strictly handled as hazardous material. Demolition was not so much a matter of knowledge but of logistics only.

Today demolition is a much more sophisticated task. In general, a typical demolition differentiates four main tasks (Fig. 8.2):

1. decontamination of the building;
2. core removal or deconstruction of non-load bearing constructions;
3. demolition by machines;
4. disposal and/or recycling.



8.2 Phases of deconstruction: Simple deconstruction model in two phases without any recycling or contamination management (left) and modern concept with four phases (right). The modern concept sorts waste into recycling fractions, contaminated waste and disposal of mixed waste.

While earlier demolition concepts do not consider any possibilities of re-use or recycling of the debris, the modern concept with its four phases separates the debris at any stage of the demolition (Kamrath and Hechler, 2011b). Nevertheless, the recycling-rate depends on the materials used in construction (Röbenack *et al.*, 2007).

During the decontamination phase, all pollutants will be removed from the building. The typical pollutants are cement, asbestos, tar-bitumen roof sheeting and polychlorinated biphenyles fillings in joints. Such materials cannot be re-used. Man-made fibres and wood painted with old preserver colours also need to be separated during the decontamination phase. In general, all pollutants have to be deposited on special dumpsites. Because of the high thermal energy rate of roof sheeting and wood, waste incineration as the lowest level of re-use is preferred to deposition.

High recycling rates are possible only if separation starts before the demolition phase with machines, for example excavators (Weiß, 2002). In particular, wooden materials and plastics tend to split into smaller pieces if not separated. In this way, poor quality of the recycled gravel due to extraneous material should be avoided. In the core removal or deconstruction phase, all non-mineral dry constructions will be removed and separated (Fig. 8.3). Typical materials are wood (doors, frames, etc.), metals (water pipes, electrical lines) and mixed waste (carpets, plastics, etc.).

(a)

8.3 Decontamination phase: Removal of cement asbestos (a) and core removal (b). At the end of phase II (core removal), the building is prepared for deconstruction with machines.

Demolition by machines is the phase during which the structure of the building (i.e. concrete, steel constructions, walls) are torn down by excavators, cranes and other equipment. In this phase, depending on the structure, mostly mineral debris or steel arises. While recycling or re-use of a steel construction is generally preferred, the recycling rate of concrete and stonework depends on the quality of the material. Pumice stone cannot be recycled because the strength of the material

(b) 

8.3 Continued

does not suite the requirements for gravel. Concrete needs to be crushed into separate core wires, as it could otherwise damage recycling machines (Fig. 8.3).

8.3.1 Calculating the mass of demolition waste

The exact masses of demolition materials are mostly unknown. One reason for the lack of exact information is the absence of exact drawings of the building. It is thus necessary to estimate masses. Data from preliminary demolition projects may be used for extrapolation purposes. The goal is to calculate the masses of concrete, bricks, wood and waste using the volume of the building. Masses can be calculated as (BMI, 1998; Kamrath *et al.*, 2011a):

$$m_i = V_{building} \cdot f_i \qquad [8.1]$$

In this formula, m_i describes the calculated mass of material i, $V_{building}$ the overall volume of the building in m^3 and f_i is a correction factor. The estimation of the correction factor is difficult since two buildings never are the same, but a rough categorization is possible. f_i depends on the size of the building – the number of floors for example. Older buildings have the ceilings made out of wood, newer buildings tend to have the floors made of concrete. Heat insulation (walls, roof) produces waste that also has to be taken into account. The former use of the building also yields information about the masses. Older houses, with a history of long-term lodgers, tend to have several overlapping carpets and/or PVC floors.

but reduced amounts of hard plaster walls. Gypsum in general has to be removed before demolition, and buildings with a history of private ownership tend to have undergone several renovation phases and thus a lot of gypsum plaster has been used.

8.3.2 Mass–volume correction factor f_i for general buildings

Values for the mass–volume correction factor f_i have been measured in the interval 0.07 up to 0.250. Lower values have been indicated for halls, which have no walls inside. Higher values are possible for concrete structures of industrial sites. In general, f_i becomes higher for concrete structures as opposed to brick-wall structures, and lower the larger the building is. The amount of wood is about three to four times higher than the amount of general waste (BMI, 1998; Kamrath *et al.*, 2011a).

8.4 Demolition technologies

The demolition industry has undergone a major transformation within the last 20 years (Hurley and Hobbs, 2003). Traditionally, it has been a labour intensive, low skill, low technology and poorly regulated activity, dealing mainly with the disassembly and demolition of simply constructed buildings (e.g. from masonry). Common methods have included demolition by hand (with portable tools) and pulling (e.g. with ropes). More recently, mechanized processes with specialized equipment replacing manual labour have been invented. This trend results from the increased complexity in building design, the financial pressures from clients, health and safety issues, regulatory and legal requirements and advances in plant design. Traditionally, much of the demolition contractors' income was from the sale of salvaged and recycled materials. Today, income is mostly generated from the contract fee – demolishing as quickly and as safely as possible.

There are a lot of different techniques to demolish a building completely. All techniques require a completed core removal/deconstruction phase, as well as the prior removal of contaminations (Coelho and de Brito, 2011). Demolition is possible by hand, machines (excavators with different tools attached), by use of explosives, and through a combination of concrete saws and cranes. Some methods are faster than others, with some producing fewer emissions or with a higher degree of control. Different methods are useful for different scenarios (Röbenack *et al.*, 2007).

Today's demolition process relies on one of eight basic methods: pulling, impact, percussion, abrasion, heating (or freezing), expanding, exploding or bending (Röbenack *et al.*, 2007). Most demolition work is done by excavators (82%, Table 8.1). Excavators can have several different attachments and are the most flexible instruments for demolition. For safety reasons, the use of wrecking balls (3% today in Germany) is limited, though for some cases, the use of cable

Table 8.1 Main techniques of building demolition and deconstruction in Germany

Main techniques (usage quota in Germany, 2002)	
Deconstruction by excavators with shears etc.	82%
Exploding	4%
Deconstruction with wrecking balls	3%
Other machines	3%
Percussion, abrasion, heating etc.	3%
Robots	0.3%
Other	4.7%

Source: Röbenack *et al.*, 2007.

excavators with wrecking balls is still the best method. If the thickness of the walls is high, and the height of the building is large, construction excavators with attached shears may not be able to perform an efficient demolition (Cohrs *et al.*, 2002).

Especially in Europe, the use of explosives is low (3%). The use of explosives instead of excavators is useful for large buildings only, and only if the surrounding building density is low. This condition is not met for most European areas. Even after blasting a building, excavators are needed afterwards for controlled crushing of the remaining parts of the building. Thus, exploding is a special technique to lower the height (Bouza *et al.*, 2002).

While demolition by excavators, wrecking balls and explosives are the main techniques to destroy whole buildings, other techniques are used for partial demolition. Abrasion is the removal of parts of concrete from floors or slabs, percussion drilling and saws are used to cut parts of concrete out of a structure, and robots may be used for dangerous areas. The next section describes the different tools in detail. A rough overview is given by Table 8.2.

8.4.1 Wrecking balls

The primary problem with the use of wrecking balls is the difficulty in controlling them. While they can be particularly effective in destroying masonry and concrete buildings, they are less precise than blasting, shearing or implosion techniques. Given the size of an average wrecking ball, a slight variation in aim can have enormous consequences. The arc of the wrecking ball must be controlled very carefully. Because safe wrecking ball operation relies on careful control, skilled labour is necessary for effective crane operations. For this reason alone, many demolition crews prefer to use less risky alternatives, thereby permitting them to employ crews of less skilled labour. With the decreasing use of wrecking balls, fewer crane operators have the necessary experience in wrecking ball demolition. When the crane operators are called on for the task, relative

Table 8.2 Pro and contra of the different demolition techniques

Wrecking ball	Shears	Push/pull	Grabbing	Hydraulic breaker
↗ Very effective	↗ Effective	↗ Only applicable with masonry	↗ Lightweight constructions	↗ High reinforced concrete/thick constructions
↘ Low control	↗ Precise, controlled deconstruction	↘ Limited height	↘ Limited height	↘ No separation
↘ No separation	↗ Separation	↘ Applicable for part of buildings (walls)	↗ Separation	↗ Precise
↘ High noise emission	↗ Little noise emission	↗ Little noise emission	↗ Little noise emission	↘ High noise emission
Limitation: no neighbouring buildings	*Preferred for controlled deconstruction*	*Extender tools expand the height*	*Roof structures, mainly of wood and light steel*	*Deconstruction of buildings*

inexperience can introduce further risks into an already dangerous technique (Cohrs *et al.*, 2002).

Misuse of the wrecking ball may have disastrous consequences. The wrecking ball may snap free from the crane, potentially destroying buildings or even causing deaths. If improperly guided, a wrecking ball can also overload the crane. Should this happen, the path of the wrecking ball becomes impossible to control. This may cause backswing, with the wrecking ball hitting the crane boom. An accident of this magnitude demonstrates the worst possible consequences of wrecking ball demolition.

Besides the danger involved, many crews will choose against wrecking balls for reasons of efficiency and convenience. While a wrecking ball can conveniently destroy masonry or concrete, once it has broken the concrete into smaller pieces, a good deal of additional labour is still required to cut through intact steel rebar. Furthermore, a wrecking ball can only be effective for those buildings with a size corresponding to that of the crane used. In addition, such factors as nearby power lines must be taken into consideration, and may render a site inappropriate for wrecking ball demolition. Finally, the wrecking ball poses inconvenience to the neighbourhood surrounding the demolition site, creating substantial noise, dust and vibration (Farfel *et al.*, 2003).

8.4.2 Demolition shears

Most demolition today is done piecewise by cracking shears. This demolition is done in the opposite way to the building process. Columns and other load bearing parts are demolished last, while slab constructions are demolished first (Fig. 8.4). With the help of cracking shears, buildings are demolished from the top down. Neglecting this rule causes stability faults and can result in uncontrolled collapse. Demolition shears are work tools with moveable jaws equipped with blades of hardened steel on both an upper and a lower jaw. Technically, the cracking shear destroys the concrete or masonry structure only between the two shear blades. The advantage of cracking shears is the high level of control (Anumba, 2003). Unlike wrecking balls, there are no kinematic forces involved in the demolition process,

8.4 Cracking shears only affect small surrounding areas. This causes a high level of control.

and therefore no moving parts which could get out of control. The demolition energy is supported by hydraulic pumps. The machine operator needs a basic knowledge of construction to use cracking shears effectively (Mikrut *et al.*, 2009).

The use of cracking shears is limited by the height of the excavator, which is usually below the height of cranes. Regular excavators are not higher than 12 to 15 m and therefore higher buildings need special excavators if demolition is to be achieved by scrapping shears. Modern long-front excavators can reach heights up to 40 m, though with extension of the height, the loading weight available for the attachment tools becomes lower. As a consequence, the shears used for greater heights are smaller and less powerful than regular shears, reducing efficiency. Demolition using cracking shears consumes little space and produces little vibration. It is thus the preferred technique in highly populated areas.

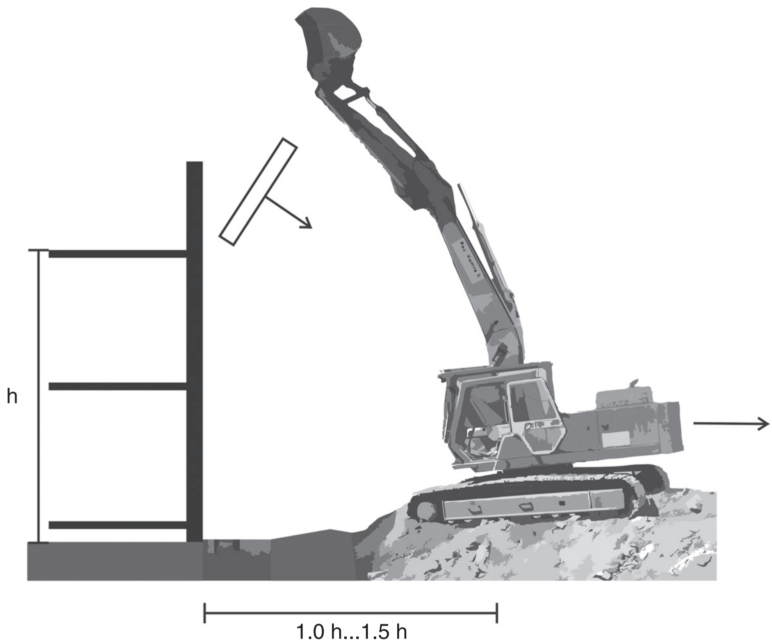
8.4.3 Pushing and pulling walls

While wrecking balls as well as demolition shears are common for the demolition of a whole building, techniques of pushing and pulling are additional measures for specific parts of a building. Before tools such as the demolition shears were invented, and at a time where buildings were mainly made of masonry, pushing and pulling of walls was the only means of demolishing a building with excavators. The idea is quite simple. The resistance of a brick wall against the power of an excavator is low, so it is possible to push a wall inside the building with the help of the backacter or to put the backacter on top of a wall and to pull the wall outwards (Fig. 8.5). Since the height is the limiting factor, pulling extenders could be used to extend the reach of the excavator (Fig. 8.6).

It is obvious that there is a risk, especially in pulling walls. Walls usually break at the bottom of the next floor, but it is not possible to forecast the exact cut. The demolition crew has to carefully determine how the wall will act. For this reason, before pulling a wall outwards, the excavator driver should test the behaviour of the wall by some small-powered test pulls to examine the exact location of the cut. Because of the danger, the safety guidelines (BG-Bau, 2010) for pushing/pulling building parts recommend a safety clearance of between h and $1.5 \times h$ from the building, where h is the remaining building's height.

8.4.4 Grabbing

Grabbing parts of a building is only suitable if the resistance is low, such as with lightweight structures, and is not suitable for structures made of concrete or masonry. Wooden structures can be dismantled quite well by the use of a grabbing device attached to an excavator (Fig. 8.7). The main advantage of grabbing tools is the possibility of avoiding mixtures of mineral waste with lightweight materials such as wood. Wooden floors or ceilings are removed as a whole instead of sorting manually after the demolition of the building. A grabbing tool can be helpful for



8.5 Safety use of pushing/pulling techniques. The excavator pulls on a wall. For safety reasons the minimum distance should not be less than the revealing height of the building.

8.6 To extend the reach of the excavator, special pushing/pulling devices can be used.

8.7 Grabbing lightweight parts of, for example, the roof structure.

some masonry parts as well, if the resistance of the masonry against demolition is low and spatial constraints are high. The use of grabbing devices for masonry walls is limited however, and this is not their main application as abrasion issues reduce the lifetime of the tools.

8.4.5 Hydraulic breaker

Most parts of buildings are not thicker than 1.0 m and may thus be demolished by the use of, for example, shears. If the thickness is higher, the use of wrecking balls could be considered. Because of the high risk with wrecking balls however, the best applicable tools for thick elements are hydraulic breakers (Fig. 8.8). While shears are used mostly for demolishing parts of a building above the ground, breakers are commonly used for demolition under the ground. Their main disadvantage is the high level of noise. To get the maximum power out of the hydraulic breaker, it is necessary to weight the breaker against the mass of the excavator. Working above the head is therefore not very effective (Röbenack *et al.*, 2007).

8.4.6 Summary

The demolition of buildings is achieved with excavators in most cases. Attached to the excavator are different tools. In general, there is not one single tool for demolition of a building, but different tools are required for different parts of a building. Thus, the roof could be grabbed, the walls pulled with pulling reach

8.8 Use of hydraulic breakers to destroy fundaments of a building.

extenders, floors crushed by shears, and the foundations demolished by hydraulic breakers. Flexibility is the greatest advantage of demolition performed by excavators, which is thus the preferred method, unless special circumstances apply. The use of wrecking balls is risky on the one hand. On the other hand, the flexibility of modern excavators is higher, so that other techniques than excavators with attached tools are used only if the special circumstances do not meet the conditions to use an excavator.

8.5 Top-down and other demolition methods

In this section the general techniques for demolition are described. Demolition by machines or manual labour is always done from the top down. However, demolition by explosion is a bottom-up method. Top-down methods imply that no load bearing parts of the building should be demolished where there still exist parts of the building that could collapse. We focus here on the demolition of steel and old-style slab floors, which are made out of steel beams with concrete between each two beams. For demolition by machine, the main focus will be on the practice of the 'top-down' method (BDHK, 2004).

8.5.1 Top-down method

Demolition from the top down means that one proceeds from the roof to the ground in a progressive manner (BDHK, 2004). Particular sequences of demolition may vary, depending on site conditions and the structural elements to be demolished. The demolition sequence is determined according to the building layout and construction, as well as the given site conditions. Generally, the following sequence can be applied:

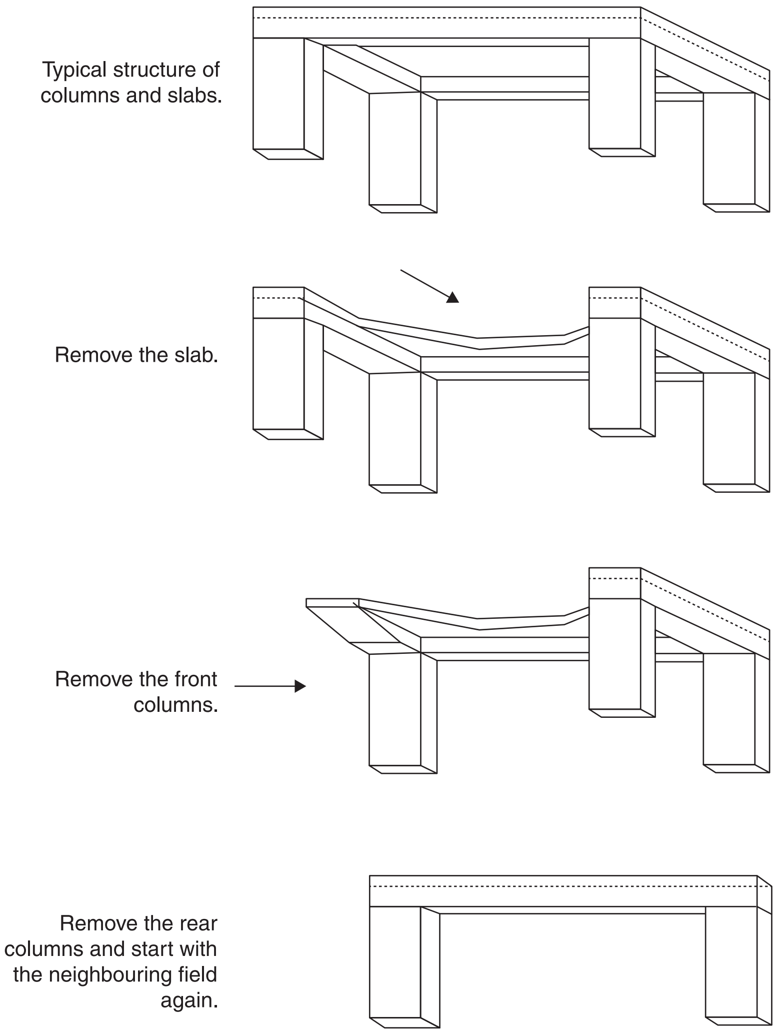
- To ensure safety at later stages, all overhanging structures such as verandahs, balconies, emergency stairs, etc. should be demolished prior to the main demolition. Roof installations (lifts, air conditioning units, etc.) should be removed, to avoid them falling down during the demolition process.
- Demolition of floor slabs begins at mid-span, working towards the supporting beams.
- Floor beams are demolished in the following order:

  (i) cantilevered beams;
  (ii) secondary beams; and
  (iii) main beams.

- Non-load bearing beams are first removed. Subsequently, load-bearing beams are removed from the top down.
- As soon as possible, the ground floor should be destroyed to avoid demolition waste lying on it. Due to the enormous load, this floor could otherwise collapse.

Figure 8.9 shows the demolition of a building using the general top-down sequence. Figure 8.10 also illustrates the technique.

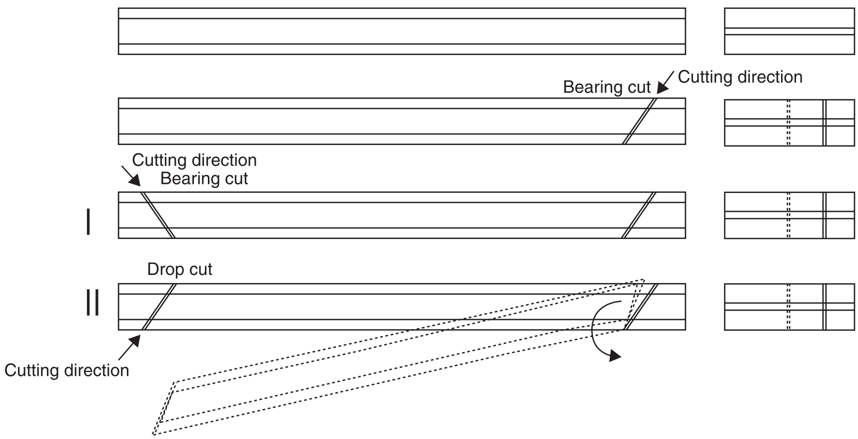
8.9 Typical demolition from top to down: After demolition of all non-bearing structures, the bearing columns are left. In the next step, the columns will be demolished beginning with the top level. After that, the demolition starts again with the next field.



8.10 How demolition with shears should be done. Modern buildings are typically constructions of columns of concrete with slab layers. First, the slab should be removed by cracking shears and then the columns from front to rear.

8.5.2 Manual demolition of steel beams

Demolition shears well suited to the demolition of concrete. Shears suitable for the cutting of steel beams also exist (Hall, 1993). Unfortunately, for higher buildings, and for thicker bearing steel construction, the possibility for cutting is reduced. Thus, beams still have to be flame-cut in many cases. Figure 8.11



8.11 Manually flame-cutting of horizontal beams. Always start with a bearing cut. Due to the direction and angle of the cut, the stability of the structure is not affected. If the beams can be grabbed by an excavator, the second cutting point is developed as a bearing cut as well (I). If the beam has to be removed by gravity, the other cutting point is developed as drop cut (II). It is important to perform the second cut from bottom to top of the beam. After the cut is done, the beam will fall down due to gravity.

explains two safe methods for flame-cutting of steel beams. Both methods consist of two possible types of cut. The bearing cut is the physical cutting of a steel beam, whilst the beam stays in place. The cut is therefore done at an angle of about 45 degrees from the top to the bottom of the beam, parallel to the beams longest dimension. The drop cut is performed in the other direction, from the bottom to the top, so that the beam will fall under gravity. It is recommended to perform bearing cuts on both sides of the beam and to grab the cut beam with the help of an excavator or crane (Fig. 8.12). If flame cutting has to be performed without the help of an excavator, first a bearing cut should be applied, and then a drop cut.

8.5.3 Demolition of building edges

Whenever possible, demolition should start on the rear side of a building, proceeding towards the front, and from the top down. The demolition will reach a point therefore, where the inner part of the building is already demolished and the front still exists. The typical technique to demolish the front is to pull the wall from the back inside the building. Most buildings made of masonry enhance stability at corners with the help of an overlapping technique, so that alternately one brick from the front and the side wall are embedded into the corner. For demolition, this technique causes danger as the corner part of the side walls could

8.12 The beam was flame-cut manually with two bearing cuts and then safely removed by crane.

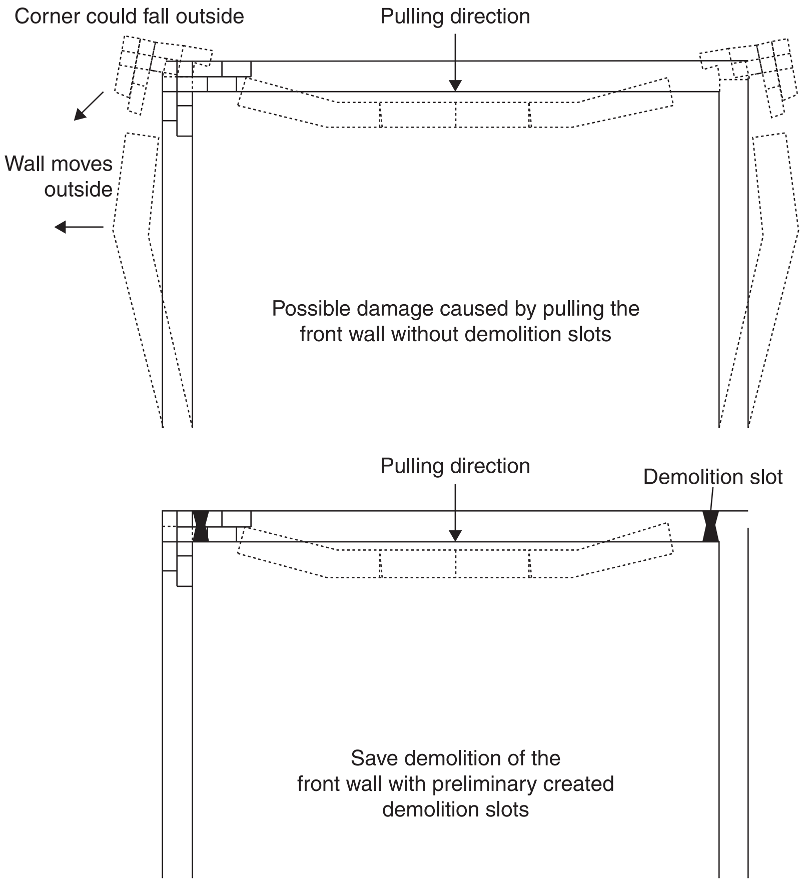
fall, uncontrolled, to the outside (Fig. 8.13). To avoid the risk of corners moving outside of the building, demolition slots have to be applied manually by compressed air hammers. With the front wall and the side walls detached from each other, the deformation during the pulling will be lower and only the front wall will be demolished. The side-walls are then demolished in a separate step.

8.5.4 Other techniques

There are a number of additional possibilities to demolish buildings or parts of a building. In some cases it is helpful not to pull walls or column and beam structures directly, but to use steel ropes to pull parts out, resulting in local collapse (BDHK, 2004). Depending on the applicable load of floor slabs, it may be possible to put excavators on the roof tops of buildings. Roof installations could easily be removed by the use of small excavators.

All techniques should comply with the top-down demolition sequence. The general concept is to demolish parts of buildings without the entire building unexpectedly collapsing. Typical techniques try to weaken the load bearing elements down to a minimum level, such that the final structure can be demolished by removal of a final element.

Of course, all demolition work is done piecewise. One part of the building is demolished up to the next stable level, and so on. Once again, the goal is to reach



8.13 How to demolish front walls made of overlapping masonry structures and how to avoid the danger of corners falling outwards.

a new, static level. Figure 8.9 illustrates such a temporary level. It is clear that, from a certain point, no one should enter a partially demolished building.

8.5.5 Demolition to minimize waste

To avoid waste during demolition, it is important to plan the demolition carefully, and to be aware of the building's construction materials beforehand. In many cases the masonry is partly contaminated. If the chemical behaviour of the different materials is not accounted for, the resulting mineral waste cannot be recycled (Symonds, 1999). If contaminated materials such as rock wool is not identified (e.g. beneath the floors), separation will not be possible after demolition. The key is taking samples prior to the demolition (Poon, 1999). The goal of the chemical

8.14 New materials need the invention of new tools. Deconstruction of insulation plates by an excavator with a newly invented sharp peeler tool.

analysis of such materials is to find hidden contaminants such as heavy metals. Smaller contaminations should also be considered, as it may be possible to use such materials where sealing is possible after rebuilding (e.g. with parking lots). Planning from demolition up to the new building may thus help to avoid landfilling.

Today, not only the contaminations of the past need to be managed, but also that from new materials, which are often combinations of mineral materials such as masonry and plastics (Roussat *et al.*, 2009). All combinations of different types of materials cause a higher rate of general waste if not separated. As one example, Fig. 8.14 shows a heat insulated building. To remove and separate the insulation, a peeling device was created to peel off the non-mineral insulation before starting the demolition process.

8.6 Types and handling of demolition waste

Wastes are defined as residue, substances or materials, generated by the production, transformation or usage, which have been or are planned to be abandoned. Final waste is a waste that undergoes no further treatment and is to be landfilled. Construction and demolition wastes (CDW) include concrete, stones and dirt generated during excavation (sometimes collectively referred to as 'fill material' or rubble), as well as asphalt, wood (treated, painted and clean), metal (ferrous and non-ferrous) and miscellaneous materials (DDC, 2003).

During construction, renovation and demolition activities, one or more of the following types of residuals may be produced:

- clean fill
- recovered materials
- regulated CDW
- hazardous materials and hazardous wastes.

These categories can be defined in more detail as follows:

- **Clean fill** is uncontaminated soil, rock, sand, gravel, concrete, asphaltic concrete, cinder blocks, brick, minimal amounts of wood and metal and inert (non-reactive) solids. When specified as uncontaminated by, e.g. metal-based paints, including lead and other heavy metals, these materials can be used directly for fill, reclamation or other purposes.
- **Recovered materials** are those removed for re-use and those removed to be recycled into new products. Potentially recyclable CDW may include scrap metals, asphalt shingles, sheet rock, lumber, glass and electrical wire.
- **Regulated CDW** are those not classified as clean fill and which will not be re-used or recycled. Regulated non-hazardous CDW must be disposed of at a permitted landfill or transfer station and are regulated by law.
- **Hazardous materials** are those which present some of hazards to human health (e.g. asbestos-containing materials)

In general, demolition has to deal with two different kinds of pollutants:

1. materials which are dangerous to health, i.e. asbestos and rockwool products;
2. materials which are dangerous for the environment, i.e. oil, heavy metals, etc.

The materials dangerous to health must not be re-used. Materials contaminated with products dangerous to the environment need special treatment to be deposited at a safe disposal site. Another main difference is the source of those types of contamination: Other materials may also have become contaminated during the lifetime of the building.

Opportunities for reducing CDW focus on three approaches, typically expressed as Reduce–Reuse–Recycle (DDC, 2003). Reducing waste, the first approach, yields the greatest environmental benefits. Using less material costs less, reduces pollution from its manufacture and transportation, saves energy and water, and keeps material out of landfills. Waste reduction should be the top priority in waste management plans. Therefore economic and sophisticated bridge and building design concepts are required (AMCS, 2009).

Re-using, the second approach, extends the life of existing materials and decreases the new resources needed. Entire constructions can be re-used, for example, through rehabilitation, whether for the same or a new use, saving both resources and money. Re-using means also that the material can be used for the same purpose and in the same manner as it was used prior to demolition. Re-use is possible for all metals through smelting.

Recycling, the third approach, again conserves resources and diverts materials from landfill. Demolition and renovation projects present numerous opportunities for recycling. The most sustainable form of recycling converts waste into new products, such as scrap to new steel or asphalt into new paving. In addition, finding alternative uses for waste constitutes recycling.

Inert waste, such as concrete and brick, can be crushed and used as alternative daily cover for municipal landfills, substituting for dirt, or wood scrap can be burned as boiler fuel. Gravel of concrete or stone is a second-level recycling product. Concrete cannot be used for the same purpose, but may be used for streets or pit fillings of new buildings. Wood of good quality can be assimilated to chipboard. Its use as an energy source is the lowest level of recycling. As well as wood, plastics and roof sheeting are used as an energy source, unless they are too highly contaminated.

8.7 Conclusions

Sustainable construction does not finish at the end-of-life of a building. Since demolition waste is one of the biggest components of all waste, its impact on resources is non-negligible. Thus landfilling should be avoided and use of the recycled material needs to be enhanced. During demolition planning, a four phase concept might be used to help the sorting of different waste types. Recycling rates of 75% are practicable. The planning of the demolition first requires a determination of the method of demolition, with most buildings demolished by excavator work. The advantages of this are the high degree of flexibility due to the possible tool attachments and applicable techniques.

In general, demolition should follow the top-down rule. This technique ensures a safe demolition procedure and dictates that non-load bearing parts shall be removed first, with loading bearing structures, such as beams and columns, removed later. Difficulties of demolition by the top-down method include the difficulty of dealing with steal, even with demolition shears. Safe demolition requires that no parts of the building fall uncontrolled to the outside. Critical parts of buildings include the corners, where side walls and front walls are connected. Here, excavator work needs to be supported by manual labour. Disconnecting the walls ensures that the deformation of the walls is minimized.

Even if demolition work still requires the support of manual labour, the high number of tools designed specifically for demolition helps to minimize dangerous and inefficient methods. New tools, such as the peeler for the removal of insulation, are still being developed. However, demolition is still producing a lot of waste and as such, there is a need for alternatives and recycling to make demolition more eco-efficient.

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9
Preparation of concrete aggregates from construction and demolition waste (CDW)

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Abstract: This chapter describes and provides data on relevant technological, economic and environmental aspects of operating construction and demolition waste (CDW) recycling facilities which, among other output materials (e.g. wood, plastics, metals), produce average to high-quality recycled concrete aggregates. It defines and frames what is meant by recycled concrete aggregates' quality and what it might take to achieve, at an industrial level, that same quality. Market and economic conditions are discussed, focusing on the CDW recycling facility perspective, as well as environmental consequences of its operation, in a life-cycle assessment (LCA) approach.

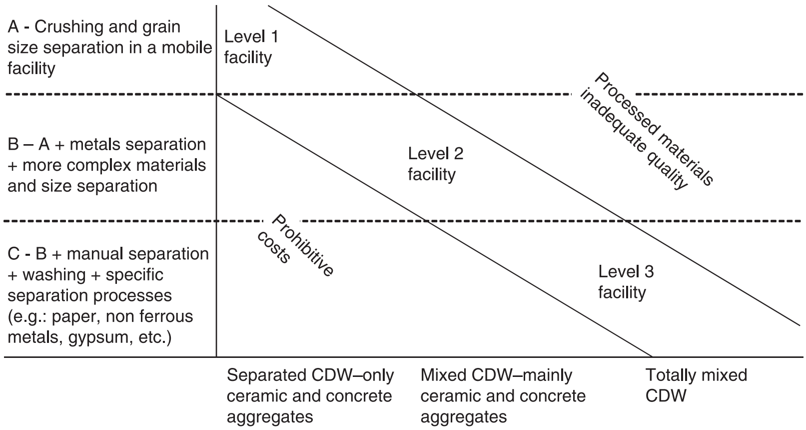
Key words: recycled concrete aggregates, CDW fixed recycling plant, recycling plant technology, economic and environmental analysis.

9.1 Introduction

According to Symonds Group (1999), construction and demolition waste (CDW) recycling plants can be classified according to three main levels: Level 1, 2 and 3. Although the degree of complexity, technology use and range of processing activities rises from level 1 facilities up to level 3, that does not imply that level 1 facilities are inferior to level 3 installations; the level is merely indicative of the technical and economical viability of a given recycling facility, when supplied with given types of CDW input. Figure 9.1 presents a general plot of this proposed classification.

Recycling plants and CDW management networks have recently been studied, focusing on technical aspects (Mulder *et al.*, 2007; Weihong, 2004; Tam and Tam, 2006; de Vries *et al.*, 2009; Nascimento *et al.*, 2005; Algarvio, 2009), as well as economic performance (Duran *et al.*, 2006; Nunes *et al.*, 2007; Pereira *et al.*, 2004; Tam, 2008; Zhao *et al.*, 2010). However, direct environmental impact has been generally overlooked as far as CDW recycling plants are concerned, although some waste management networks have been investigated (Chong and Hermreck, 2010; Blengini and Garbarino, 2010; Weil *et al.*, 2006; Hiete *et al.*, 2011; Marinković *et al.*, 2010).

Concrete waste can account for as much as 75% of waste in demolition activities (especially from the demolition of more recent services buildings) (Coelho and de



9.1 General viability plot for CDW recycling plants, classified into three levels (adapted from Symonds, 1999).

Brito, 2010b). On average though, including demolition of housing and office buildings, buildings retrofitting and public works (roads), concrete waste accounts for around 12% of mass (Coelho and de Brito, 2012), which is nonetheless considerable for a single material in the CDW flux.

In order to be used again in concrete production, recycled aggregates must be virtually free of contaminants. Materials separation technology has evolved considerably in the last few years, and is now able to automatically separate all contaminants – mainly paper and cardboard, glass, gypsum, ferrous and non-ferrous metals, wood and plastics – from concrete aggregates, with an efficiency of 98% (Mulder *et al.*, 2007). Several techniques, such as air sifting, eddy current magnetic separation, dry density separation and spirals are today applicable to CDW recycling, which make it possible to turn traditionally landfilled mixed materials into valuable products, yielding important environmental benefits (Coelho and de Brito, 2012).

Studies have presented various results in terms of the economic viability of CDW recycling plants. For instance, Nunes *et al.* (2007) and Peng *et al.* (1997) have concluded that, within the specific conditions of their studies, operating a CDW is not profitable for private investors. Others have pointed to possible economic viability dependent on certain conditions (Pereira *et al.*, 2004), for example by only operating used equipment (Zhao *et al.*, 2010). Significant economic opportunities were also identified in Duran *et al.* (2006) and Coelho and de Brito (2012), as a consequence of average to high disposal costs and the possibility of charging for waste tipping at the plant's gate.

Waste management facilities are crucial installations in terms of lowering the environmental impact of production industries, construction-related or not. This is accomplished by avoiding traditional raw materials used for industrial processes in favour of recycled materials, which are produced in waste treatment plants. Typical municipal solid waste (MSW) in Italy has been shown to save as much as 87 and 115 kg of primary energy and 27 to 155 kgCO_2 per ton of processed waste (Buttol *et al.*, 2007; de Feo and Malvano, 2009). This can be exceeded by CDW recycling plants which, in part due to the denser nature of the materials they process, can enhance primary energy savings, per processed mass unit, by a factor of 5, and CO_2 emission savings by a factor of 10 (Coelho and de Brito, 2012).

9.2 Technological aspects of concrete recycling

Level 3 plants include methods and technologies used in Level 1 and 2 facilities, and also employ extra equipment and tighter quality control and automation, resulting in both higher purity and variety in recovered material fluxes. Installations may be fixed or mobile, as it may make economic and/or environmental sense, even in regions where Level 3 plants are common, to manage CDW locally with a mobile station, if for instance waste source separation and locally used aggregates are expensive. Transportation distance is also critical to the choice of fixed or mobile CDW processing.

9.2.1 Mobile plants

Mobile CDW recycling installations have risen in popularity due to the need for traditionally fixed equipment, such as feeders, crushers, magnetic separators and even vibrating screens, used at different locations and at different times. Sometimes it is better, technically and/or economically, to place the CDW recycling plant itself – even if in a simplified version – at the worksite, instead of transferring the CDW mass to a fixed installation.

Mobile plants will typically be diesel fired, whereas fixed plants are usually connected to the electricity grid and therefore have some inherent advantages, such as higher operation efficiency and lower environmental impacts. Electrical motors, when installed in automobiles, can be as much as three times more efficient than diesel motors (Kendall, 2008). Moreover, electricity may be derived from renewable sources of energy. However, mobile plants reduce material transportation needs, and therefore reduce the noise, dust and gas pollutants typical of diesel motored trucks.

The technology applied in the mobile plant is essentially the same as in fixed plants, though limited to feeders, crushers, vibrating screens and magnetic separators. Its stage of development is similar to that of fixed facilities. Mobile plants are usually mounted on tracks, but there are also some tyre mounted plants commercially available. Their weights vary widely, from 14 up to 215 tons, with most of the used equipment between 30 and 50 tons (Terex, 2008; Metso, 2009). Capacities can also range from 50 up to 1200 ton/h, while remaining fully mobile

9.2 Mobile crushing unit, with magnetic separator, without grain size separation capacity.

(Metso, 2012). Usual features include jaw or cone crushers, hydraulic crusher protection mechanisms, flexible variable speed built-in conveyor belts (single or multiple attachable discharge arms), fully automatic discharge adjustment systems, safety and anti-clogging mechanisms, ferrous metals separator (normally as an option), radio control of essential features (e.g. on/off, jaw crusher opening, discharge speed), track mounted extra heavy-duty steel chassis and built-in or optional dust suppression (hose). An example of one of these machines is presented in Fig. 9.2, which shows a standard low to medium capacity track mounted model, equipped with a single conveyor belt discharger, magnetic separator and side discharger and a jaw crusher.

9.2.2 Level 1 fixed plants

Level 1 fixed plants are similar to mobile plants as far as equipment and technology usage are concerned. Technologically, these plants consist of simple crushing steps, followed by grain size separation (occasionally with ferrous metals separation), delivering the lowest possible recycled material output, only usable in general fills and sub-bases in secondary roads. However, some features can be implemented within a fixed installation, which are not possible with a mobile arrangement, such as use of a manual separation cabin (Fig. 9.3). Occasionally, a Level 1 CDW recycling plant is equipped with an air sifter, which is a relatively

9.3 Manual separation cabin, at a fixed level 1 CDW recycling plant.

cheap and light piece of equipment, consisting of a centrifugal fan and a duct through which the mixed CDW is forced to pass (Fig. 9.4). In this separation step, the air speed will force light materials (i.e. paper, cardboard, plastics and some wood and non-ferrous metals) to be blown away and collected in a separate bin. However, this simple air sifting arrangement will not remove fine and very fine light contaminants, since these will be sheltered by heavier grains and/or attach themselves to the latter by moisture. Moreover, light contaminants separated in a single air sifting step will be removed from the core CDW flow, but remain in a single separated bundle, making separation of its constituents difficult.

9.2.3 Level 2 fixed plants

Level 2 plants are intermediate between Level 1 and 3 plants. These will commonly produce relatively pure output products and in selected grain sizes for certain applications, such as generic fills (all size grains), cover layers (coarse and fine selected aggregates) and buildings' foundation layers (all size selected grains). To produce such average-quality recycled aggregates, which may technically and economically compete with natural aggregates, some specific separation steps must be employed, such as magnets, air sifters and/or float separators. This latter technique consists of a water-filled bin onto which the mixed CDW will be sent, and where all materials/grains less dense than water will float and thereby are separated from the core aggregate fractions. This method is effective, separating

9.4 Air sifting separation arrangement within a level 1 CDW recycling plant.

both coarse and fine light (less dense than water) materials from heavier concrete and ceramic aggregates, but does so irrespective of the type of light contaminant (it generates, as with single stepped air sifters, only one bundle of separated light materials). Moreover, it introduces water into the process, which make the resulting waste fractions 'stickier' (moisture will in part hold the grains together) and generates contaminated sludge, which are considered hazardous waste and are difficult to dewater (de Jong *et al.*, 2004).

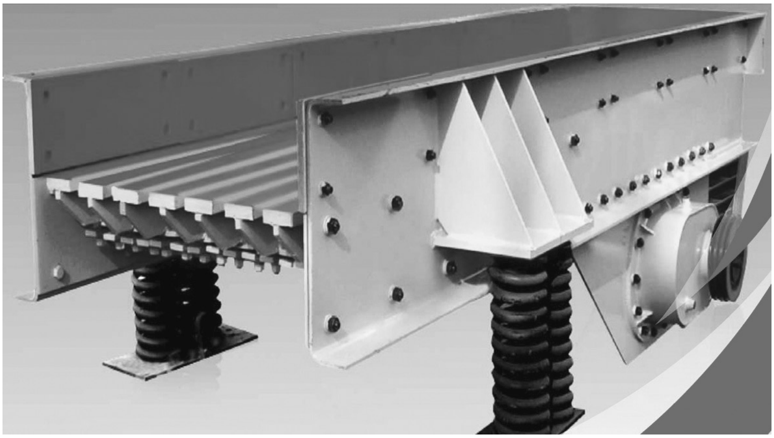
9.2.4 Level 3 fixed plants

Level 3 CDW recycling plants consist of complete processing systems for separating all fractions of mixed CDW. The technological development of these plants is ongoing. Combining optimized crushing steps, widened grain size separation and contaminant separation – air sifters, floaters, jiggers, spirals – it is possible to obtain high purity (e.g. >99%) fractions for all materials.

The following technologies are described within the context of Level 3 recycling plants, although such equipment may also be employed in less elaborate CDW recycling plants.

Feeders

Traditionally used in the mining industry, feeders are used to guarantee a uniform and continuous flow into a single or several crushing units. These can be simple vibrating units (vibrating feeders), pan feeders, grizzly scalpers or mixed type units. Feeders are heavy duty steel framed granular material distributors, usually assembled with bolts (welds can lead to abrupt failure due to fatigue phenomena), which introduce oscillations to the feeding table. This movement can be circular, elliptical or sinusoidal, and serves the purpose of homogenizing the input mix and removing incoming undesired fines. Oscillations are generated by eccentric heads within rotors. The equipment is placed over springs, which absorb all or part of the induced vibrations (Fig. 9.5). Pan feeders and grizzly scalpers are variants of this basic unit. Pan feeders are vibrating feeders without fine separation, and grizzly scalpers can also introduce separation within or between grains (which, for concrete grains, can assure some separation between stone aggregates and adherent cement paste).



9.5 Example of a lightweight vibrating feeder (Break Day, 2011).

Magnets

For ferrous metal separation within industrial CDW recycling plants, permanent or electric magnets are usually placed over the CDW flux (cross belt). The equipment is attached to a side compact conveyor, which can also be a magnetic – magnetic pulley. These are generally built from mild steel, with a steel-manganese face plate (for extra wear resistance) and a variable speed motored pulley. Permanent magnets, unlike electromagnets, are made from a permanent magnetic ceramic material (class 1 to 8) and operate in a completely passive way, without energy consumption. However, electromagnets derive their magnetic attraction force from an input electrical current (alternate), which heats the magnet's face up to 220 °C. Consequently, this equipment is supplied with a pressure valve, aluminium heat exchanger (for dissipation), thermal overload ventilated electrical motor and oil cooling system. For extra magnetic force penetration (within the CDW flux) and effective metals discharge, motored pulleys can also be magnetic (of the permanent kind). An illustrative example is presented in Fig. 9.6.

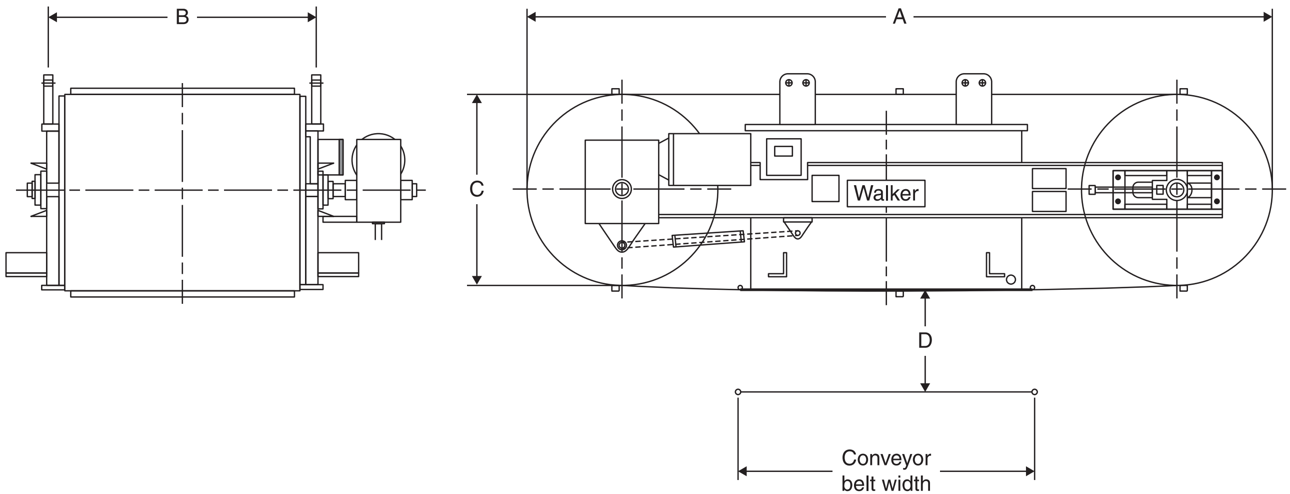
Crushers

In CDW applications, jaw crushers are the most commonly used, mainly due to their heavy duty construction, easy operation, low maintenance and large input openings (Jadovski, 2005). Generally, crushers are made of steel, bolted or welded; their wear parts are reinforced with steel-manganese, steel chromium-molybdenum or iron-chromium alloys; hydraulic relief systems are in place to accommodate different size, density and hardness materials (e.g. steel bars, wood parts). Most modern crushers are controlled by velocity and temperature sensors, and foundation connections assure elastic-plastic behaviour through rubber or neoprene attenuators or springs, in order to limit or eliminate vibration transmission to the support structure. Specifically, jaw crushers make use of an eccentric rotor which moves one of the jaws (the other is static), and are equipped with two massive inertial wheels which have standard or optional toothed plates suitable for different applications. These crushers are preferred for their large input openings (i.e. when compared with impact crushers, for analogous capacities) and relatively small fine grain size production. One of these crushers is shown in Fig. 9.7.

Vibrating screens

Three main types of vibrating screens are used in CDW recycling applications:

1. **'Banana'**: decks are sloped, with at least two inclinations; these are used



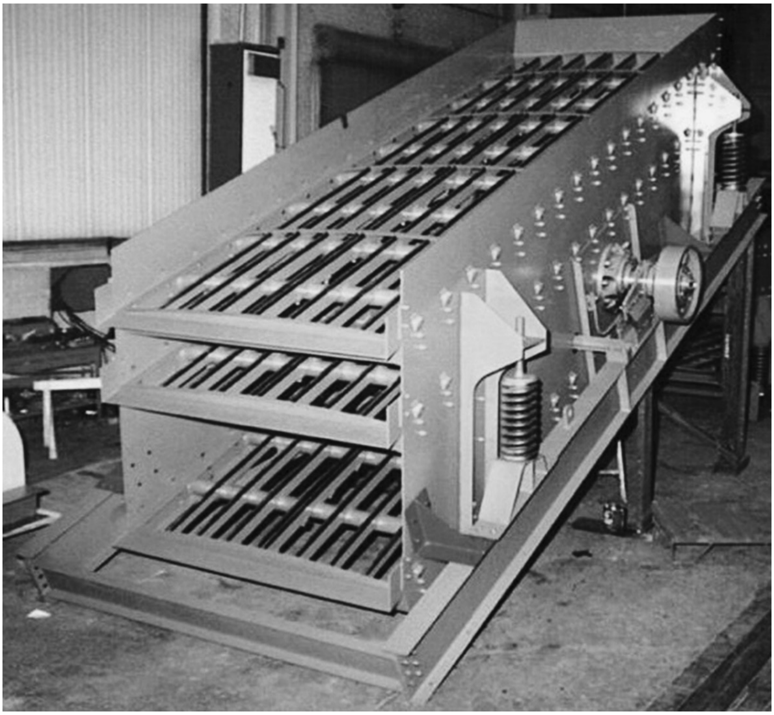
9.6 Views of a cross belt electromagnet, attached to a compact permanently magnetic pulley (Walter Magnetics, 2011*a*).

9.7 Example of a jaw crusher (Metso, 2012).

when there is a large fine percentage of input material in need of separation, i.e. between crushing steps;

2. **Sloped screens**: decks with only one sloped surface, which are more generally applied;
3. **Horizontal screens**: flat decks, particularly suitable for high moisture input flux and/or flux containing a high percentage of friable material.

Vibrating screens are usually heavy-duty welded steel framed, with at least two decks, which are vibrated through eccentric rotor heads. Decks are fabricated in tensioned natural or synthetic rubber or polyurethane, and their oscillatory movement can usually be adjusted. Packages can be supplied with water sprayers and, as with crushers and feeders, are equipped with foot springs or rubber attenuators. An example is presented in Fig. 9.8.



9.8 Example of an (inclined) vibrating screen (Transdiesel, 2012).

Air sifters

Air sifters can be used for automatic light particle separation up to 400 mm in size (Nihot, 2011) or up to 80 mm with a prior hand separation step. These represent a dry separation method and are lighter and cheaper than other methods, such as floating, for equivalent mass throughput. Within the CDW recycling context, gravitational air sifting is the most widely applied, separating particles larger than 2 mm. Sifters can be classified into diagonals, verticals, zigzag or drum (a vertical example is shown in Fig. 9.9). This classification is related to the exhaustion column shape, which changes the obstacle range that the heavier particles must go through, as well as the pattern of separation of lighter particles (Hamatec, 2011). General air sifter characteristics for CDW processing include input pressurized air flow, air recirculation circuit with terminal filtering, steel air ducts and mass separators, sensors and valves for controlling air flow, mass separators and input speed.

Non-ferrous metal separators

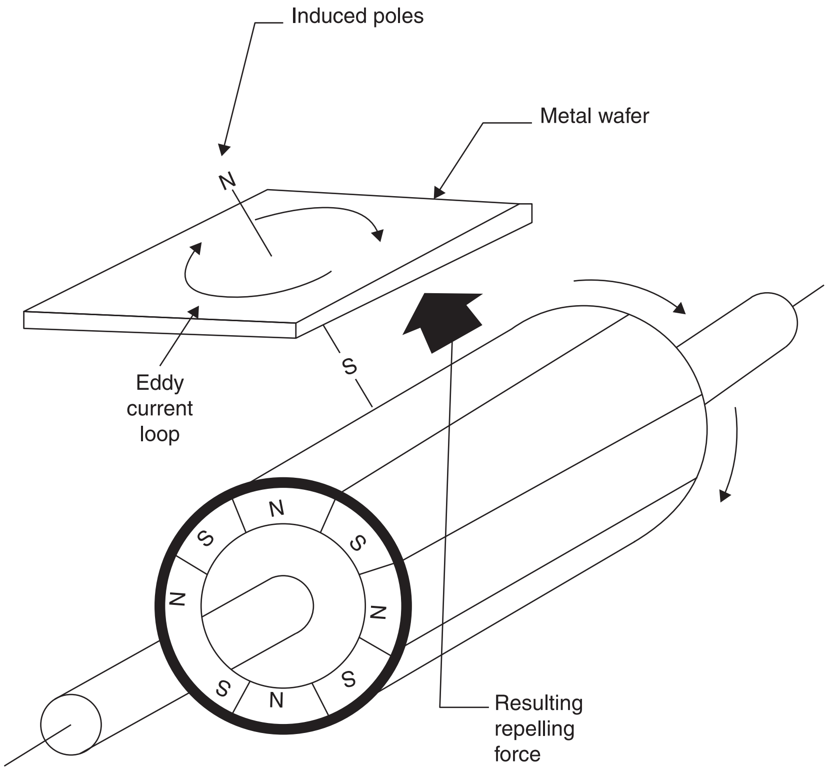
Non-ferrous metal separation is achieved using the spinning of an alternate polarity magnet, which generates circular electrical currents in non-ferrous

9.9 Example of a vertical type air sifter (Hamatec, 2011).

elements present within the CDW flux ('eddy currents'). A force is generated perpendicular to the magnet spinning axis (Fig. 9.10). This pushes non-ferrous metals forward at the horizontal spinning magnet's edge, whilst all other grains fall due to gravity. This equipment is generally composed of a rare earth metals rotor, permanent magnetic (with alternate poles), variable speed heavy duty rotor axis, and a special polyurethane reinforced transport belt. An example is presented in Fig. 9.11.

Air jigs

Air jigs are a variant of the more traditional water jigs. In the former, air is pushed through the CDW flux in continuous and pulsed air fluxes, which pass through an air permeable cross belt. By calibrating these air fluxes, it is possible to create stratified layers of heavy aggregates – namely concrete and ceramic brick aggregates – and therefore separate them. These machines have two air flux inputs, from which air is pushed from the bottom up, and are equipped with

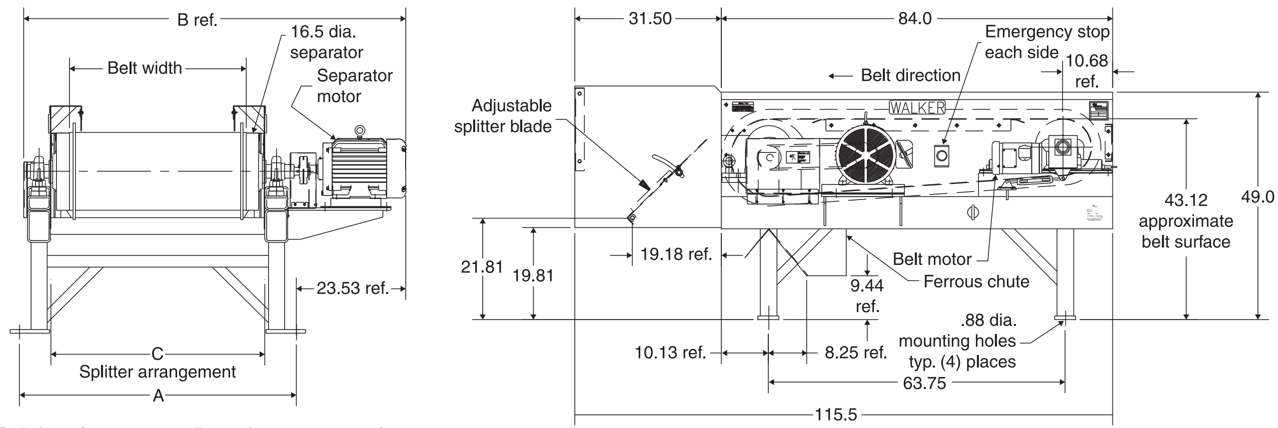


9.10 Operating principle of the non-ferrous metal separator (Walter Magnetics, 2011c).

sensors to measure the separated layers depth. Operating parameters allow particle separation from 1 mm up 50 mm in size. An illustration of the equipment is shown in Fig. 9.12.

Spirals

Spirals are passive separators, relying only on gravitational and inertial forces to achieve grain separation. Particles are separated according to their size, density and, to a lesser degree, shape. Using water as a separation medium, spirals separate fine grains, usually below 4 mm. Lighter particles are swept to the outer rim of the spiral, while denser/heavier particles remain close to the centre region. For effective separation, grains must present an average relative density of at least 1. Solid mass within the water flux must be between 20 and 40%, and the water is recirculated using water pumps. Separated grains are collected, usually at the bottom, within calibrated boxes. Spirals can be simple, double or even triple, for each spiral column (Fig. 9.13), fabricated in rigid polyurethane and fibre glass

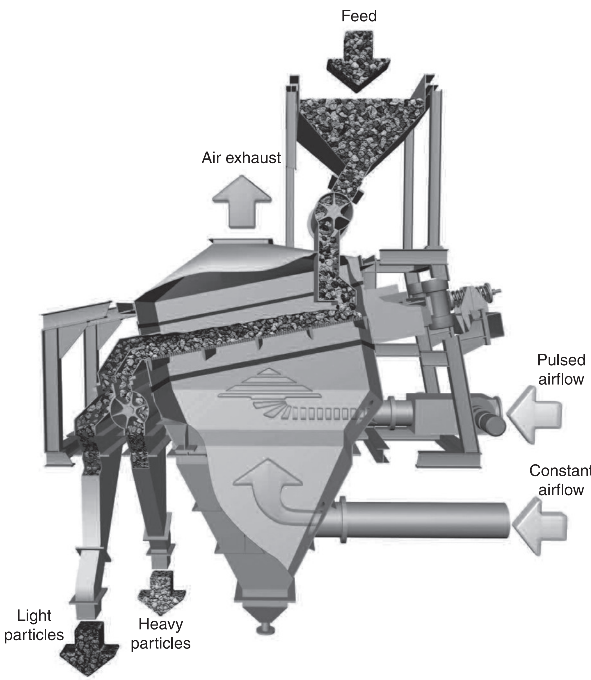


(End view of system excluding splitter arrangement)

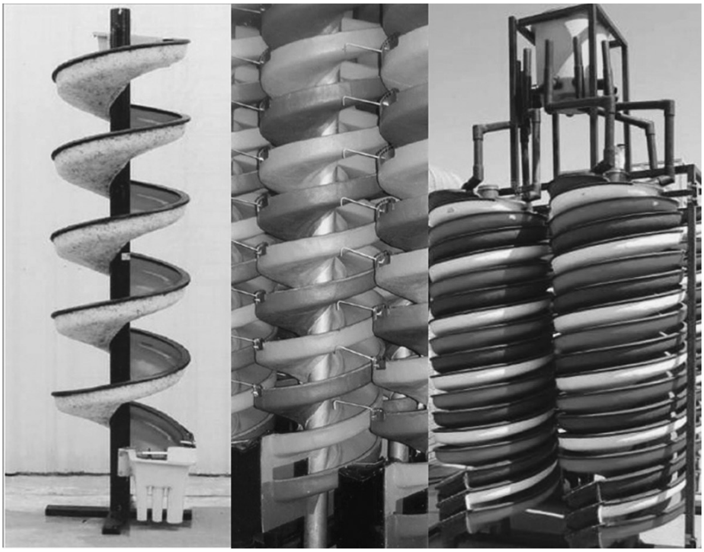
Notes:
1. Variable speed controller not shown.
2. Belt guard not shown for clartiy.

Belt width	Separator motor	Belt motor	A	B	C	Maximum rotor speed	Approximate weight
24 inches	10 H.P.	1.5 H.P.	47.5	65.75	34.0	2500 RPM	2750 LBS
30 inches	10 H.P.	2.0 H.P.	53.5	71.75	40.0	2500 RPM	3050 LBS
36 inches	10 H.P.	2.0 H.P.	59.5	82.0	46.0	2500 RPM	4150 LBS
42 inches	10 H.P.	2.0 H.P.	65.5	88.0	52.0	2500 RPM	4500 LBS
48 inches	15 H.P.	2.0 H.P.	71.5	95.0	58.0	2500 RPM	4900 LBS
54 inches	20 H.P.	2.0 H.P.	77.5	105.0	64.0	2500 RPM	5150 LBS
60 inches	20 H.P.	2.0 H.P.	83.5	106.0	70.0	2500 RPM	5300 LBS

9.11 Side view of a non-ferrous metals separator construction drawing (Walter Magnetics, 2011b).



9.12 Illustrative perspective of an air jig (Allmineral, 2012).



9.13 Spiral separators: single (Mine Engineer, 2011), double (Outotec, 2012) and triple (Outotec, 2010).

(light, wear resistant material), with average heights ranging from 1 m up to 2 m. Collection boxes and input heads are usually detachable and adjustable, and associated piping is made of moulded polyurethane and PVC.

9.2.5 Properties and purity levels in concrete aggregates for recycling

Recycled aggregate products, composed of concrete (ceramic and/or stone) or masonry, or mixtures of the two, are subject to several quality standards, depending on the region. In Table 9.1, a summary of these standards requirements is provided. For recycled aggregate concrete fabrication, at least 90% of input aggregates must be clean concrete (the Netherlands, United Kingdom and Denmark require at least 95% of concrete-originated aggregates). Contaminants, either organic or from other sources, are limited to 5% in the most tolerant regulation (Brazil) and to 1% in the most strict ones (Germany, Belgium, Switzerland). The maximum percentage of contaminants allowed in concrete made with recycled aggregates is 1.5% under most regulations. Some regulations, such as those of Germany and Portugal, consider two recycled concrete qualities, allowing a given masonry aggregate content which cannot exceed 30%; although organic content and other contaminants (covering light materials) must always be kept below 1.5%.

This regulation analysis shows that recycled concrete fabrication standards are very tight, which implies a thorough quality control in CDW separation and especially in purity control of the concrete aggregate fraction. This is achievable with today's technology and is within the reach of most Level 3 CDW recycling plants. Mulder *et al.* (2007) has shown purity levels for aggregate streams of 99.8% using dry density separation (e.g. air jigs) followed by automated X-ray transmission sorting. Weihong (2004) has shown purity levels of over 99% for the concrete aggregate fraction using wet jigging separation (at the final stage, to separate brick and concrete aggregates).

Concrete recycled aggregates are usually more porous than the natural aggregates from which they derive, due to attached cement paste. This is one of the main factors affecting recycled concrete performance, both in mechanical and durability terms (de Brito and Alves, 2010; Gomes and de Brito, 2009). Even though recycled concrete regulations do not require further treatment of concrete aggregates, only its purity regarding possible mixing with masonry aggregates and other contaminants, there are ways of further separating attached cement paste from the original stone aggregates. This has been successfully tested by Mulder *et al.* (2007), through increasing the temperature up to about 700 °C and using the difference in temperature expansion coefficient of cement paste and stone aggregates to effectively separate them. This not only recovers the stone fraction cleanly from the attached cement paste, but also allows the return of the cement paste to Portland cement fabrication, replacing virgin materials.

Table 9.1 Recycled aggregates for construction standards requirements summary (Gonçalves and de Brito, 2010)

Origin of the specification	Classification	Composition (maximum %)					
		Concrete	Masonry	Organic content	Other contaminants	Light materials	Fillers
Brazil	CRA (1)	>90	–	2	3 (4)	n.d.l. (5)	7
	MRA (3)	<90	-	2	3 (4)	n.d.l.	10
Germany	CRA	>90	<10	n.d.l.	1 (6)	n.d.l.	n.d.l.
	CRA	>70	<30	n.d.l.	1 (6)	n.d.l.	n.d.l.
	CMRA (2)	<20	>80	n.d.l.	1 (6)	n.d.l.	n.d.l.
	MRA	>80		n.d.l.	1 (6)	n.d.l.	n.d.l.
Hong-Kong	CRA	<100	–	n.d.l.	1	0.5	4
Japan	MRA	n.d.l.			12 kg/m3		n.d.l.
RILEM	CMRA	–	<100	1	5	1	3
	CRA	<100	–	0.5	1	0.5	2
	NA (7)	<20	<10	0.5	1	0.5	2
United Kingdom	MRA	n.d.l.		1 (6)		1	n.d.l.
	CRA	>95	<5	1 (6)		0.5	n.d.l.
Netherlands	CRA	>95	<5	0.1	1 (6)	0.1	n.d.l.
	CMRA	–	>65	1	1 (6)	n.d.l.	n.d.l.
Portugal	CRA	>90	<10	0.2		1	n.d.l.
	CRA	>70	<30	0.5		1	n.d.l.
	MRA	> 90		2		1	n.d.l.
Belgium	CRA	>90	<10	0.5	0.5 (6)	n.d.l.	n.d.l.
	MRA	>40	>10	0.5	1 (6)	n.d.l.	n.d.l.
	CMRA	<40	>60	0.5	1 (6)	n.d.l.	n.d.l.

Norway	CRA	>94		<5	0.1	1 (8)	0.1	n.d.l.
	MRA		>90		0.5	2.5 (8)	0.1	n.d.l.
Switzerland	CRA	<100		–	n.d.l.	1 (8)	n.d.l.	n.d.l.
	MRA	<100		–	n.d.l.	1	n.d.l.	n.d.l.
Denmark	Tested CRA	>95		–	n.d.l.	n.d.l.	n.d.l.	n.d.l.
	Non-tested CRA	>95		–	n.d.l.	n.d.l.	n.d.l.	n.d.l.
	CMRA		>95		n.d.l.	n.d.l.	n.d.l.	n.d.l.

Notes:
(1) CRA: Concrete recycled aggregates.
(2) CMRA: Ceramic and/or stone recycled masonry aggregates.
(3) MRA: Mixed CRA and CMRA.
(4) Not including organic content.
(5) No defined limit.
(6) Not including bituminous material.
(7) Natural stone aggregates.
(8) Not including bituminous material but including light materials.

9.3 Uses of recycled construction and demolition waste (CDW) materials

The CDW recycling plant must route the materials it separates for further recycling or re-use. Concrete aggregates can account for as much as 41% of all input CDW and almost 50% of all output materials production in a CDW recycling plant (Coelho and de Brito, 2012). Such aggregates are thus the single most abundant material within CDW global input flux. Other materials, such as masonry ceramic brick aggregates, wood, glass, plastics and metals, must also be separated by the CDW recycling plant, contributing to economic payback, and especially to environmental benefits (Coelho and de Brito, 2012). Some other material fluxes will be separated, such as glass, paper and cardboard, bituminous materials, non-ferrous metals, insulation materials and gypsum. Regional estimates are relative to the Lisbon Metropolitan Area (LMA). This is a region with 2 815 851 inhabitants, an area of 3167 km^2 and with 450 574 buildings (according to the latest censuses). These numbers imply one of the highest levels of population density at almost 9 times the national average (112 inhabitants/km^2).

9.3.1 Concrete aggregates

Most recycled concrete aggregates are used for road construction, especially in bases and sub-bases. Demand of concrete aggregates for this application can be as high as 83 000 tons per year, considering average road base and sub-base volumes (based on Portuguese road construction legislation), with recycled aggregates average density of 1950 kg/m^3 and a void factor of 44% (Illston and Domone, 2001). Potential needs for concrete fabrication are estimated at 1640 thousand tons per year, given the data provided by the ready-mixed industry in Portugal (APEB, 2011). This value accounts for 35% cement use by ready-mix producers (of total national consumption), average densities of 1870 and 2400 kg/m^3 for the aggregates and hardened concrete, respectively, and an average of 20% recycled concrete aggregate usage in new concrete fabrication. However, this value is highly conservative, as there is evidence for acceptable concrete properties containing a much higher recycled aggregates percentage (Kou *et al.*, 2004; Richardson, 2010; Zaharieva *et al.*, 2003).

9.3.2 Ceramic masonry

Recycled ceramic masonry aggregates may be used for foundation base layers, ground slab support layers, and as an input for cement fabrication. For foundation base layers alone, an estimated requirement of 439 000 tons per year is possible within the LMA region. This figure was derived from statistical data concerning the number of houses per (building) floor, rooms per house, average room area, and total number of housing and service buildings finished per year (Statistics

Portugal, 2007). From this data a global yearly foundation average area was calculated at around 1290 thousand m^2. This estimate used a 30 cm average foundation base layer depth, which is considered a conservative value. For cement production, if 1% of the regional output was made with recycled ceramic masonry aggregate, it would result in an additional demand of 26 500 tons per year.

9.3.3 Other CDW materials

Wood

CDW recovered wood may be used for wood particle/fibre board production, animal beds and organic fills. An estimated potential demand of 55 800 tons per year is possible, considering a 30% replacement rate of raw wood fibres with recycled wood ones (for an average board density of 600 kg/m^3). In animal bedding, a demand of 8 100 tons per year of recycled wood particles/fibres is attainable, considering a yearly average of 30 100 animals bred in the region (Statistics Portugal, 2007), each needing 22.5 kg/month of bedding material (Pereira, 2005). Organic compost could absorb an extra 16 900 tons of wood particles/fibres per year, considering the 425 000 tons of yearly processed organic waste in Portugal (INE, 2010; Statistics Portugal, 2009, 2010). This demand potential assumes a 15% input of recycled wood particles/fibres in processed organic waste compost (Mota, 2002).

Plastics

Actual recycling of plastics in the Portuguese industry up until 2005 was insignificant, with only 1.8% in mass being recaptured by plastic industries (Texugo de Sousa, 2008). This low value is probably related to lack of awareness, and availability of cheap raw materials, rather than a result of any technical issue concerning plastics recycling. A conservative estimate of 20% was attributed to plastic recycling directly within the industry, corresponding to 104 000 tons per year nationally, or around 28 000 tons per year regionally.

Ferrous metals

From the 1 800 000 tons of ferrous metals produced in Portugal annually (Eurostat, 2009), presently around 27 700 tons per year are recovered by source separation and at MSW incineration plants (Magrinho *et al.*, 2006). An extra 9 200 tons per year of recovered ferrous metals is possible from CDW (Coelho and de Brito, 2010a). If the use of recycled ferrous metals in the Portuguese industry matches that of the European Community, which is about 50% (Commission of the European Communities, 2005), around 882 000 tons per year of material is potentially absorbed by the industry at the national level, which translates into about 234 000 tons per year at the regional level.

9.4 Economic aspects of recycled aggregate for concrete

An accurate economic analysis of any industrial installation must quantify its costs and benefits. CDW recycling facilities costs are related to initial installation, operation (including maintenance and personnel), transportation and rejected materials routing. Installation costs accrue from purchasing processing equipment, as well as the necessary infrastructure – steel frames, foundation blocks, electricity supply, water supply, and so on – for its operation. These installation costs will be replicated through the plant's lifetime as machines reach the end of their service life and must be replaced. Operational costs include energy use, equipment maintenance and personnel directly or indirectly related with equipment operation. Costs of transportation will essentially be related to carrying rejected materials to further processors or landfills, as well as the delivery of separated fluxes to related industries or further recyclers. Rejected materials routing costs refer to gate fees at destinations. Other costs, such as planning/engineering projects, land, taxes, marketing, insurance, legal and accounting services, administrative personnel and financing must also be taken into account.

9.4.1 Mobile recycling plant costs and benefits

A mobile plant's advantage over fixed installations is usually related to reduced transportation costs, as well as reduced fixed installation costs (infrastructure, land). In some cases, when recycled materials, especially concrete and masonry aggregates, are intended to be applied on-site, significant savings result from the avoided transportation requirements. When near other sites which might be in need of these materials, selling may be a more attractive option (instant availability and proximity) than processing them in a relatively remote fixed installation, to and from which the materials must be transported. Zhao *et al.* (2010) have studied the implementation of fixed and mobile CDW recycling plants in Chongqing, China and have confirmed the economic viability of the mobile option, with benefits twice as large as costs, for an 88 000 ton/year processing yield (Table 9.2). The same study also establishes a comparison with a mobile centre in the Netherlands, which is also profitable, although with larger costs and benefits per processed CDW ton (which is as expected, given the countries under consideration). Also Duran *et al.* (2006) analysed a mobile centre operating in Ireland, concluding a record high profitability, with benefits exceeding costs by a factor of almost 40 (for a 720 000 ton/year processing yield).

9.4.2 Fixed and operation recycling plant costs

In order to list and describe the fixed costs of a CDW recycling plant a case study is used, which concerns an analysis performed by Coelho and de Brito (2012). The

Table 9.2 Costs and benefits from fixed and mobile CDW recycling plants in Chongqing, China

	€/ton	Fixed plant in Chongqing, 100 t/h		Mobile plant in Chongqing, 50 t/h		Mobile plant in the Netherlands, 50 t/h	
Fixed costs	Depreciation	0.38	Fixed costs/Total costs, %	0.81	Fixed cost /Total costs, %	0.81	Fixed costs/Total costs, %
	Insurance and cost opportunity	0.11		0.23		0.06	
	Maintenance	0.16	62.5	0.34	73.4	0.34	22.2
Variable costs	Labour	0.07	Variable costs/Total costs, %	0.14	Variable costs/Total costs, %	2.99	Variable costs/Total costs, %
	Energy	0.12		0.16		0.32	
	Transport and dumping fees	0.2	37.5	0.2	26.6	0.92	77.8
Total benefits		2.1		3.8		11.45	
Total costs		1.04		1.88		5.44	
Total costs – benefits		–1.06		–1.92		–6.01	
Annual output, ton/year			240000		88000		88000

Source: Zhao *et al.*, 2010.

most relevant fixed cost is naturally the equipment and infrastructure needed to enable the CDW facility to process the input waste flux. Table 9.3 lists the initial investment needed for a 350 ton/h installation. Equipment costs were determined through a market survey, averaging prices from various suppliers where possible. As machines reach their expected lifetime, they will of course need to be replaced, which will entail further fixed costs. An example is given in Table 9.4, for the expected durability years of each piece of equipment. Equipment installation must be preceded by infrastructure construction, such as steel support frames, foundation blocks and electricity supply, which has been evaluated at 35% of equipment costs. For planning and engineering projects, a further 5% of installation costs (equipment plus infrastructure) will be necessary. Other fixed costs, such as land acquisition, must also be considered. In the case referred to, a market survey in Lisbon's outskirts was performed, which resulted in an average land price of 156 €/m^2, leading to a total 4 280 000 € for an installation area of 27 500 m^2.

Operational costs are mainly related to maintenance, energy use, labour, disposal fees, transportation and capital costs. Equipment costs must naturally include maintenance, which for this case study is listed in Table 9.5. Energy, although used intensively in such a heavy-duty industry, only accounts for approximately 3% of all operational costs. Labour also accounts for a small portion of operational costs, at about 4%. In a Level 3 CDW recycling plant, disposal of rejected materials costs can account for up to 84% of operational costs, and around 80% of all costs within the facility economic lifecycle. Transportation costs are also an important contributor to operational costs, representing about

Table 9.3 Summary table for the detailed process – simplified and full operation modes (equipment fixed costs)

Process step, number	Description	Related equipment	Quantity	Fixed initial costs, €
1	Weighting station	Scales	1	19170
2.1	Visual inspection			
2.2	Loading for manual separation section	Excavator	1	135000
		Conveyer belt #1	1	68833
3	Aggregate size separation (<80 mm; >80 mm)	Vibrating feeder	1	114000
		Conveyer belt #2	1	68833
3.1	Manual separation section	Conveyer belt #3	1	103250
		Hand separation cabin	1	7250
3.2	Crushing section	Jaw crusher	1	130000
4	Ferrous metals separation	Magnet	1	47522
5	Aggregate size separation (>4 mm; <40 mm)	Vibrating screens #1	1	82325
		Conveyer belt #4	1	34417
6	Air separation section	Air sifters	3	300000
7	Non-ferrous metals separation	Eddy current generator	1	98114
		Conveyer belt #5	1	34417
8	Aggregate size separation (<4 mm; 4–8 mm; 8–16 mm; 16–32 mm; 32–40 mm)	Vibrating screens #2	1	82325
9.1.1	Density separation section	Dry density separators #1	4	2753333
		Conveyer belt #6	1	68833
9.1.2		Dry density separators #2	3	1376667
9.2.1	Density and friction separation section	Spirals #1	4	150583
		Conveyer belt #7	1	68833
9.2.2		Spirals #2	4	200777
	Total			5944483

9% of the total (70%, if discounting disposal costs). Finally, capital/financing costs account for less than 1% of operational costs, for a 1.5% interest rate and loan term of 30 years.

9.4.3 Fixed recycling plant benefits

From a regional market survey, prices for marketing recycled products generated by a CDW recycling plant are listed in Table 9.6 (Coelho and de Brito, 2012).

Table 9.4 Expected service life for several types of CDW recycling installation equipment

Equipment	Number of years until replacement
Scales	30
Excavator	20
Conveyor belts	20
Vibrating feeder	8
Hand separation cabin	30
Jaw crusher	10
Magnet	15
Vibrating screens	6
Air sifters	20
Eddy current generator	15
Dry density separators	20
Spirals	15

Table 9.5 Maintenance costs for several types of CDW recycling installation equipment

Equipment	Value	Unit
Scales	1	% of the initial purchase, yearly
Excavator	6400	€/year
Conveyor belts	1.9	% of the initial purchase, yearly
Vibrating feeder	1.4	% of the initial purchase, yearly
Jaw crusher	1.3	% of the initial purchase, yearly
Magnet	367	€/year
Vibrating screens	1.8	% of the initial purchase, yearly
Air sifters	1.9	% of the initial purchase, yearly
Eddy current generator	367	€/year
Dry density separator	0.2	€/ton (per machine)
Spirals	1.9	% of the initial purchase, yearly

Table 9.6 Regional LMA average market prices for recycled materials

Output material	€/ton
Mixed metals	–678
Paper and cardboard	–25
Plastics	–40
Wood	–22.5
Iron and steel	–105
Gypsum	63
Concrete – coarse aggregates	–2.75
Concrete – fine aggregates	–2.75
Heavy metals	–50
Ceramics – coarse aggregates	0
Ceramics – fine aggregates	0

Note: Negative values represent benefits.

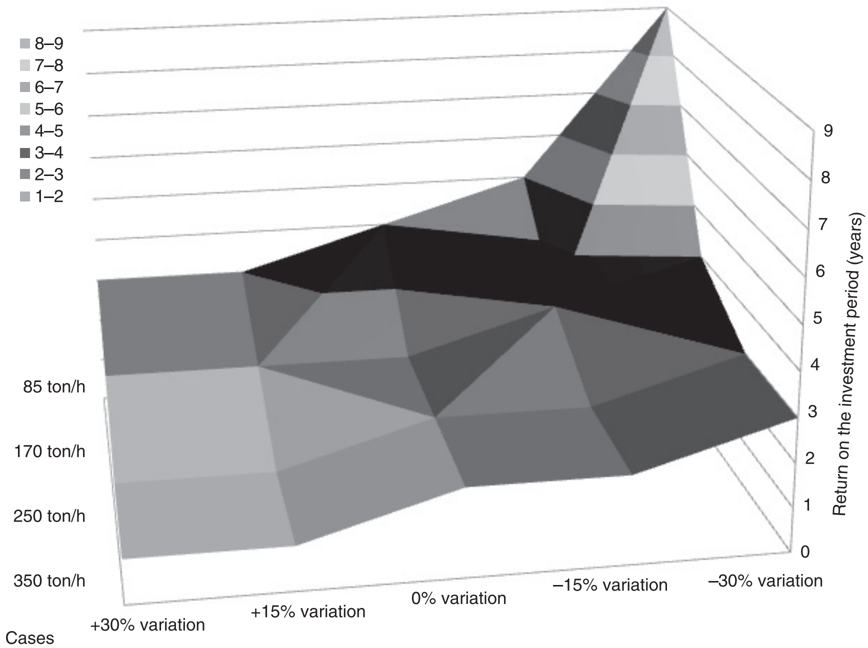
From this table it is clear that, at least in the LMA, concrete aggregates (coarse or fine) are the least valued of recycled materials, although recycled concrete aggregate may sell for as much as four times more in the Netherlands, at almost 9 €/ton (Weihong, 2004). When compared to the value of separated metals (678 €/ton), it is clear that concrete aggregates, both in Portugal and in the Netherlands, are not a scarce resource. This is probably the case in Portugal, because rock stocks are perceived as plentiful (therefore cheap), and in the Netherlands because the road building industry still absorbs most of the recycled concrete aggregates generated. However, this last practice can be considered down-cycling as recycled concrete aggregates, if clean enough (the latter is controlled by regulations), can be successfully used in new concrete fabrication, as described in Section 9.3.1. Within a CDW recycling plant, all output materials sales account for only 14% of all benefits, from which around 5% is due to recycled concrete aggregates sales (considering a 2.75 €/ton selling price).

9.4.4 Fixed recycling plant economic sensitivity analysis

An economic sensitivity analysis concerning the operation of a CDW Level 3 recycling plant was performed by Coelho and de Brito (2012), which looked at the consequences of changing the following operating parameters: CDW input gate fee, recycled concrete aggregate marketing price, reject disposal fees, proportion of mixed/separated flux in input CDW, and amount of CDW input mass. By varying these parameters individually, coupled with the installed capacity, CDW input gate fee and CDW input mass quantity are found to be the most influential parameters. A variation of ±30% in the CDW input gate fee can impact return on investment by a factor of 2.5, and up to 4, while a reduction of input mass of up to 50% can lower the return on investment by a factor of 2.5, and global economic balance (after 60 years of operation) by a factor of 3.4. The latter refers to the smallest facility considered, with 85 ton/h of processing capacity.

The recycled concrete aggregate sales price has the lowest impact on global economic performance, with every variation accounting for a less than 2% change in the return on investment period and global economic performance (for any installed capacity). The other two operational parameters, such as reject disposal fees and proportion of mixed/separated flux, have moderate/intermediate influence on the final results.

An example of the CDW input gate fee influence on the return of investment period can be seen in Fig. 9.14. Whilst not an operating parameter in itself, the installed capacity does have a pronounced influence on profitability due to a scale effect. From the above study it has been shown that the installed capacity alone can be responsible for doubling the return on investment period (from a 350 ton/h down to a 85 ton/h facility), and reduce by almost 6 times the 60 year global economic balance.



9.14 Influence of CDW input gate fee on the return of investment period.

However, globally, and in spite of all the performed parameter variations, the return on investment period is generally below 8 years, and the global economic balance (cost and benefit final accounting for a 60-year operation period) is always favourable to the owner/manager, with profits ranging from 16.6 million € up to 555 million €.

9.5 Environmental aspects of recycled aggregate for concrete

Installing and operating a CDW recycling plant entails environmental impacts. These impacts result from CDW transportation from generation locations – building demolitions, retrofits and new constructions – to the facility, output material routing from the facility to other industries, rejected materials to landfill or further processors, and the plant's own operation (which results mainly from the equipment's energy consumption). Surely environmental benefits also occur from recycling CDW, with recycled concrete as its main mass flux, since replacement of virgin materials entails considerable environmental benefits.

Capturing the environmental factors used in LCA analysis (Ortiz *et al.*, 2009), only CO_2 equivalent emissions (CO_2eq) and primary energy consumption were considered in the following analysis.

9.5.1 CDW recycling facility generated direct impacts

Incorporated impacts

Incorporated primary energy consumption and CO_2eq emissions were derived from the weights of the installed equipment, since environmental impacts are caused by the fabrication of such machines. Apart from the spirals and the manual separation cabin, the first made mainly of glass fibre and the latter of concrete, steel plates and injected polyurethane steel plate sandwich panels, all equipment was considered to be made of steel (which is a reasonable approximation). These impacts are listed in Table 9.7. Just as new costs must be incurred in purchasing new equipment as machines reach their life expectancy, incorporated impacts are also accrued each time equipment is replaced.

Table 9.7 Average incorporated environmental impacts for the studied CDW recycling facility, per installed unit

Equipment	Average weight, kg/unit	Primary energy consumption, kWh/unit	CO_2eq emission, kgCO_2eq/unit
Scales	8825	62315	15709
Excavator	20315	143448	36161
Vibrating feeder	4466	31537	7950
Magnet (ferrous metals)	4458	31479	7935
Manual separation cabinet	(1)	35830	11770
Crusher	35775	252614	63680
Horizontal screen 1	5657	39943	10069
Air sifter	1190	12902	3356
Eddy current generator (non-ferrous metals)	2406	16989	4283
Horizontal screen 2	7341	51839	13068
Air jig	40000	282448	71200
Spirals	1029	6720	1306
Conveyors 5 m	1295	9146	2305
Conveyors 10 m	2307	16293	4107
Conveyors 15 m	3320	23441	11770

Notes:
(1) Environmental impact factors determined per m^2, based on figures from ITEC (2011) (Catalonia institute of construction technology). All other impact factors determination based on data from ICEv2.0 (Geoff and Craig, 2011).

Operation impacts

Operation impacts are essentially related to energy consumption from processing machines. This is dependent on the rated power of the equipment used, and although each may be equipped, for example, with motor frequency actuators, consumption can be calculated in a simplified manner by multiplying the rated power with operating hours. Final energy consumption in kWh is converted into primary energy (measured in kgoe) by applying a conversion factor, which in Portugal is officially taken as 0.29 (RSECE, 2006). Primary energy kgoe can then be converted into CO_2eq emissions by applying another conversion factor, 1.2 (kgCO_2eq/kgoe), which is used in the Portuguese energy certification system (Rodrigues *et al.*, 2009). Table 9.8 lists the operational impacts resulting from equipment operation, assuming operation hours which cover 300 working days per year, 8 hours per day (Zhao *et al.*, 2010).

Transport related impacts

CDW recycling facility transportation impacts can be divided into three main categories: from construction/demolition/retrofit sites to the CDW recycling centre (input material), from the latter to other processors or industries (output material routing) and from the facility to landfills (rejected fraction). For the case

Table 9.8 Yearly environmental impacts from equipment operation, based on rated power and energy source used

Equipment	Rated power, kW/unit	Energy used	Primary energy consumption, kgoe/year (1)	CO_2eq emission, kgCO_2eq/year (1)
Scales	0.05	Electricity	35	42
Excavator	90	Diesel	18576	56322
Vibrating feeder	16.2	Electricity	11275	13530
Magnet (ferrous metals)	6.5	Electricity	4524	5429
Manual separation cabinet	0.28	Electricity	136	164
Crusher	110	Electricity	76560	91872
Horizontal screen 1	18.5	Electricity	12876	15451
Air sifter	6.3	Electricity	9135	10962
Eddy current generator (non-ferrous metals)	16.4	Electricity	7990	9588
Horizontal screen 2	22.3	Electricity	15544	18653
Air jig	127	Electricity	476189	571427
Spirals	27	Electricity	114631	137557
Conveyors	Variable	Electricity	49010	58812

Note:
(1) All installed units in the CDW recycling plant (350 ton/h capacity).

study referred to above, an average distance of 21 km was considered, given an optimum location of the plant within the LMA region (proportional to the CDW generation rates of its different municipalities), which for a 350 ton/h capacity facility leads to a primary energy consumption of 169 200 kgoe/year, and entails an annual CO_2eq emission rate of 545 500 kgCO_2eq. Transportation distances to further recyclers and industries is calculated for each possible routing, which in this case has resulted in a further distance of 625 000 km annually, therefore amounting to 162 000 kgoe/year of primary energy consumption and 522 200 kgCO_2eq/year of emissions. Landfilling of rejected materials, considering a 42 km average transportation radius in the LMA region (Lourenço, 2007), leads to a further 191 600 km/year of necessary transportation distance, and consequently an extra 160 000 kgCO_2eq/year of emissions. A standard 19.3 m^3 volume diesel motored truck was considered to satisfy all transport needs.

9.5.2 CDW recycling facility prevented impacts

Each ton of CDW processed by the recycling facility will be routed to another processor or industrial facility and will be recycled. This means that all the extraction, preparation, transportation and part of the operation impacts attributed to raw materials used in these industries can be avoided by using material fluxes routed by the CDW recycling facility. This leads directly to environmental benefits, since the environmental impacts of these avoided processes no longer take place. For the relevant CDW recycling facility output material fluxes, avoided primary energy consumption and CO_2eq emissions are listed in Table 9.9 (Coelho and de Brito, 2012).

It becomes immediately clear that the larger environmental benefits, at least in respect of primary energy consumption and CO_2eq emissions, result from recycling all but the concrete and ceramic aggregates, although the latter amount to almost 75% of all output mass. This is related to the much larger environmental impacts of producing, for instance, aluminium and plastics on an equivalent mass basis, than those relating to stone aggregates for concrete. For the latter, however, other impacts such as raw materials depletion and landscape alteration may be more prominent, judging from extraction volumes (as an indication, around 18 million tons of stone aggregates are extracted yearly (Mália, 2010), while primary aluminium production stands at 44 000 thousand tons per year (World Aluminium, 2012)).

9.5.3 Environmental sensitivity analysis

In a similar sensitivity analysis method to the one used for the economic performance of the CDW recycling plant, a few operating parameters were chosen which might affect the primary energy consumption and CO_2eq emissions of the facility over its lifecycle. These parameter variations were coupled with the facility's capacity, as

Table 9.9 Avoided environmental impacts due to raw materials replacement (for recycled materials) in several industries

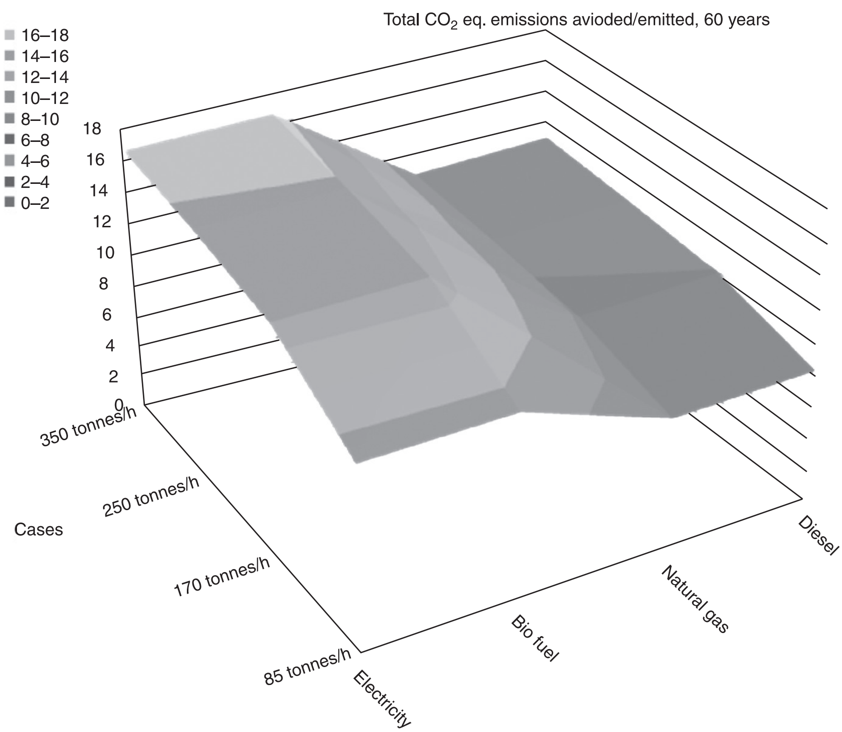
CDW recycling facility output material	Replaced portion of the industrial process	Virgin raw material to be replaced	Industrial processes: prevented energy and CO_2 emissions		
			Primary energy		Emissions
			kWh/ton	kgoe/ton	$kgCO_2eq/ton$
Ferrous metals	From extraction to final stage input (1)	Iron ore	2740	236	805
Non-ferrous metals (mainly aluminium)		Bauxite ore	47083	4048	9944
Heavy metals		Ores of Mercury	24169	2078	5236
		Nickel	45559	3917	13144
		Cadmium	–	–	–
Concrete aggregates (coarse)	From extraction to the factory output gate (2)	Limestone crushed aggregates	12.39	1.07	3.1
Concrete aggregates (fine)					
Ceramic aggregates (coarse)		River/sea sand	9.58	0.82	2.2
Ceramic aggregates (fine)					
Paper and cardboard	From extraction to the factory input gate	Cellulose (3)	5452	469	862
Plastics		Oil derivatives	22363	1923	3310
Wood		Wood particleboard and fibreboard	972	84	168

Notes:
(1) For ferrous and non-ferrous metals, the replaced industrial processes include: agglomeration and fusion (steel production); oxidation and primary alumina precipitation and dewatering (aluminium production).
(2) All recycled aggregate at the plant gate can replace the whole extraction and production stages (of natural aggregates), up to the ready-mixed concrete facility's gate.
(3) Only the wood extraction, production and transport processes are prevented.

the base variation parameter. In the conditions of the case study (Coelho and de Brito, 2012), the parameter with the most impact on CO_2eq emissions over the lifecycle is the type of fuel used in transportation, with an average 32% variation range when compared to the base case (diesel powered trucks).

The alternatives in transportation energy sources considered in the study were natural gas, biofuel (100% rapeseed oil methyl ester) and electricity. In this case, biofuel is the option which most decreases CO_2eq emissions (e.g. the corresponding plant CO_2eq variation is shown, for this parameter, in Fig. 9.15). The second parameter with the most impact on life-cycle CO_2eq emissions is input CDW mass flux, with an average 25% variation range in this case compared to the plant's full capacity (350, 250, 170 and 85 ton/h). The third most relevant variation parameter in terms of CO_2eq emissions is total transportation distance, which was considered to vary as much as ±30%; here, with an average variation of 15%. Two less relevant parameters for global CO_2eq emissions from the CDW recycling plant are the carbon intensity of each supplied electricity unit, and the percentage of mixed/separated CDW input.

This ranking of parameters is maintained for the primary energy consumption standpoint, only now considering that the electricity unit carbon intensity only affects CO_2eq emissions, not primary energy consumption. The best environmental performance – 60 year overall balance, quantified as avoided/emitted CO_2eq or avoided/consumed primary energy – was observed for the 350 ton/h capacity case when the transportation system is converted to biofuel (100% rapeseed oil methyl



9.15 Influence of transportation energy source in the global CO_2eq balance.

ester), with a global relation of 16.7, which means that in this case, the total avoided CO_2eq emissions are 16.7 times greater than generated ones over the facility's life cycle. A minimum value was found for the 85 ton/h capacity plant when input mass is cut by 50%, resulting in 5.3. This case was also a minimum for the primary energy global balance, with a value of 3.2. The maximum result was observed for the 350 ton/h facility when the percentage of mixed/separated CDW input relationship is increased by 30% (which equals about 91% mixed CDW input mass).

9.6 Conclusions and future trends

A technical, economic and environmental analysis was undertaken, concerning primarily Level 3 type CDW recycling plants, which are able to separate to required purity aggregates for concrete production, as well as other material fractions present in CDW input mass. From this study, the following conclusions can be drawn:

- For thorough concrete aggregate separation from mixed CDW, Level 1 CDW recycling plants cannot perform to requirements since equipment such as air sifters, air jigs, non-ferrous metals separators and/or spirals must be used in order to achieve sufficient output purity.
- A Level 3 fully equipped CDW recycling facility, as described above, will be capable of purifying the concrete aggregate fraction in compliance with current legislation concerning the use of recycled concrete aggregates in new concrete production.
- Both mobile and fixed CDW recycling installations can be highly profitable (for the studied fixed installation, the return on investment period is generally below 8 years for most situations), especially in relation to avoidance of transportation needs with mobile plant (and when low-quality output materials are acceptable, given the applications), and for fixed plants where there are large volumes of mixed CDW input mass (and higher standards concerning output material quality).
- In spite of the large input and output mass percentage that concrete aggregate represents within the CDW recycling plant operation, its marketing price is almost irrelevant to the facility's economic performance (which depends more sensitively on input tipping fees).
- The separation of concrete aggregates from CDW mixed input mass, with sufficient quality to be included in new concrete fabrication, brings small environmental benefits when compared, for instance, with ferrous metals, as far as primary energy and CO_2eq emissions are concerned (for the 350 ton/h facility, separating and recycling ferrous metals will avoid around 46% of all avoidable CO_2eq emissions, while concrete aggregates recycling will only account for 4% of these impact savings).
- Overall, separation of the several CDW fluxes and routing of these separated masses to other industries to be recycled has huge environmental

benefits; global CO_2eq balance (avoided impacts divided by generated impacts) can be as high as 10, while primary energy global balance can rise to 8.

9.6.1 Future trends

Although technically possible, separating pure concrete aggregates from mixed CDW and using it to produce new concrete is not usually undertaken anywhere in the world. However, as economically exploitable stone deposits dwindle, and environmental concerns over CDW generation and accumulation rise, the trend can only be to increase the use of recycled concrete aggregates in concrete fabrication. The road construction industry cannot absorb recycled concrete aggregates indefinitely, which means that the growing mass of demolished concrete generated as 30- to 60-year-old concrete buildings decay, will need to be transformed into new concrete.

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10

Separation processes to improve the quality of recycled concrete aggregates (RCA)

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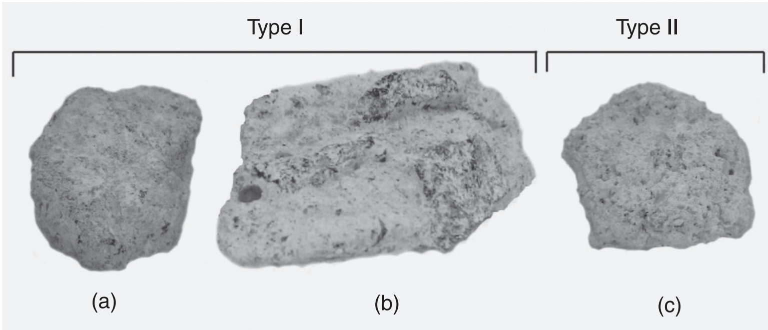
Abstract: After undergoing a number of conventional cycles of crushing and sieving, coarse recycled concrete aggregates (RCA) particles generally comprise one or more particles of natural coarse aggregates surrounded fully or partially by a layer of mortar and/or cement paste. Some of the RCA particles may even be just lumps of mortar embedded with smaller sized natural aggregates (NA). It is this presence of mortar that has been reported as the main factor contributing to quality issues relating to RCA. This chapter begins by discussing how reducing mortar content can improve the properties of RCA and recycled aggregate concrete (RAC). It makes references to available international standards and reviews and compares various state-of-art, effective and innovative beneficiation processes available.

Key words: beneficiation, recycling, concrete, aggregates, mortar content.

10.1 Introduction

Depending on the size, the coarse recycled concrete aggregates (RCA) may comprise one or more particles of natural coarse aggregates, surrounded fully or partially by a layer of mortar and/or cement paste (hereafter denoted as Type I RCA; Figs 10.1a and b) or may essentially be a lump of mortar with varying proportions of smaller size embedded natural aggregates (hereafter denoted as Type II RCA; Fig. 10.1c). There is as yet no practical method to separate the lumps of mortar from the RCA particles with the embedded natural coarse aggregates in a recycling plant, and therefore batches of RCA usually comprise varying proportions of both types (Akbarnezhad *et al.*, 2011). As a result, coarse RCA has relatively lower density, higher water absorption and higher abrasion loss when compared to natural aggregates (NA), rendering it being classified as unsuitable for large-scale replacement of NA used in structural grade concrete (Dhir *et al.*, 2004; Dos Santos *et al.*, 2004; Tam and Tam, 2008).

The mortar present in RCA is considered to be the main factor leading to a lowering of RCA quality when compared to NA (Akbarnezhad *et al.*, 2011, Sanchez de Juan and Gutierrez, 2009). The extent by which the RCA properties differ from those of NA is directly proportional to the mortar content of RCA



10.1 Various types of RCA: Type I comprising one or more NA particles surrounded fully or partially by layers of mortar (a) and (b) and Type II comprising mainly mortar (c) (adapted from Akbarnezhad *et al.*, 2011).

(Section 10.2) (Sanchez de Juan and Gutierrez, 2009; Tam and Tam, 2007). Therefore, particular attention has recently been focused on reducing the amount of mortar present in RCA through fine tuning of recycling process and/or using additional processing stages to remove as much of the mortar as economically viable. The techniques associated with the latter approach are usually categorized as RCA beneficiation techniques.

This chapter begins by discussing how reducing the mortar content can improve the properties of RCA and recycled aggregate concrete (RAC), the latter being concrete cast using RCA as part of its constituents. It then makes references to available literature and reviews and compares various state-of-the-art, effective and innovative beneficiation processes available. Finally, the importance of evaluating the environmental and economic impacts of the various beneficiation methods before implementation is discussed.

10.2 Recycled concrete aggregates (RCA): properties and mortar content

When compared with the typical NA used in concrete, cementitious mortar has a lower density, higher water absorption, lower Los Angeles abrasion resistance and higher sulphate content (Sanchez de Juan and Gutierrez, 2009). It is these differences in mortar and NA properties that have been reported as the main factors contributing to quality issues relating to RCA. The mortar content varies widely from one RCA sample to another, depending on the strength of mortar in the original concrete, particle size, the crushing procedure and number of crushing stages used during production (Akbarnezhad *et al.*, 2013a). As a result of this sensitivity to the parameters involved in the production process, the average mortar contents of the RCA samples reported in literature show a wide scatter and range from 20 to 70% (by mass) (Li, 2008; Akbarnezhad *et al.*, 2013b).

The negative effects of the presence of mortar on a particular property of RCA are proportional to its content (Sanchez de Juan and Gutierrez, 2009). In addition, the relative severity of impact of such effects (the proportionality ratio between mortar content and changes in the respective property of RCA) is basically a function of the ratio between the values of the respective property for mortar and NA. It has been shown that for RCA particles comprising similar types of embedded NA and mortar, the relationship between the mortar content and RCA properties is close to linear (Akbarnezhad *et al.*, 2013b). The most important properties determined regularly to assess the quality of RCA are water absorption, density and Los Angeles abrasion resistance. The relationship between these properties and mortar content are discussed in the following section.

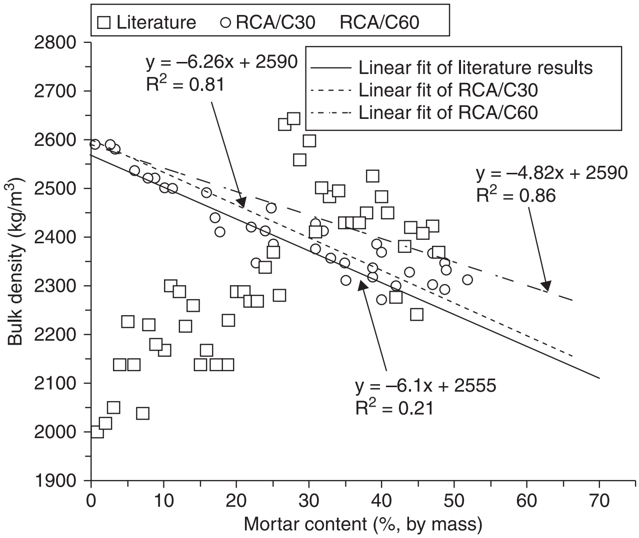
10.2.1 Density and water absorption

Water absorption and density of aggregates are key parameters in mix proportioning to establish weight-volume relationships. Moreover, density and water absorption measurements are among the simplest and fastest tests for determining the overall quality of RCA in ready-mix concrete plants and recycling plants. These two properties are usually considered as being inter-related, showing an almost linear inverse relationship. This is basically because the variations in either of these properties stem from a variation in the mortar content of RCA. The presence of a higher proportion of the intrinsically less dense mortar, vis-à-vis NA, results in a decrease in the bulk density and increase in the water absorption of RCA. The variations in bulk density and water absorption of RCA with mortar content for the two types of RCA with known parent concrete properties (and similar types of embedded NA) and that for a variety of RCA samples with unknown parent concrete properties, as reported in available literature, are shown in Figs 10.2 and 10.3, respectively.

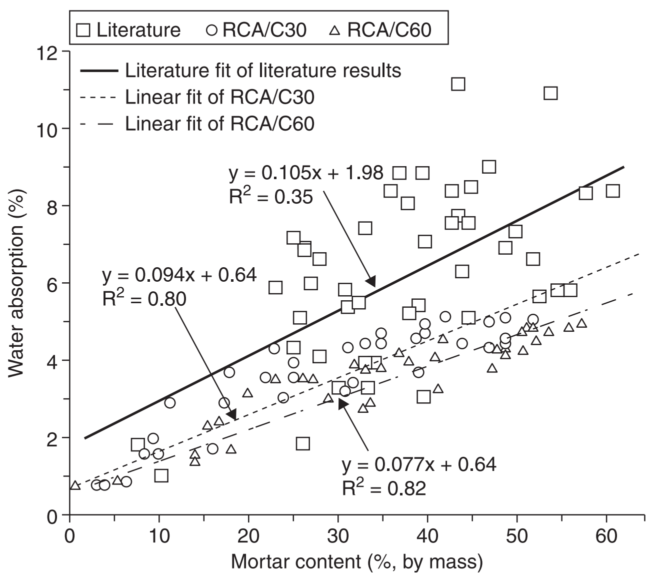
As can be seen, for RCA produced from typical concrete, regardless of the type of parent mortar and NA, there is a high level of proportionality between the mortar content and the bulk density (and/or water absorption) of RCA. For RCA produced from similar concrete sources, the water absorption and bulk density of RCA increase and decrease almost linearly respectively with an increase in mortar content. However, the linear proportionality of such relationships tends to decrease significantly when the RCA used are produced from various concrete sources. This is because besides the mortar content, the variation in the type of the NA and the properties of mortar used in the parent concrete may also contribute to the variations in the RCA properties.

10.2.2 Abrasion resistance (toughness)

Abrasion resistance is an important property of aggregates and is often used as a general measure of the aggregate's quality to resist being worn away or

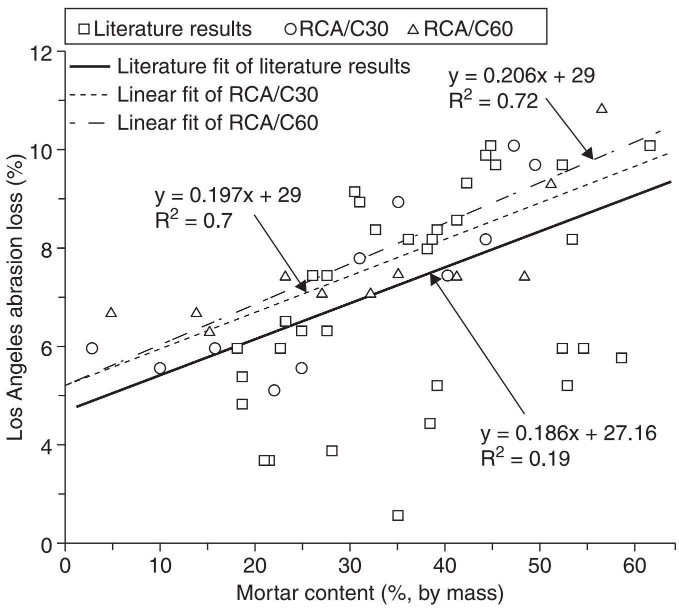


10.2 Relationship between the bulk density and mortar content for the RCAs produced from Grade 30 concrete (RCA/C30), Grade 60 concrete (RCA/C60) and various other concrete sources reported in Sanchez de Juan and Gutierrez, 2009; Tam *et al.*, 2007; and Akbarnezhad *et al.*, 2013b.



10.3 Relationship between the water absorption and mortar content for the RCAs produced from Grade 30 concrete (RCA/C30), Grade 60 concrete (RCA/C60) and various other concrete sources reported in Sanchez de Juan and Gutierrez, 2009; Tam *et al.*, 2007; and Akbarnezhad *et al.*, 2013b.

shattered through rubbing, friction and impact under loading or during handling, stockpiling and mixing. RCA are acknowledged to have on average lower abrasion and impact resistance (Los Angeles abrasion loss >30%) than NA. Nevertheless, the abrasion and impact resistance of RCA is usually within most of the acceptable limits stipulated for structural applications (Tabsh and Abdelfatah, 2009). The lower toughness of RCA has also been attributed to the presence of mortar, which typically has a lower toughness compared to NA. The most popular test for determining the toughness of aggregates is the Los Angeles abrasion test (ASTM C131). It has been observed that the Los Angeles abrasion resistance of RCA decreases proportionally with an increase in the mortar content (Akbarnezhad *et al.*, 2013a; Sanchez de Juan and Gutierrez, 2009). The relationship between the mortar content and abrasion resistance is relatively linear when RCA from similar concrete sources are examined (Fig. 10.4). Similar to the trends for water absorption and density, the linearity of this relationship also decreases with the extent of the variation in the properties of parent concrete.



10.4 Relationship between the Los Angeles abrasion loss and mortar content for the RCAs produced from Grade 30 concrete (RCA/C30), Grade 60 concrete (RCA/C60) and various concrete sources reported in Sanchez de Juan and Gutierrez, 2009; Tam *et al.*, 2007; and Akbarnezhad *et al.*, 2013b.

10.3 Beneficiation of RCAs: innovative methods

As illustrated in Section 10.2, the extent to which the basic properties of RCA differ from those of the original NA is directly proportional to the mortar content. Therefore, efforts have been made recently to produce RCA with lower mortar contents. Two routes have been mainly investigated and proved to be effective in achieving the latter: (a) additional crushing stages; and (b) additional beneficiation processes.

The effects of additional crushing stages have been investigated widely in the available literature (Sanchez de Juan and Gutierrez, 2009; Akbarnezhad *et al.*, 2013a; Tam and Tam, 2008). It is generally recognized that the use of additional crushing stages can reduce considerably the mortar content of RCA, depending on the RCA particle size, strength of mortar and type of crusher used. The reduction in mortar content due to the incorporation of an additional crushing stage has been reported to range from 10 to 40% (Akbarnezhad *et al.*, 2013a). However, additional crushing normally results in a rather significant decrease in the overall yield of coarse RCA, due to breaking of a significant portion of the relatively weaker original aggregates into finer aggregates. In addition, implementing additional crushing stages in the recycling process is accompanied by a considerable increase in costs and therefore a trade-off between the quality of RCA and cost is usually required. Further discussions on the effects of the crushing stages on mortar content are not in the scope of the present chapter and interested readers are encouraged to refer to reference (Akbarnezhad *et al.*, 2013a).

Alternatively, a number of beneficiation processes have been proposed to reduce the mortar content of RCA. These techniques generally use mechanical, chemical or thermal processing or combinations of one or more such techniques to remove the mortar, partially or completely. These methods include thermal beneficiation (Shima *et al.*, 2005), acid treatment (Tam *et al.*, 2007), mechanical rubbing (Yonezawa *et al.*, 2001; Yoda *et al.*, 2003), microwave-assisted beneficiation (Akbarnezhad *et al.*, 2011) and various combinations of these techniques (Shima *et al*, 1999, 2005). In the following, the working principles as well as the advantages and disadvantages of these methods are reviewed.

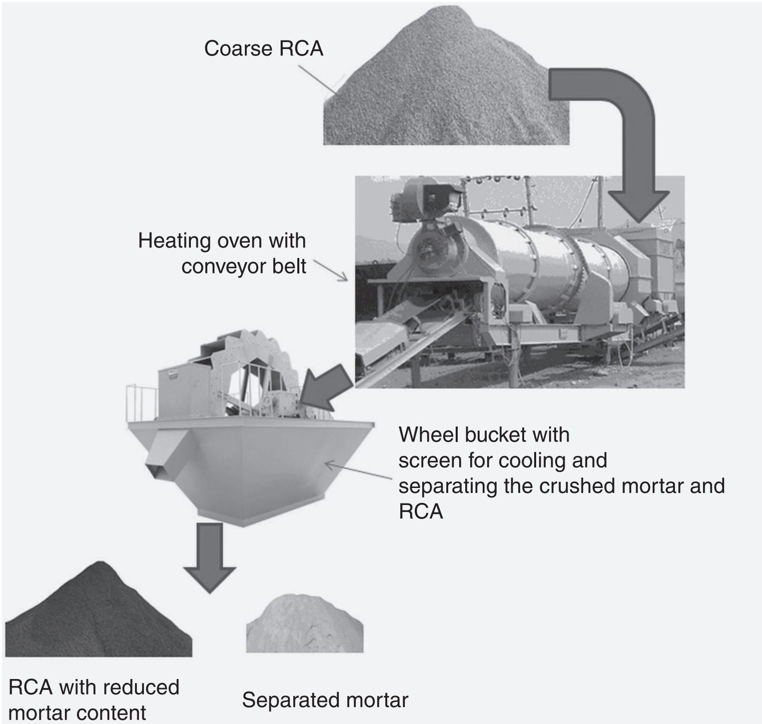
10.3.1 Thermal beneficiation

Thermal beneficiation takes advantage of the differences in the thermal expansion rate of mortar and NA as well as the weaker nature of mortar, vis-à-vis NA, to remove as much of the mortar as possible through differential thermal stresses developed in RCA. Depending on the type of NA and strength of mortar, RCA particles are usually heated at about 300 to 600 °C for about 2 h to break up and separate the mortar (Shima *et al.*, 2005; Noguchi, 2010). Mortar typically has a higher thermal expansion coefficient compared to NA (usually of the order of 2 times) and therefore expands more than NA when heated to a similar temperature. As a result, considerably higher differential thermal stresses are expected to develop in the mortar than in the NA. In thermal beneficiation, such differential thermal stresses are harnessed to break up the intrinsically weaker mortar, vis-à-vis NA, without causing damage to the integrity of NA. Because of the sudden changes in properties, considerably high differential thermal stresses are expected to develop, especially at the interfacial transition zone between mortar and NA, leading to delamination of the adhering mortar in Type I RCA particles.

After heating, RCA particles are sieved thoroughly to separate the broken mortar from coarse aggregates. It has been reported that pre-soaking the RCA batches in water for a few minutes (optimum duration varies depending on the water absorption rate of mortar) to saturate the mortar portion may improve the efficiency of this technique (Sanchez de Juan and Gutierrez, 2009). This is because the rapid evaporation of the internal water due to exposure to relatively high temperatures gives rise to the development of considerable pore pressure within the mortar, leading to a faster fracture of the latter. In addition, quenching of the heated aggregates in cold water (relative to the RCA temperature) after heating, which contributes to an increase in the differential thermal stresses, has been proposed as another method for increasing the efficiency of the thermal beneficiation technique (Sanchez de Juan and Gutierrez, 2009).

The overall methodology of the thermal beneficiation process is summarized in Fig. 10.5. In addition, the results of a study conducted by the authors to investigate the effects of thermal beneficiation at various temperatures on RCA properties are summarized in Table 10.1. As can be seen, conventional heating at 300, 500 and 600 °C reduced the mortar content by only about 6.4% (from 47 to 44%), 12.8% (from 47 to 41%) and 19.1% (from 47 to 38%), respectively. It is important to note that because of the rather uniform heating of RCA in the thermal beneficiation technique (using conventional heating equipment), considerably high stresses may be also developed within the NA present in the RCA during the beneficiation process. Therefore, to avoid such detrimental effects on the quality of the NA present in RCA, the thermal beneficiation method is more suitable for RCA comprising NA that are substantially stronger than the mortar present.

Although thermal beneficiation has a number of advantages, such as the ease of use and operator insensitivity, it also has significant drawbacks including the relatively high energy consumption, low mortar removal efficiency and long processing duration, which render it unsuitable for large-scale industrial application. In addition, heating the RCA to temperatures as high as 600 °C may also degrade the quality of the original NA present in the RCA. For instance, it has been reported that for RCA with granitic NA, the pressure resistance decreases by up to 16% when heated at about 400 °C and by 44% when heated at 600 °C. Also, the tensile strength decreases by 30% and by 60% for the same heating temperatures, respectively (Homand-Etienne and Houper, 1989).



10.5 The thermal beneficiation procedure.

Table 10.1 Effects of thermal beneficiation at various temperatures on the properties of RCA

Type of RCA	Heating temp. (°C)	Process duration (hr)	Properties of RCA		
			24-hr Water absorption (%)	Bulk density (OD) (kg/m^3)	Mortar content (%) by mass
Before beneficiation		0	4.2	2370	47
	300	2	4.1	2380	44
After thermal beneficiation	500	2	3.8	2390	41
	600	2	3.8	2390	38

10.3.2 Mechanical beneficiation

The mechanical beneficiation class of methods apply mechanical forces in the form of rubbing and impact to break up and separate the mortar from the NA present in RCA. The two most commonly adopted mechanical RCA beneficiation methods are the 'eccentric-shaft rotor' (Yonezawa *et al.*, 2001) and 'mechanical grinding' (Yoda *et al.*, 2003). The eccentric shaft rotor apparatus comprises an outer and an inner steel cylinder that rotate eccentrically at a high speed. The RCA batches are passed downward between the two cylinders. As a result of forces exerted by the cylinder walls and the rubbing of RCA against each other, the mortar portion of the RCA is turned into fine powder. The surface of the inner cylinder is in the form of a 2 to 4 mm sieve to collect the separated mortar powders.

In the mechanical grinding method, the RCA samples are rotated using a drum containing a number of iron balls (Yoda *et al.*, 2003). The mortar portion of RCA is removed by rubbing against the iron balls housed in the rotating partitioned sections. Table 10.2 presents the results of an experimental study conducted by the authors of this chapter to assess the efficiency of mechanical beneficiation methods. The experiments were conducted using the Los Angeles abrasion testing apparatus with a load of 10 steel balls. The Los Angeles testing equipment was used to rub the RCA samples (10 kg oven dried weight) against each other and the steel balls for 100, 200 and 300 revolutions of the rotating drum.

After this process, the mortar content of RCA reduced by about 28%, from 47 to 34% and 40 and 64% for 200 and 300 revolutions respectively. As can be seen, relatively high mortar removal rates can be achieved by mechanical grinding using the Los Angeles testing apparatus with an increase in the number of drum revolutions. However, the latter is accompanied by a considerable increase in energy consumption and therefore a trade-off between the quality of the RCA, the processing time and the energy consumption is necessary.

The mechanical beneficiation method is easy to use and relatively more efficient in terms of the overall reduction in mortar content when compared with the thermal beneficiation method. However, it also has a number of well-known

Table 10.2 Effects of mechanical beneficiation using Los Angeles abrasion resistance testing apparatus on the properties of RCA

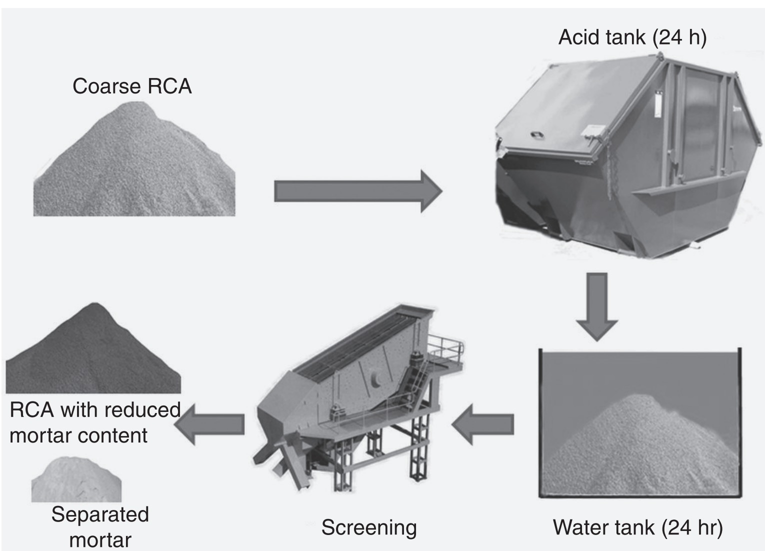
Type of RCA		Process duration (hr)	Properties of RCA		
			24-hr Water absorption (%)	Bulk density (OD) (kg/m^3)	Mortar content (%) by mass
Before beneficiation		0	4.2	2370	47
After mechanical beneficiation	100 (revolutions)	~ 0.1	3.5	2410	34
	200 (revolutions)	~0.2	3.0	2420	28
	300 (revolutions)	~0.3	2.8	2440	17

drawbacks such as high energy consumption and high noise generation. In addition, mechanical beneficiation techniques may reduce the final yield of the coarse recycled aggregates, because a considerable portion of the original NA are also broken down, sometimes to powder, due to the mechanical rubbing and impact.

10.3.3 Acid treatment

Cementitious materials are corroded easily, and thus can be removed, using strong acids because of the alkaline nature of cement (Mindess *et al.*, 2003). Acid corrosion beneficiation technique uses this capability of strong acids to reduce the mortar present in the RCA. In this method, RCA batches are soaked in diluted acidic solutions for durations of about 24 h and are then washed thoroughly and immersed in water for another 24 h to remove the corroded mortar (Fig. 10.6). Subsequently, similar to other beneficiation methods, sieving is used to separate the corroded mortar from the coarser portions of the treated RCA.

It has been reported that among the various strong acids tested for the removal of mortar from RCA, the highest mortar removal rates were achieved using sulphuric (H_2SO_4) and hydrochloric (HCL) acids (Tam *et al.*, 2007). However, besides the mortar removal efficiency, another important parameter in choosing a suitable acid for RCA beneficiation is compatibility with the type of NA present in the RCA being treated. Therefore, it is very important that the acid used for



10.6 Acid treatment procedure.

RCA beneficiation has no or negligible effect on the quality of the NA present. For instance, when the NA used in RCA are predominantly granite, HCL and H_2SO_4 are most appropriate for use to remove mortar. This is because of the significantly lower solubility of the constituent minerals of granite in these acids compared to other strong acids (Sheppard, 1986; Kessler *et al.*, 1940). However, hydrofluoric acid may be considered the worst choice for acidic corrosion beneficiation of granitic RCA, as it is the only acid in which quartz, feldspar and mica, the major constituents of most industrial granites, are easily soluble (Sheppard, 1986; Kessler *et al.*, 1940). Because of the high resistance of granite to chemicals, such acidic corrosion beneficiation methods are known to be especially suitable for RCA containing granitic coarse aggregates. In fact, granite is a popular material for many applications when corrosion resistance is required (Sheppard, 1986).

The efficiency of the acid treatment method depends on various parameters, including the porosities of mortar and NA, type of acid, acid concentration, acid volume, RCA volume, temperature, container type (static vs. dynamic), soaking duration, etc. Table 10.3 lists four different possible acid beneficiation procedures investigated by the authors of this chapter, to study the effects of acid concentration, acid to RCA volumetric ratio, rotary agitation of the RCA/acid samples, as well as the effect of using an additional washing stage to remove the previously corroded mortar from RCA before further acid treatment (Akbarnezhad *et al.*, 2013b). Similar acid (sulphuric acid) and similar soaking duration of 24 h were used in all the experiments.

In addition, 6 different acid concentrations (1 to 6 molar) and 2 different volumetric ratios between the acidic solution and RCA ($V_{acid}/V_{RCA}=2.5$ and 5) were investigated for each procedure. The results of this study (Table 10.4) showed that an obvious increase in the mortar removal efficiency of the acid

Table 10.3 The four different acid treatment testing procedures considered

Action taken	Testing procedure (method)			
	I	II	III	IV
Soaking in sulphuric acid solution (for a total duration of 24 h)	×	×	×	×
Continuous rotary agitation of RCA/acid container (10 ± 1 rpm)			×	×
Washing away the corroded mortar after 8 h		×		×
Replacement of the acidic solution with a fresh acidic solution (after 8 h)		×		×
Washing and cleaning of RCA samples on 4-mm sieve after 24 h	×	×	×	×

Source: Adapted from Akbarnezhad *et al.*, 2013b.

Table 10.4 Efficiency of various acid treatment methods

Acid ($[H^+]$) concentration (mol/L)	Mortar removed/total mortar (%, by mass)							
	Method I V_{acid}/V_{RCA}		Method II V_{acid}/V_{RCA}		Method III V_{acid}/V_{RCA}		Method IV V_{acid}/V_{RCA}	
	2.5	**5**	**2.5**	**5**	**2.5**	**5**	**2.5**	**5**
1	12±4	22±6	27±5	45±8	23±8	79±6	37±4	86±7
2	23±9	54±8	33±4	79±6	35±5	86±5	44±5	94±4
3	35±8	73±4	57±7	88±2	47±10	85±8	42±7	~100
4	43±8	68±2	59±5	85±8	44±7	91±4	67±7	~100
5	51±4	70±7	55±4	91±4	56±3	82±6	77±9	~100
6	48±5	75±7	50±8	88±3	55±5	89±3	70±3	~100

Source: Adapted from Akbarnezhad *et al.*, 2013b.

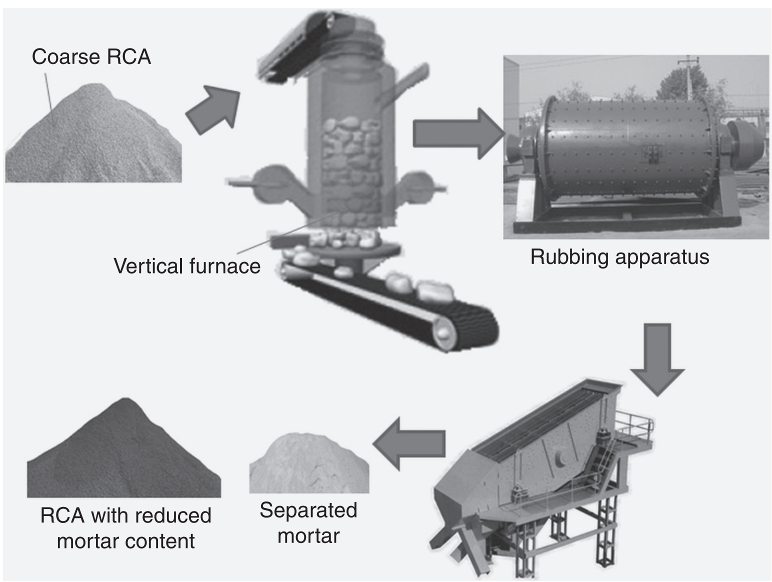
treatment technique can be achieved by increasing the acid concentration and/or the volumetric ratio between the acid and RCA, both leading to an increased presence of the H^+ ions required for corrosion (Akbarnezhad *et al.*, 2013b). However, there seemed to be a particular H^+ concentration, associated with each method, after which the effects of further increases in the acid concentration reduced significantly. This is because when the required amount of H^+ ions for the underlying reactions is already sufficient, further acid corrosion of the RCA is controlled mainly by the permeability of the mortar; the latter is reduced gradually with successive stages of acid exposure as the RCA surface becomes covered with a layer of silica and aluminosilicate gels released by the C–S–H present (Sheppard, 1986).

In addition, it was observed that the mortar removal rate was improved significantly by also incorporating suitable rotary agitation systems and/or adding a washing stage to remove the previously corroded mortar from RCA before further acid treatment. This is because either of these actions result in an increase in the accessibility of acid to reach unexposed mortar and in exposing fresh surfaces for corrosion to take place through the washing away of the previously corroded mortar and aluminosilicate gel material from the already exposed RCA surface (Akbarnezhad *et al.*, 2013b).

Acid treatment at high acid concentrations can be considered an efficient RCA beneficiation technique. However, the use of strong sulphuric and hydrochloric acids results in a significant increase in the sulphate and chloride content of the aggregates after treatment, which may in turn lead to durability problems in concrete. To prevent this, available literature has mainly emphasized the use of relatively low acid concentrations (~0.1 molar) (Tam *et al.*, 2007). However, as shown in Tables 10.4, the use of acid at such low concentrations may lead to only marginal improvements in the resultant RCA properties and thus acid treatment with dilute acids may not be considered an efficient beneficiation technique. Besides the obvious associated durability concerns, this method is more time-consuming compared to other beneficiation methods, taking more than 24 h overall. On the basis of the above discussion, acid treatment techniques at high concentrations has been proposed, mainly as an effective method for the accurate measurement of the RCA mortar content present in RCA rather than a bona fide RCA beneficiation technique in itself.

10.3.4 Thermal-mechanical beneficiation

In thermal-mechanical RCA beneficiation methods, a combination of thermal stresses generated through conventional heating at 300 to 500 °C and the mechanical stresses generated through rubbing of RCA are used to remove the mortar (Shima *et al.*, 2005; Noguchi, 2010). In this technique, also known as 'heating and rubbing', concrete debris is first heated in a vertical furnace to make the cement paste brittle due to dehydration. The heated concrete is subsequently sent to the rubbing equipment for the mortar from the surface of coarse aggregates to be removed. The rubbing apparatus comprises a tube-type mill with outer and inner cylinders (Fig. 10.7). The heated concrete is rubbed by steel balls loaded into the equipment and the mortar portion of RCA separated by rubbing is discharged through screens provided in the inner cylinder. It has been claimed that



10.7 'Heating and Rubbing' RCA Beneficiation Method (adapted from Shima *et al.*, 2005).

this method can increase the quality of RCA significantly, so that it can comply with the JCI (Japan Concrete Institute) standard for high-quality RCA (Shima *et al.*, 2005). However, due to the use of highly energy intensive heating and rubbing treatments, this method is considered as not being environmentally friendly. The energy used and carbon emissions incurred by this beneficiation process may easily counterbalance the environmental benefits gained from recycling.

Table 10.5 presents the results of an experimental study conducted to simulate the original thermal-mechanical beneficiation process described above. In this experiment, RCA samples were heated in a conventional oven for 2 h. The heated RCA batches were then rubbed using a Los Angeles abrasion testing apparatus loaded with 10 steel balls for 100 revolutions. Two different heating temperatures of 300 and 500 °C were investigated. As can be seen, the thermal-mechanical beneficiation at 300 and 500 °C resulted respectively in about 34 and 55% reduction in the mortar content of RCA. The reduction in mortar content can be further increased by increasing the number of drum revolutions. However, this may further increase the cost and energy use incurred and may thus compromise some of the economic and environmental justifications of this beneficiation process.

10.3.5 Chemical-mechanical beneficiation

Chemical-mechanical beneficiation applies combined chemical degradation through exposure of RCA to sodium sulphate solution and mechanical stresses created through subjecting RCA to freeze-and-thaw action to separate the mortar from RCA (Abbas *et al.*, 2008). Although this method can effectively remove the mortar from RCA (almost completely), it requires 7 days to complete, rendering it unsuitable for large-scale beneficiation of RCA. In addition, when processing a large amount of RCA, the long freeze and thaw cycles required for removal of

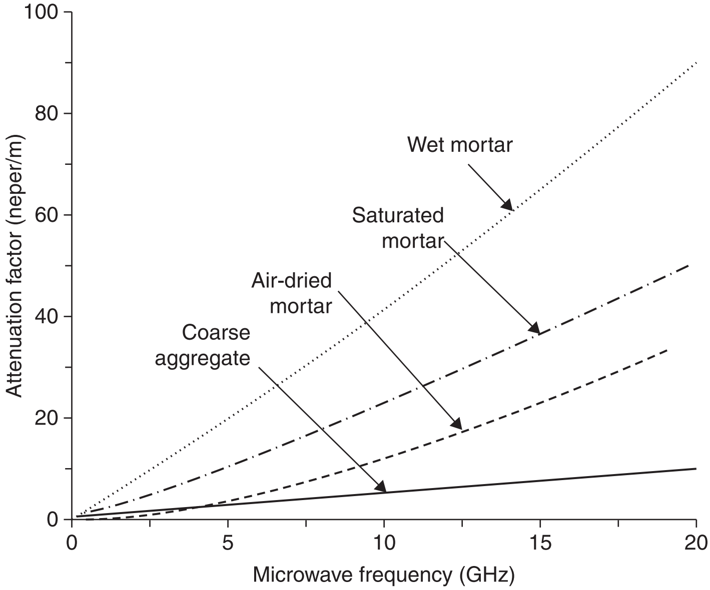
Table 10.5 Effects of thermal-mechanical beneficiation on the properties of RCA

Type of RCA	Heating temp (°C)	Process duration (hr)	Properties of RCA		
			24-hr Water absorption (%)	Bulk density (OD) (kg/m^3)	Mortar content (%) by mass
Before beneficiation		0	4.2	2370	47
After thermal-mechanical beneficiation	300	~2.1	3.3	2430	31
	500	~2.1	2.1	2480	21

mortar render this method highly energy intensive. Therefore, this method has been considered mainly as a method for determining the mortar content of RCA for use in classifying the material rather than a scalable RCA beneficiation method.

10.3.6 Microwave-assisted beneficiation

Natural aggregate and mortar are both dielectric materials and are heated up due to dielectric losses when exposed to microwaves. The extent and pattern of microwave heating of dielectric materials depend on microwave frequency, microwave power and most importantly the electromagnetic (EM) properties of the material. Attenuation factor (β) is an important EM property commonly used to estimate the microwave heating rate. In general, in typical microwave heating cases, the heating rate increases exponentially with the attenuation factor. Figure 10.8 compares the attenuation factors of the mortar and NA. As can be seen, generally mortar has a higher attenuation factor and thus is expected to heat up faster than NA when exposed to microwaves. In addition, the difference between the attenuation factor, and thus the heating rate, of mortar and NA increases further with an increase in the water content of mortar. By taking this into consideration and capitalizing on the significantly higher water absorption of

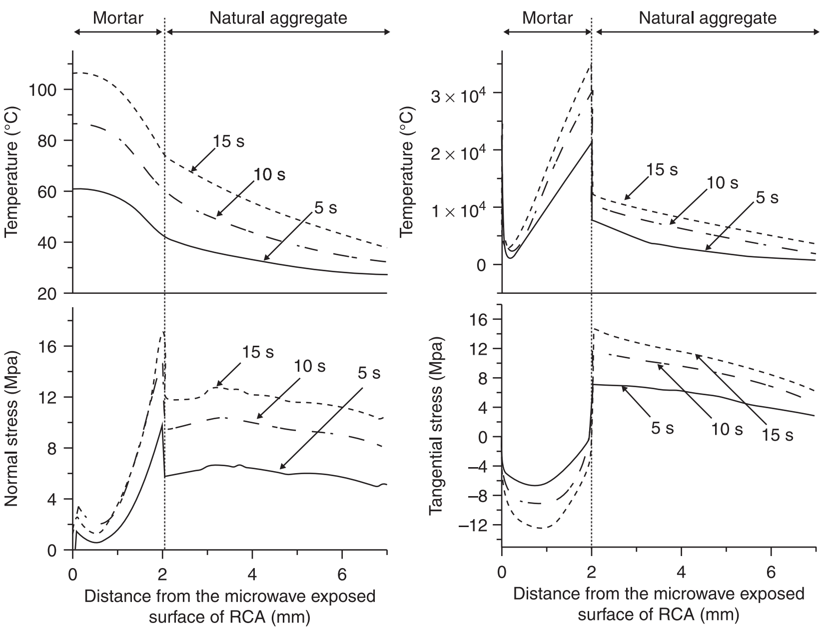


10.8 The variation in the attenuation factor of natural aggregates and mortar with microwave frequency and water content. The microwave heating rate of dielectric materials increases exponentially with attenuation factor (adapted from Akbarnezhad *et al.*, 2011).

typical mortar compared to NA, soaking the RCA particles in water for a short duration (a few minutes) can be used to increase the differences in the water content and thus the microwave heating rate of mortar and NA.

In the microwave-assisted RCA beneficiation technique, the above-mentioned inherent differences between mortar and NA are used to generate a localized field of differential thermal stresses in the mortar and especially at its interface with the NA present in the RCA in a relatively short duration (a few seconds to a few minutes, depending on the microwave power and the volume of the RCA processed) without causing a significant temperature rise in the NA present in the RCA (Akbarnezhad *et al.*, 2011, Lippiatt and Bourgeois, 2012).

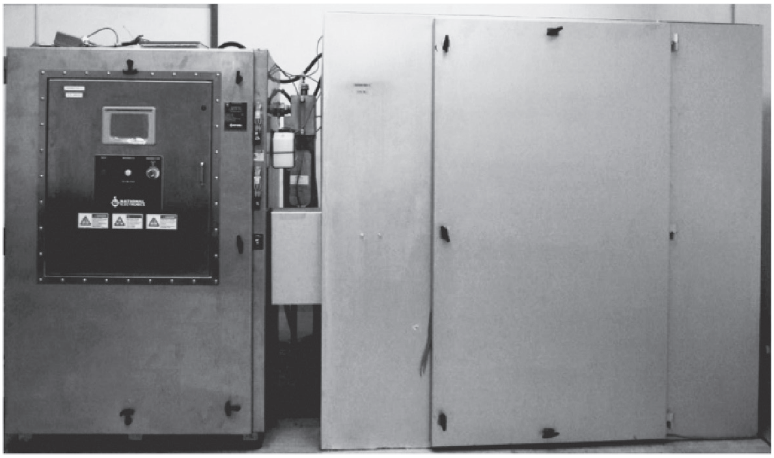
Figure 10.9 presents the results of a numerical simulation study conducted to investigate the temperature rise and thermal stresses developed in a 14 mm diameter spherical RCA, comprising a 10 mm granite aggregate surrounded by a 2 mm thick layer of mortar, during microwave-assisted RCA beneficiation (Akbarnezhad *et al.*, 2011). The results of this study showed that after only a few



10.9 Results of a numerical analysis showing the variation in temperature, temperature gradient, normal stresses and tangential stresses along a straight line normal to the exposed surface of a spherical RCA particle (14 mm diameter) comprising saturated adhering mortar (2 mm thick) and air-dried granite core when subjected to microwaves of 2.45 GHz frequency and 10 kW power. The tensile and compressive stresses are distinguishable by positive and negative signs, respectively.

seconds (5–15 s) of microwave heating using a 10 kW industrial microwave heating system, considerable tensile stresses (~10–16 MPa) may be developed at the interface between mortar and NA that can be harnessed to detach the adhering mortar from RCA. When the microwave-assisted beneficiation technique is fine-tuned to avoid excessive heating, because of the slower heating rate and significantly higher tensile strength of NA compared to mortar, the respective stresses developed in the NA at the time of mortar delamination and pulverization are significantly lower than the aggregate's tensile strength and therefore any damage to the integrity of the NA present can be easily prevented, unless fissures are originally already in existence in the NA.

The results of an experimental study conducted by the authors of this chapter to investigate the efficiency of microwave-assisted RCA beneficiation are summarized in Table 10.6. In this study, 2 kg samples of RCA were heated at 10 kW power for 1 min using an industrial microwave heating system (Fig. 10.10) and were subsequently quenched by immersing in 25 °C water. To investigate the effects of RCA water content on the mortar removal efficiency of the microwave-assisted beneficiation technique, the experiments were conducted on either saturated or air-dried RCA samples. The results of this study (Table 10.6) showed that promising mortar removal rates after a very short duration (1 min) of exposure can be achieved using microwave-assisted RCA beneficiation technique. The efficiency of the microwave-assisted beneficiation seemed to increase significantly by pre-saturating the RCA samples.



(a) (b)

10.10 A 10 kW microwave-assisted RCA beneficiation system comprising: (a) a microwave generator unit; and (b) a RCA microwave heating chamber (adapted from Akbarnezhad *et al.*, 2011).

Table 10.6 Effects of microwave-assisted beneficiation on properties of RCA

Type of RCA		Process duration (hr)	Properties of RCA		
			24-hr Water absorption (%)	Bulk density (OD) (kg/m^3)	Mortar content (%) by mass
Before beneficiation		0	4.2	2370	47
Microwave-treated	Pre-saturated RCA	~ 0.02	2.8	2460	24
	Air dried RCA	~ 0.02	3.4	2430	32

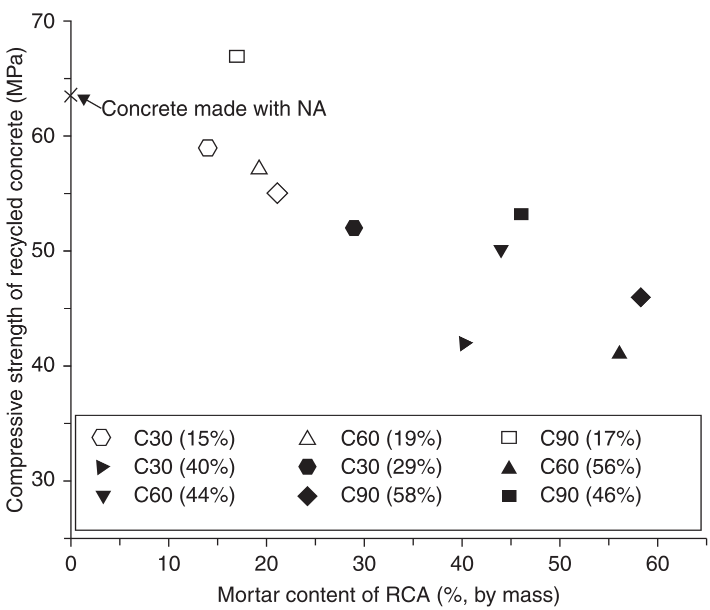
Microwave-assisted RCA beneficiation was observed to result in almost 48% (47–24%) reduction in the mortar content when RCA samples were pre-saturated. Such a high reduction in the resultant mortar content led to almost 33% (4.2–2.8%) decrease in the water absorption as well as 3.8% (2370–2460 kg/m^3) increase in the bulk density of RCA produced. In addition, microwave heating of air-dried RCA particles led to, on average, 32% reduction in the mortar content, 19% reduction in the water absorption and 2.5% increase in the bulk density of the RCA samples tested (Akbarnezhad *et al.*, 2011). Exposing RCA to repeated rounds of microwave-assisted RCA beneficiation may result in more complete removal of the mortar present in the RCA being processed; however, this may compromise the economic and environmental benefits derived and should be carefully assessed prior to large-scale implementation.

Considerably short processing time, posing no concrete durability risks and relatively low energy consumption in comparison with mechanical, thermal and thermal-mechanical methods, are among the advantages of this method. In addition, in microwave-assisted RCA beneficiation, the NA present in the RCA being processed are heated at considerably lower temperatures and for significantly shorter durations (a few minutes only) than in the thermal beneficiation method, and therefore this method is unlikely to cause damage to the integrity of NA present.

10.4 Effects of RCA beneficiation on the mechanical properties of recycled aggregate concrete (RAC)

The effects of a reduction in the mortar content on the basic properties of RCA, including water absorption, density and abrasion resistance, were discussed in Section 10.1. It was observed that reducing the mortar content using RCA beneficiation techniques can result in a proportional decrease in the water absorption and an increase in the bulk density and abrasion resistance of RCA produced. By taking this into consideration, RCA beneficiation techniques are expected to result in considerable improvements in the properties of the concrete made with the recycled aggregates as well. In line with this, the results of a recent study, investigating the compressive strength of the concretes made with acid-treated RCA, showed that for RCA comprising similar types of NA and mortar, the compressive strength of recycled concrete decreased generally with an increase in the mortar content of RCA (Fig. 10.11). However, it should be noted that although the acid treatment technique used in this study is not considered a viable candidate for large-scale beneficiation of RCA, due to the potential durability concerns, one of the major advantages of the acid treatment technique is that the mechanical strength of the embedded NA is not compromised during processing, especially with proper selection of the acid used. Thus any reduction in the mortar content of RCA is likely to lead to an improvement in performance of RAC.

This, however, is not always the case when other beneficiation techniques are applied. As discussed in Section 10.3, most of the mechanical and thermal beneficiation techniques, or combinations of these techniques, usually apply considerable stresses to NA present and this may also concomitantly affect the quality of RCA produced after beneficiation. This should be taken into consideration, as it is important in evaluating the potential effects of a particular RCA beneficiation process on the mechanical properties of the RAC produced.



10.11 Variation in the compressive strength of recycled aggregate concrete with the mortar content of RCA. The C30, C60 and C90 labels represent the grade of the parent concrete. The value in the parenthesis shows the mortar content of the RCA.

Among the available beneficiation techniques, only the effects of microwave-assisted RCA beneficiation on the concrete properties have been comprehensively investigated to date, albeit on granitic aggregates.

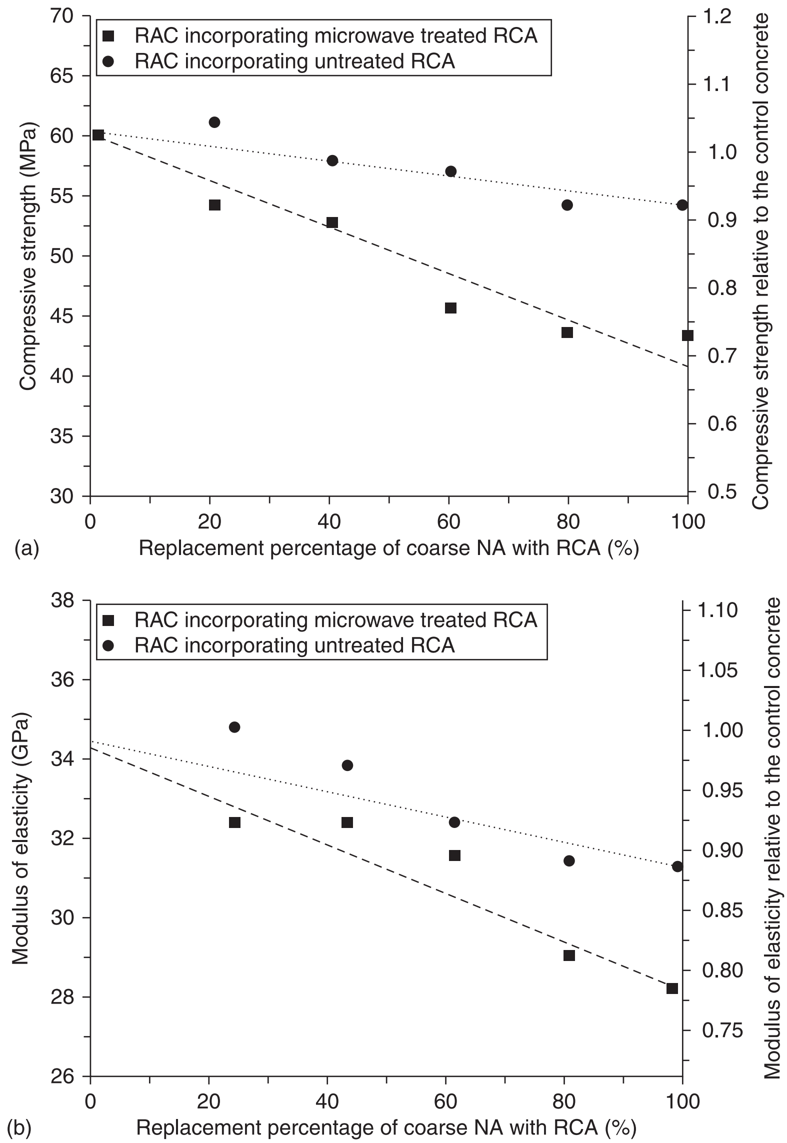
Comparison between the 28-day compressive strength of the RAC incorporating various amounts of RCA treated with the microwave-assisted beneficiation technique and that of the RACs made with original untreated RCA, showed that the negative effect of replacement of NA with RCA on the strength of RAC can be significantly reduced through RCA beneficiation (Fig. 10.12) (Akbarnezhad *et al.*, 2011). Incorporating 100% coarse microwave-treated RCA in place of NA has been reported to result in only a 10% reduction in the compressive strength of RAC as compared to almost 30% reduction when similar amounts of untreated RCA were used. Moreover, incorporating up to 40% microwave-treated RCA was reported to result in only a negligible reduction in the compressive strength of RAC produced (MRAC).

The use of microwave-assisted beneficiation has also been shown to improve the modulus of elasticity and the flexural strength of RAC (Akbarnezhad *et al.*, 2011). The reduction in the modulus of elasticity and flexural strength of concrete due to replacing NA with the microwave-treated RCA were observed to be only about 10%. This may be compared with the up to 25% reduction in the modulus of elasticity and 15% reduction in the modulus of rupture when NA were replaced with untreated RCA. Besides the decrease in the mortar content of the RCA used, breaking of the weakened RCA particles as a result of the stresses developed during microwave heating may also help to contribute to the significant improvements achieved in the mechanical properties of RAC produced with RCA obtained through microwave-assisted beneficiation (MRAC).

10.5 Economic and environmental assessment of RCA beneficiation

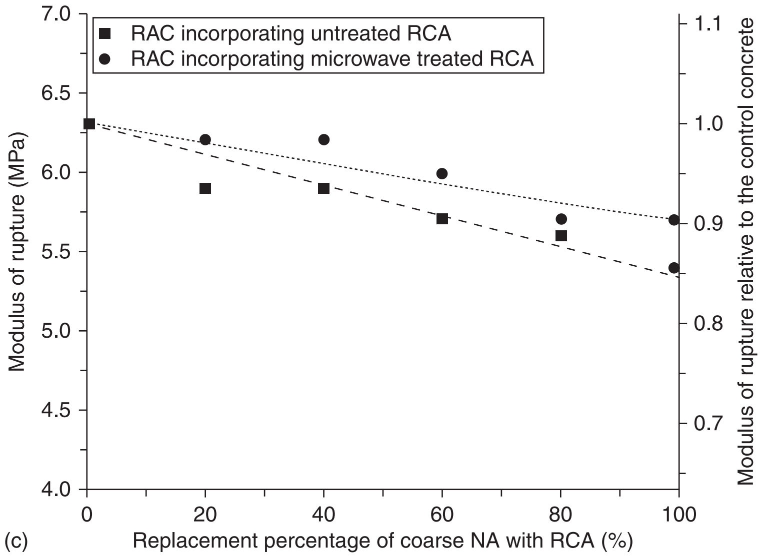
All the RCA beneficiation techniques described in this chapter require a variety of additional mechanical and thermal processes including heating, rubbing, sieving and conveying, which result in an increase in the overall cost, energy use and carbon footprint of recycling. Therefore, before implementing a particular beneficiation process, the economic and environmental impacts of the additional processes should be investigated and compared with those of other alternative options.

The alternative options available depend on the local conditions and requirements and vary from one region to another. For instance, in countries with limited natural resources such as Singapore, concrete recycling plays a strategic role in providing alternative sources of aggregates for the concrete industry. In such cases, the economic and environmental benefits of recycling help satisfy a strategic need of the country, especially if there is a possibility of disruption in the supply of imported NA. However, even in this case, the economic and environmental impacts of producing high-quality RCA through various alternative beneficiation processes should be compared with that associated with the sourcing of suitable NA imported



10.12 (a) Compressive strength and (b) modulus of elasticity of recycled aggregate concretes containing untreated RCA (RAC) and microwave-treated RCA (MRAC).

from overseas sources, for example using lower-quality RCA for non-structural concrete and only NA for structural concrete. This may be feasible if there is a reliable supply of relatively affordable imported NA. The latter is not likely to be sustainable nor is it a good strategy to adopt at the national level in the long term.



10.12 (Continued) (c) Modulus of rupture of recycled aggregate concretes containing untreated RCA (RAC) and microwave-treated RCA (MRAC).

The most common approach for quantifying the environmental impact of a process is to estimate the energy use and/or the carbon emissions caused by the latter. The energy use and the carbon emissions incurred by an activity or a chain of activities conducted to develop a particular product are usually summed up and expressed respectively as the embodied energy and embodied carbon of the respective product (Van Den Heede and De Belie, 2012). The total embodied energy of RCA is therefore considered as the total amount of energy consumed in transportation of debris to the recycling plants, breaking of concrete into smaller pieces, crushing, sieving and additional beneficiation processing. Similarly, the total embodied carbon of a product is the carbon dioxide emissions associated with the stages considered in the energy embodiment calculations.

To achieve a good comparison basis, the cost, energy use and emissions incurred at the various stages of concrete recycling (including the beneficiation process) shall be estimated and compared with those of other alternative sources of aggregates. This may be achieved through multiplying the unit values of energy and carbon emissions incurred by each process by the volume of the concrete to be recycled. For illustration purposes, the embodied energy and embodied carbon of microwave-assisted RCA are compared with that of the natural granitic aggregates imported from a neighbouring region through road based transportation in Table 10.7. The distances to the natural aggregate source and to the landfills are assumed to be 200 and 100 km, respectively. The reduction in the carbon emissions and energy use due to eliminating the need for landfilling of concrete debris

Table 10.7 Estimated embodied energy and embodied carbon of 1 ton microwave-treated RCA (including the energy savings from eliminating the need for landfilling) and natural aggregates imported from a remote source

Processes involved	Energy use/ carbon emissions	Microwave-treated RCA	Imported natural aggregates
Production including extraction, crushing, conveying and sieving, etc. (excluding transportation)	Carbon (kgC/t) Energy (Mj/t)	2.5 27.3	4.8 83.0
Beneficiation process	Carbon (kgC/ton) Energy (Mj/ton)	56.5 600	
Transportation (200 km, road base)	Carbon (kgC/ton) Energy (Mj/ton)		47.0 652.5
Saving through eliminating the need for landfilling of debris (100 km to landfill)	Carbon (kgC/ton) Energy (Mj/ton)	23.5 326.6	
Total (consumption – savings)	Carbon (kgC/ton) Energy (Mj/ton)	**35.5** **300.7**	**51.8** **735.5**

through recycling was also considered in estimating the overall impact of each recycling option. As can be seen, due to the very high energy consumption and carbon footprint of transportation, microwave-assisted RCA beneficiation seemed to be a more environmentally friendly option compared with the import of NA from remote sources. Similar methodology may be used to evaluate the suitability of various other beneficiation strategies before implementation.

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11
Quality control of recycled aggregates (RAs) from construction and demolition waste (CDW)

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Abstract: Various studies have shown that recycled aggregate (RA) can be used to make new concrete. However, in the same way as with natural aggregate (NA), RA also needs to be assessed in terms of grain-size distribution, absorption, abrasion, etc., for it to comply with national standards. In order to evaluate the use of RA, standards and guidelines from 16 countries have been analyzed and compared to determine the most important quality criteria for the physical and mechanical properties of concrete. The results of this analysis have led to the proposal of the set of recommendations included in this chapter.

Key words: recycled aggregate (RA), concrete, quality control, classification, properties.

11.1 Introduction

Over the last 50 years, rising volumes of construction and demolition waste (CDW) have become a cause for growing concern (Debieb and Kenai, 2008). For this reason, the construction sector has made significant efforts to find ways to re-use the huge amounts of this waste that are generated each year. A practical solution for this problem is the use of recycled aggregate (RA) as a replacement for natural aggregate (NA). This has become a frequent practice since it conserves natural resources, decreases production energy and reduces the amount of waste deposited in landfills (Barbudo *et al.*, 2012; Oikonomou, 2005; Uchikawa, 2000). Although demolition waste was first recycled in Germany after the Second World War, only recently has this practice spread to other countries as a promising method of reusing construction waste (Rao *et al.*, 2007).

Various studies have shown that RA can be used to make new concrete (ACI Committee, 2002; Rao *et al.*, 2007; Tam *et al.*, 2008). However, in the same way as with NA, RA also needs to be assessed in terms of grain-size distribution, absorption, abrasion, etc. (Rao *et al.*, 2007), since it must comply with national standards. In order to evaluate the use of RA, standards and guidelines from 16 countries were analyzed and compared, in order to determine the most important quality criteria for the physical and mechanical properties of concrete. The results of this analysis led to the proposal of a set of recommendations.

11.2 Composition and classification of recycled aggregates (RAs)

Research studies have recently combined and integrated various classifications of RA, based on its composition. Most of these classifications are derived from the criteria in the European Standard EN 933-11 of 2009 (Agrela *et al.*, 2011; Barbudo *et al.*, 2012; González-Fonteboa *et al.*, 2011; Jimenez *et al.*, 2011; Mas *et al.*, 2011). Alternatively, RA classification can also be performed 'de visu' (Poon *et al.*, 2002), as well as by the measure of the porous structure of RA using the mercury intrusion technique (Corinaldesi and Moriconi, 2010a), or by petrographic tests to obtain its mineralogical characterization (Calvo Pérez *et al.*, 2002).

It is well-known that the composition of RA can directly affect the mechanical behavior of concrete (Angulo and Müeller, 2009; Angulo *et al.*, 2010). For example, ceramic material can produce adverse effects (Yang *et al.*, 2011), though according to Khatib (2005), there is a higher rate of strength development between 28 and 90 days of concrete attributed to the pozzolanic reaction brought on by the silica and alumina content of ceramic and the products of cement hydration. Similarly, the presence of minor damaging components (e.g. asphalt, gypsum, glass, etc.) is the result of inadequate selection and cleaning at the site where the CDW have been originated. The hand-sorting of aggregate should thus be improved, since visual inspection is not sufficiently accurate and is largely based on the external appearance of the grain (Angulo *et al.*, 2004). Most technical standards thus specify RA types, based on composition, and to a lesser extent, on the density, maximum compressive strength and application of the RA.

Metal impurities in RA concrete have also been studied but the technical documents reviewed do not consider this type of substance separately, but rather mention it in conjunction with various others. More specifically, the standards consulted refer to the percentage of metal, glass, soft materials, bitumen, etc. The maximum permitted values for these substances range from 0.1 to 5% (Table 11.1).

According to Park and Noguchi (2013), aluminum impurities in RA diminish the mechanical properties and durability of RA concrete, even when the aluminum content is minimal (0.1%). This occurs because the aluminum impurities react with the alkaline concrete to produce hydrogen gas. This gas creates gas layers, foam, cracks and rock pockets in hardened concrete, thus causing its mechanical properties to deteriorate. As a result, there is an evident need for more effective screening methods, even though such methods increase the cost of RA and reduce its positive environmental value (Pacheco-Torgal *et al.*, 2012).

Table 11.1 shows the composition-based classification in the national standards and guidelines analyzed in this study. As can be observed, there are basically two or three categories, depending on the presence of concrete and/or ceramic material,

Table 11.1 Classification of recycled aggregate based on composition (in %)

Scope	Standard/ Guidelines	Standard class	Unified class	Concrete	Masonry	Natural aggregate	Organic material	Contaminants/ Impurities	Lightweight materials	Fines
Australia	CSIRO	Class 1A	RCA	<100	–	–	n.a.	1	n.a.	n.a.
		Class 1B	MRA	<70	<30	–	n.a.	2	n.a.	n.a.
Belgium	PTV 406	Crushed concrete debris	RCA	>90	<10	–	0.5	0.5 (a)	n.a.	n.a.
		Crushed mixed debris	MRA	>40	>10	–	0.5	1 (a)	n.a.	n.a.
		Crushed brickwork debris	RMA	<40	>60	–	0.5	1 (a)	n.a.	n.a.
Brazil	NBR 15116	ARC	RCA	>90	–	(b)	n.a.	3	n.a.	7
		ARM	MRA	<90	–	(b)	n.a.	3	n.a.	10
China (c)	DG/ TJ07/008	Type I	RCA	>95	<5	–	0.5	1	n.a.	n.a.
		Type II	MRA	<90	>10	–	n.a.	n.a.	n.a.	n.a.
Denmark	DS 2426	GP1	RCA	>95	–	–	n.a.	n.a.	n.a.	n.a.
		GP2	MRA	>95		–	n.a.	n.a.	n.a.	n.a.
Germany	DIN 4226-100	Type 1	RCA	>90	<10		n.a.	1 (e)	n.a.	1
		Type 2	RCA	>70	<30		n.a.	1 (e)	n.a.	1.5
		Type 3	RMA	<20	>80	<20	n.a.	1 (e)	n.a.	3
		Type 4	MRA		>80 (d)		n.a.	1 (e)	n.a.	4
Hong Kong	WBTC 12	Type II	RCA	<100	–	–	n.a.	1	0.5	4
Japan (c)	JIS A 5021	ARH	RCA	–	–	–	n.a.	3	n.a.	1
Netherlands	CUR	ARH	RCA	>95	<5	–	n.a.	0.1	n.a.	–
	NEN 5905	ARH	RCA	<80	–	<20	n.a.	n.a.	0.1	3

Norway	NB 26	Type 1	RCA	>94	<5	(b)	n.a.	1 (e)	0.1	n.a.
		Type 2	MRA	>90		(b)	n.a.	1 (e)	0.1	n.a.
Portugal	LNEC E 471	ARB 1	RCA	>90	<10	(b)	n.a.	0.2 (f)	1	n.a.
		ARB 2	RCA	>70	<30	(b)	n.a.	0.5 (f)	1	n.a.
		ARC	MRA	> 90		>10	n.a.	1 (f)	1	n.a.
Spain	EHE-08	RCA	RCA	–	<5	–	0.5	(g)	1	2
Switzerland	SIA 2030	BC	RCA	–	<3	–	n.a.	1	n.a.	n.a.
		BNC	MRA	–	–	–	n.a.	2	n.a.	n.a.
United Kingdom	BS 8500-2	RCA	RCA	>95	<5	–	n.a.	1 (h)	0.5	5
		RA	MRA	–	<100	–	n.a.	1 (h)	1	3
	BRE Digest 433	RCA I	RMA	–	<20	>80	n.a.	5	1	n.a.
		RCA II	RCA	<20	–	>80	n.a.	1	0.5	n.a.
		RCAIII	MRA	<10	<10	>80	n.a.	5	2.5	n.a.
	RILEM	Type I	RMA	–	<100	–	1	5	1	3
		Type II	RCA	<100	–	–	0.5	1	0.5	2
		Type III	RCA	<20	<10	>80	0.5	1	0.5	2

Notes:

n.a: no limit available in the standard or guideline.
(a) Less than 5% of bituminous material in all types.
(b) Included in the percentage of recycled concrete aggregate.
(c) This standard classifies recycled aggregate according to its properties.
(d) 20% bituminous materials and others.
(e) For bituminous materials 1% in all types.
(f) Contaminants of bituminous materials. ARB 1 <5%; ARB 2 <5%; ARC <10%.
(g) Bituminous materials <1%; glass, metals, plastics, etc. <1%.
(h) Bituminous materials, RCA<5%; RA<10%.

as well as other components that might affect the quality of the RA. The RA classification proposed in this research is derived from the relevant bibliography and facilitates the analysis of aggregate properties. This new classification, which is based on composition, includes the following categories of RA made from CDW (Table 11.1):

- recycled concrete aggregate (RCA);
- recycled masonry aggregate (RMA); and
- mixed recycled aggregate (MRA).

11.3 Quality criteria for the use of RAs

RA properties are not uniform because of the following:

- the aggregate source (Marinkoviç *et al.*, 2010);
- the recycling plant that produces the CDW aggregate, especially the method used to crush the concrete (Katz, 2003; Mas *et al.*, 2011; Padmini *et al.*, 2009); and
- the lack of quality data for the CDW (Oikonomou, 2005).

The quality criteria to be met by RA for structural concrete must be the same as those required for NA for the same use. This means that RA should be classified according to conventional physical and chemical specifications. The revision of approximately 60 research studies, as well as 26 sets of regulations and guidelines, made it possible to systematize RA quality criteria into four categories based on the following property types:

1. physical;
2. mechanical;
3. chemical; and
4. geometrical.

RA properties and their typical values in these standards and guidelines are described in the following sections.

11.3.1 Physical characteristics of RAs

Table 11.2 shows the physical characteristics of RA in national standards and guidelines, according to RA type. As can be observed, density and absorption are the ones most widely included, regardless of aggregate type. Although porosity is a parameter that is not explicitly in the regulations, it is closely linked to density and absorption. Since it has been studied by numerous authors, it is also mentioned in this section.

As shown in Table 11.2, the physical properties in practically all of the regulations for RCA are oven-dry density and absorption. Those that appear less frequently are bulk density, specific gravity and losses of ignition. Despite density

Table 11.2 Physical requirements for RA in standards and guidelines

Scope	Standard/ Guidelines	Oven-dry density	Surface dry density	Bulk density	Specific gravity	Absorption	Losses of ignition (LOI)
Australia	CSIRO		●✦	●✦	●✦	●✦	●
Belgium	PTV-406	●■				●■	
Brazil	NBR 15116					●✦	
China	DG/ TJ07/008	●				●	
Denmark	DS 2426	●	✦				
Europe	EN 12620	●					
Germany	DIN 4226-100	●■✦				●■✦	
Hong Kong	WBTC 12	●				●	
Italy	NTC	●					
Japan	JIS A 5021 /5022/5023	●				●	
Korea	KS F2573				●	●	
Netherlands	NEN 5905					●	
	CUR	●■					
Norway	NB 26	●	●✦	✦		●	✦
Portugal	LNEC E 471			●	✦	●	✦
Spain	EHE-08					●	
United Kingdom	BS 8500-2						
	RILEM	●■				●■	

Notes: ● RCA; ■ RMA; ✦ MRA.

and absorption being two of the main characteristics that define RA quality, there are regulations that do not include density or any of its variants (EHE-08, 2008; NBR 15116, 2005; NEN 5905, 2010). Others do not include absorption either (BS 8500-2, 2006; CUR, 1984, 1986, 1994; DS 2426, 2009; prEN 12620, 2010; NTC, 2008). In fact, the BS 8500-2 standard does not mention any of these properties.

Density

RA is always less dense than NA because of the presence of adhered mortar, ceramic materials and other impurities, such as gypsum (Katz, 2003; Sim and Park, 2011). Although research has been conducted on how cement paste, such as chloride binding, can benefit RA (Ann *et al.*, 2008), it is less likely to inhibit corrosion and aggressive ions because RA is more porous than NA.

The lower density of RA produces concrete of lower workability and a higher water demand in its fresh state. This type of concrete is also not as strong and is less durable after it hardens. Table 11.3 shows the density types in the references as well as the maximum and minimum density values, depending on RA type (all-in-one, coarse and fine aggregate). The data seem to indicate that higher

Table 11.3 Maximum and minimum density values in the references

Density	Aggregate	Minimum (kg/m^3)	Maximum(kg/m^3)
Oven-dry density	All-in-one	2045 (WRAP, 2007)	2620 (Tam and Le, 2007)
	Coarse	1170 (Limbachiya *et al.*, 2000)	2760 (Angulo and Mueller, 2009)
	Fine	1913 (Evangelista and de Brito 2007, 2010)	2500 (Martín-Morales *et al.*, 2011)
Surface-dry density (10 min)	All-in-one	2470 (Gonzalez-Fonteboa and Martínez-Abella, 2005)	2480 (Gonzalez-Fonteboa and Martínez-Abella, 2005)
	Coarse	2070 (Agrela *et al.*, 2011)	2450 (Fonseca *et al.*, 2011)
Surface-dry density (24 h)	All-in-one	1940 (WRAP, 2007)	2650 (WRAP, 2007)
	Coarse	2060 (Agrela *et al.*, 2011)	2678 (Zhu *et al.*, 2011)
	Fine	1310 (Barbudo *et al.*, 2012)	2650 (Barbudo *et al.*, 2012)
Bulk density	All-in-one	1427 (Padmini *et al.*, 2009)	1568 (Padmini *et al.*, 2009)
	Coarse	1060 (Lovato *et al.*, 2012)	2730 (Angulo *et al.*, 2004)
	Fine	1010 (Debieb and Kenai, 2008)	1530 (Miranda and Selmo, 2006)
	Fines	1320 (Miranda and Selmo, 2006)*	
Specific gravity	All-in-one	2380 (Padmini *et al.*, 2009)	2670 (Tam and Le, 2007)
	Coarse	1860 (Becerra Cabral *et al.*, 2010)	2890 (Bairagi *et al.*, 1993)
	Fine	1850 (Müller, 2004)	2680 (Miranda and Selmo, 2006)
	Fines	2600 (Miranda and Selmo, 2006)*	

*Only typical values have been included in these studies.

density values are linked to larger aggregate grain size (Becerra Cabral *et al.*, 2010; Etxeberria and Vazquez, 2010; Kou *et al.*, 2011a,b; Padmini *et al.*, 2009; Poon *et al.*, 2007; WRAP, 2007). Research also confirms that RCA has higher density values than RMA (Agrela *et al.*, 2011; Angulo and Müeller, 2009; Becerra Cabral *et al.*, 2010; Corinaldesi and Moriconi, 2009; Dapena *et al.*, 2011; Jiménez *et al.*, 2011; Martín-Morales *et al.*, 2011; Padmini *et al.*, 2009; Tam and Le, 2007; WRAP, 2007; Yong and Teo, 2009). In contrast, RMA was found to have the lowest density values (Becerra Cabral *et al.*, 2010; Debieb and Kenai, 2008; Miranda and Selmo, 2006; Müeller, 2004; Poon and Chan, 2007).

Water absorption

Water absorption values also reflect important differences between RA and NA. In the case of RA, the water absorption rate is increased by the presence of adhered mortar (Lopez-Gayarre *et al.*, 2009; Padmini *et al.*, 2009), ceramic material and impurities, such as gypsum (Agrela *et al.*, 2011). These substances are also closely related to density values and affect the behaviour of concrete both in its fresh and hardened state.

The role of water absorption is decisive in the manufacture of concrete. For this reason, specifications can be found in most standards (86% for RCA, 75% for RMA and 62% for MRA). It has an important impact on the relationship between water and binder. If RA is not pre-treated, more water is required to mix the cement paste. This reduces the strength of the product. To solve this problem, some researchers presoaked granular materials and then measured absorption after 10 min, 30 min and 24 h, in order to model absorption behaviour (Evangelista and de Brito, 2010; González-Fonteboa and Martínez-Abella, 2008; Mas *et al.*, 2011; Tam *et al.*, 2005, 2008; Tam and Tam, 2008; Dejerbi, 2012). This presoaking improved the final performance of the concrete, due to the formation of a more solid and denser interface (Barra de Oliveira and Vazquez, 1996; Kou *et al.*, 2011a). It also created an internal water supply that reduced drying-shrinkage.

This pre-treatment was found to prevent excessive water absorption in the aggregate during mixing, and consequently, maintained concrete workability (Corinaldesi and Moriconi, 2010b; Domingo-Cabo *et al.*, 2009). In this sense, certain authors propose the presoaking of coarse RA in a saturated surface-dry condition before the mixing procedure, in order to ensure a uniform blending during the concrete manufacturing process (Debieb and Kenai, 2008; Katz, 2003). Etxeberria and Vazquez (2010) prepared concrete using RA with an 80% moisture content, whereas Tam *et al.* (2007) propose presoaking RA in acid before it is used to make concrete, to remove the attached cement mortar.

In comparison to RA density, the higher the absorption value of RA, the smaller the size of the aggregate (Barbudo *et al.*, 2012; Corinaldesi and Moriconi, 2009; Debieb and Kenai, 2008; Müeller, 2004; Poon and Chan, 2007). Therefore, absorption values of ceramic material (Agrela *et al.*, 2011; Becerra Cabral *et al.*, 2010; Debieb and Kenai, 2008; WRAP, 2007; Yang *et al.*, 2011) are higher than MRA and RCA absorption values (Becerra Cabral *et al*, 2010; Evangelista and de Brito, 2007, 2010; Katz, 2003; Kou and Poon, 2009; Kou *et al.*, 2011a; Martín-Morales *et al.*, 2011; Miranda and Selmo, 2006; Sim and Park, 2011; Zega and di Maio, 2011).

The absorption coefficient of RA, measured at 24 h, was 0.57 to 13.2% (Tam and Le, 2007; WRAP, 2007), although values depended on aggregate type. In the case of coarse RA, values ranged from 1.21% (Angulo *et al.*, 2004) to 15.62% (Becerral Cabral *et al.*, 2010). For fine RA, they were between 2.0% (Miranda and Selmo, 2006) and 30.9% (Poon and Chan, 2007). Finally, in the case of coarse RA from sanitary ceramic material and from electrical ceramic material, the water absorption rate was lower than 1% because of the compactness of these materials (Medina *et al.*, 2012; Senthamarai *et al.*, 2011).

Porosity

Porosity is closely linked to density and absorption. This means that higher porosity values (59.54%) are found in RA from ceramic material (Debieb and Kenai, 2008), and lower values (0.32%) are found in RA from sanitary ceramic material because of its compactness (Medina *et al.*, 2012). No significant differences were found between RCA (8.46%) (Kou *et al.*, 2011a) and MRA with porosity values between 9.13 and 14.86% (Etxeberria and Vazquez, 2010; Gómez-Soberón, 2002; González-Fonteboa and Martínez-Abella, 2008; Kou *et al.*, 2011b; Poon *et al.*, 2007). Nonetheless, Katz (2003) states that the porosity of RA rises significantly when the amount of cement paste is increased and aggregate size is smaller.

11.3.2 Mechanical behaviour of RAs

The mechanical properties of the original materials have a significant impact on the mechanical performance of concrete made with RA (Ajdukiewicz and Kliszczewicz, 2002). Table 11.4 shows the mechanical requirements specified in

Table 11.4 Mechanical specifications for RA in standards and guidelines

Scope	Standard/ Guidelines	Los Angeles abrasion coefficient	10% fine value TFV	Soundness
Australia	CSIRO			●
Belgium	PTV-406	●■✦		
Brazil	NBR 15116			
China	DG/TJ07/008			●
Denmark	DS 2426			
Europe	EN 12620	●		
Germany	DIN 4226-100	●■✦		
Hong Kong	WBTC 12		●	
Italy	NTC	●✦		●✦
Japan	JIS A 5021/5022/5023			
Korea	KS F2573	●		●
Netherlands	NEN 5905	●		
	CUR			
Norway	NB 26			
Portugal	LNEC E 471	●		
United Kingdom	BRE Digest 433			
	BS 8500-2			
	RILEM			
Spain	EHE-08	●		●
Switzerland	SIA 2030			
	OT 70085			

Notes: ● RCA; ■ RMA; ✦ MRA.

the standards and guidelines, depending on RA type. As can be observed, the mechanical performance of RA is mainly defined in terms of its Los Angeles abrasion coefficient, 10% fine value (TFV) and soundness. These tests are only performed on the coarse fraction of the aggregate, and the results are extrapolated to the fine fraction.

The mechanical specification that is most frequently included in the standards is the Los Angeles abrasion coefficient. This coefficient is included in most of the European standards and is the Korean norm. To a lesser extent, soundness also appears in the norms. However, in all cases, it is required for RCA and in some cases, for ceramic and mixed RA.

The Los Angeles abrasion coefficient

The Los Angeles abrasion coefficient is the basic parameter used to measure the resistance of aggregate to fragmentation during handling. Despite its contribution to the mechanical strength of concrete, relatively few standards recommend it for RCA (EHE-08, 2008; prEN 12620, 2010; KS F2573, 2011; LNEC E 471, 2009; NEN 5905, 2010; PTV-406, 2003). Only the Belgian and German standards (DIN 4226-100; PTV-406, 2003) require it for RMA and MRA and the Italian standard (NTC, 2008) requires it for MRA.

Because of the adhered mortar, RA has a higher Los Angeles coefficient value than NA (Domingo Cabo *et al.*, 2009). This value ranges from 29 to 53 for all-in-one RA (Padmini *et al.*, 2009; WRAP, 2007) and from 17.4 to 44 (Yoon *et al.*, 2007; Mas *et al.*, 2011) for coarse RA. There was also a high correlation between the ceramic material content and gypsum content. RMA has a low resistance to fragmentation, which means that it has a lower Los Angeles coefficient (IHOBE, 2011).

The 10% fine value (TFV)

The 10% fine value (TFV) is only included in the Hong Kong Standard (WBTC 12, 2002) and is limited to RCA. The TFV measures aggregate resistance to crushing, which is applicable to both weak and strong aggregate to a minimum value of 100 kN. This parameter ranges from 61.36 to 189.38 kN (Tam and Le, 2007). The British guide WRAP (2007) points to a possible link between a high TFV and the RA content in concrete.

Soundness or mass loss

Soundness determines the mass loss of the aggregate through its resistance to disintegration by weathering and, in particular, freeze-thaw cycles. In the standards where it is mentioned, soundness is only recommended for RCA (CSIRO, 1998; DG/TJ07/008, 2007; EHE-08, 2008; NTC, 2008). Nevertheless, this parameter has not been studied in depth since RA is considered to be less resistant to freezing because of its marked water absorption (Barra de Oliveira and Vazquez, 1996). Furthermore, the available research data are contradictory. Whereas Gokce *et al*. (2004) obtained results well over the limit specified for coarse RA (18.4–48.3), attributable to the adhered mortar, Marinkovic *et al*. (2010) obtained positive results (1.2–1.8).

11.3.3 Chemical suitability of RAs

Table 11.5 shows the chemical requirements for RA in the various standards and guidelines. As can be observed, the most frequently included ones pertain to the chloride and sulfate content of the aggregate. The reason for this is that these chemicals can potentially lead to the corrosion and deterioration of hardened concrete. Also mentioned are the presence of substances, such as clay lumps, soft particles and lightweight particles, which can prove harmful to the setting and hardening of concrete. Finally, the presence of organic matter is also mentioned.

Chemical requirements are most frequently specified for RCA. Surprisingly, the Danish, Australian, Japanese and Korean standards do not limit these substances. They only make recommendations concerning chlorides and organic material, without any reference to sulfur compounds, which can be so detrimental to concrete.

Sulfur compounds

Sulfur compounds can cause cement expansive reactions, which significantly reduce the durability of concrete. RA can contain dangerously large amounts of these compounds because of the gypsum used in construction work. In this regard, IHOBE (2011) confirmed the high solubility of sulfates in RA by performing a thermogravimetric study that detected gypsum, ettringite and portlandite. The results obtained also reflect the close correlation between the contents of granular gypsum and sulfate (in whatever form) in RA. To a lesser extent, this correlation also exists with the ceramic content, though the WRAP (2007) disagrees.

The standards and guidelines limit the total sulfur compounds in RCA (CUR, 1984, 1986, 1994; EHE-08, 2008; NEN 5905, 2010). Acid-soluble sulfates are those most often mentioned for RCA (BS 8500-2, 2006; CUR, 1984, 1986, 1994; DIN 4226-100, 2002; DG/TJ07/008, 2007; EHE-08, 2008). In contrast, specifications for water-soluble sulfates are not included as frequently (NBR 15,116, 2005; RILEM, 1994; WBTC 12, 2002).

Research on sulfur compounds in RA shows that their presence is variable. Values for total compounds ranged from 0.003% (Tam and Le, 2007) to 6.0% (Jimenez *et al.*, 2011) in MRA. For acid-soluble sulfates, the values were between 0.00% (WRAP, 2007) and 6.98% (Mas *et al.*, 2011), whereas water-soluble sulfate values ranged from 0.00% (Calvo Pérez *et al.*, 2002) to 3.93% (Barbudo *et al.*,

Table 11.5 Chemical requirements for RA in standards and guidelines

Scope	Standard/ Guidelines	Water-soluble sulfates	Acid-soluble sulfates	Total sulfur compounds	Acid-soluble chloride	Water-soluble chloride	Total chloride	Lightweight particles	Clay lumps	Organic matter
Australia	CSIRO						✦			●✦
Belgium	PTV-406	■✦	●	●■✦	●■✦	■✦		●■		●■✦
Brazil	NBR 15116	●		✦		●✦		✦	●✦	●
China	DG/TJ07/008		●		●					●
Denmark	DS 2426						✦			
Europe	EN 12620	●	●	●						
Germany	DIN 4226-100	■✦	●		●✦	■✦	■✦			
Hong Kong	WBTC 12	●				●		●		
Italy	NTC	●	●✦	●✦	✦					
Japan	JIS A 5021/5022/5023					●				●
Korea	KS F2573								●	●
Netherlands	NEN 5905			●	●					●
	CUR	■	●		●■			●		●■
Norway	NB 26						✦	●✦		●
Portugal	LNEC E 471	●	●✦	●✦	●✦	✦	✦	●✦		●
Spain	EHE-08		●	●		●	●	●	●	
Switzerland	SIA 2030	●	●	✦	✦		●✦			
	OT 70085		●✦		●		✦			
United Kingdom	BRE Digest 433		●	■		■				
	BS 8500-2	■	●■	✦				●■		
	RILEM	●	■			■		●■		●■

Notes: ● RCA; ■ RMA; ✦ MRA.

2012). The highest values corresponded to RA that had been in contact with gypsum, which is highly soluble in alkaline media (IHOBE, 2011).

Chlorides

In the presence of moisture, the chloride in aggregate can corrode the steel reinforcement in concrete. Unlike sulfates, the presence of chlorides in RA is not related to aggregate type, but rather to factors such as the use of certain additives and exposure to marine environments or to freezing with deicing salts (Sánchez de Juan and Alaejos, 2006).

With the exception of the Chinese standard (DG/TJ07/008, 2007), most of the regulations and guidelines set an overly restrictive limit on acid-soluble chloride in RCA (CUR, 1984, 1986 and 1994; DIN 4226-100, 2002; OT 70085, 2006; WBTC 12, 2002). With regards to RMA and MRA, the acid-soluble chloride content has basically the same limits (CUR, 1984, 1986 and 1994; DIN 4226-100, 2002), though the German and Brazilian standards (DIN 4226-100, 2002; NBR 15116, 2005) permit a higher content in RMA. However, the EHE-08 and SIA 2030 are the sole standards that mention total chlorides for RCA. In fact, in Annex 15 of EHE-08, it recommends the total chloride test for RA since, in some circumstances, combinations of certain chlorides can be reactive and attack the steel reinforcement in concrete.

According to recent research, the acid-soluble chloride content of RA was found to range from 0.00 to 0.08% for all-in-one RCA and MRA (WRAP 2007). Water-soluble sulfates reached values between 0.00% (WRAP, 2007) and 0.13% in coarse MRA (Mas *et al.*, 2011). Finally, total chloride had values ranging from 0.00 to 0.17% in fine RMA (Müeller, 2004).

Damaging substances

Clay lumps, soft particles and lightweight particles are all regarded as damaging substances that can be found in certain types of aggregate. Their presence alters the setting of concrete and decisively affects its strength and durability. Specifications only appear in research conducted in Spain, since the former Spanish EHE Concrete Code (EHE, 1998) included limit values for these substances.

The Brazilian standard (NBR 15116, 2005) limits clay lumps in RCA and MRA, whereas the current Spanish Code (EHE-08, 2008) and the Korean Standard (KS F2573) limits them in RCA. Values for clay lumps in coarse RMA ranged from 0.00% (González-Fonteboa and Martínez-Abella, 2005) to 0.22% (Mas *et al.*, 2011). Limits for soft particles only appear in the former Spanish Code (EHE 1998). In the one research study available on this topic (González-Fonteboa and Martínez-Abella, 2005), the soft particle content was found to be as high as 20.36%. Lightweight particles in RCA (BS 8500-2, 2006; CUR, 1984, 1986 and 1994; EHE-08, 2008; LNEC E 471, 2009; NB 26, 2003; PTV 406, 2003; RILEM, 1994; WBTC

12, 2002), RMA (BS 8500-2, 2006; PTV 406, 2003; RILEM, 1994) and MRA (LNEC E 471, 2009; NB 26, 2003; NBR 15116, 2005) ranged from 0.00% (Jiménez *et al.*, 2011) to 5.85% in coarse MRA (Alaejos and Sánchez de Juan, 2004).

Organic matter

The organic matter in aggregate can slow the setting of cement, even to the point of paralyzing the process completely. Consequently, many standards and guidelines limit organic matter for concrete made from RCA (CUR, 1984, 1986 and 1994; NBR 15,116, 2005), RMA (CUR, 1984, 1986, 1994; RILEM, 1994) and MRA (CSIRO, 1998; NBR 15116, 2005).

Given that there is little quantitative data available with results between 0.15 and 0.95% (Barbudo *et al.*, 2012), it is impossible to evaluate RA quality, based on this parameter. However, in Martín-Morales *et al.* (2011), RA compliance with this parameter was qualitatively analyzed, based on colour as compared to a reference substance (EN 1744-1, 2010). In contrast, IHOBE (2011) showed that there was a low correlation between organic matter and RA quality, which confirmed that this test should not be used to evaluate aggregate.

11.3.4 Geometric properties of RAs

Table 11.6 shows the geometric requirements in the standards and guidelines according to RA type. As can be observed, compliance with specifications of particle size, shape and distribution is crucial to assure high-quality concrete. According to Table 11.6, limits are most often specified for fines content and then for flakiness index. In contrast, the maximum aggregate size and crushing value are seldom mentioned. Finally, various standards do not include any geometric requirements at all for RA (BRE Digest 433, 1998; CUR, 1984, 1986, 1994; DS 2426, 2009; NB 26, 2003; OT 70085, 2006; SIA 2030, 2010).

Aggregate size

Aggregate size should be as large as possible, since the larger the grain size, the greater the strength of the aggregate. This evidently enhances the mechanical strength of concrete. In this respect, only the Korean and British Standards (BS 8500-2, 2006; KS F2573, 2011) limit maximum aggregate size for RCA and the BS 8500-2 (2006) in the case of RMA. However, for the selection of the maximum particle size of the aggregate, it is necessary to take into account the sieve effect produced by the reinforcement and formwork.

Recycled sand content

The standards and guidelines limit the recycled sand content, because it can reduce the compressive strength of concrete (Padmini *et al.*, 2009; Sim and Park,

Table 11.6 Geometric specifications for RA in standards and guidelines

Scope	Standard/ Guidelines	Maximum size of the aggregate	Recycled sand content	Shape index	Flakiness index	Crushing value	Sand equivalent index	Fines content	Shell content
Australia	CSIRO			● ✦		● ✦			
Belgium	PTV-406				●■✦			●■	●■✦
Brazil	NBR 15116							● ✦	
China	DG/TJ07/008		●		●	●			
Denmark	DS 2426								
Europe	prEN 12620			●	●		●	●	
Germany	DIN 4226-100							●■✦	
Hong Kong	WBTC 12		●		●			●	
Italy	NTC		✦	● ✦	●✦		●	● ✦	
Japan	JIS A 5021/5022/5023							●	
Korea	KS F2573	●						●	
Netherlands	NEN 5905								●
	CUR								
Norway	NB 26								
Portugal	LNEC E 471		✦		●			● ✦	✦
Spain	EHE-08		●		●		●	●	
Switzerland	SIA 2030								
	OT 70085								
United Kingdom	BRE Digest 433								
	BS 8500-2	●■						●■	
	RILEM		●■					●■	

Notes: ● RCA; ■ RMA; ✦ MRA.

2011). Most regulations do not permit the use of fine RCA. However, in certain standards and guidelines, recycled sand is allowed for RCA (Annex 15 of EHE-08, 2008; DG/TJ07/008, 2007; RILEM, 1994; WBCT 12, 2002), RMA (RILEM, 1994) and MRA (LNEC E 471, 2009; NTC, 2008).

A recycled fine sand fraction replacement by natural sand invariably improves recycled concrete (Ajdukiewicz and Kliszczewicz, 2002). According to Corinaldesi and Moriconi (2009) and Evangelista and de Brito (2007), 30% is the maximum replacement percentage of natural sand by recycled sand that will not jeopardize its mechanical properties. These authors base their assertion on Katz (2003), who claims that the optimal compressive strength of concrete is directly related to a high fine RA replacement. The reason for this lies in the levels of hydrated and non-hydrated cement, which can be as high as 25% of the mix weight. In contrast, as a way of increasing concrete strength, Becerra-Cabral *et al*. (2010) propose the use of fine recycled ceramic material because of its pozzolanic capacity (Khatib, 2005). This occurs because of the high content of portlandite that is fixed by the ceramic fines in the initial days of reaction (IHOBE, 2011).

Therefore, if the standards and guidelines are strictly complied with, there is no fine or all-in-one RA suitable for manufacturing concrete.

Shape index

As its name implies, the shape index is a method used to measure the shape of coarse aggregate, and indicates how rounded the particles are. The standards studied, including this index as a quality parameter for RA, are the CSIRO, prEN 12620 and NTC. Research results show shape index values ranging from 0.07% for fine RMA (Mas *et al.*, 2011) to 0.47% for coarse RMA (Gómez-Soberón, 2002). Nevertheless, RA particle shape is more irregular than NA and has a coarser surface (Padmini *et al.*, 2009).

Flakiness index

The flakiness index is also used to measure the shape of coarse aggregate. However, unlike the shape index, it determines the quantity of aggregate particles that are elongated instead of cubic. The standards and guidelines that include this index specify very different limit values. Based on the little data available, RMA and MRA generally seem to have a higher flakiness index than RCA. A higher flakiness index seems to be characteristic of aggregate with a small particle size (Gómez-Soberón, 2002; WRAP, 2007). The percentage values were found to range from 5.50% for RCA (WRAP, 2007) to 29.52% for MRA (Tam and Le, 2007). However, the flakiness index is not only dependent on the type of RA, but is also related to the crushing method of the CDW plant (Padmini *et al.*, 2009).

Crushing value

The crushing value quantifies the percentage of crushed particles and is seldom mentioned in standards and guidelines. Only the CSIRO restricts concrete and mixed RA in regard to crushed particles, although the DG/TJ07/008 (2007) establishes limit values for RCA. There is also very little research on this parameter and the few studies available show extremely heterogeneous values, such as 1.73% for coarse MRA (Gokce *et al.*, 2004) and 32% for all-in-one RCA (Padmini *et al.*, 2009). This seems to indicate that aggregate of larger particle size has a lower crushing value.

Fines content

The fines content is evidently more limited in the coarse aggregate fraction. This not only applies to its quantity but also to its quality. From a granulometric perspective, a good-quality fines content (i.e. one that is not clayey) enhances the workability and cohesion of fresh concrete and improves the impermeability and durability of hardened concrete. Moreover, there is no need to add more water or cement.

Practically all standards and guidelines limit the fines content in RA. According to research results, values range from 0.2% (González-Fonteboa and Martínez-Abella, 2005) to 1.17% (Martín-Morales *et al.*, 2011) for all-in-one RA; from 0.1% (González-Fonteboa and Martínez-Abella, 2005) to 1.14% (Alaejos and Sánchez de Juan, 2004) for coarse RA; and from 0.5% (Corinaldesi and Moriconi, 2009) to 46% (Miranda and Selmo, 2006) for fine RA.

Sand equivalent index

The sand equivalent index measures the quality of the fines content in fine aggregate (i.e. its degree of clayeyness). Only the Spanish concrete code (EHE-08, 2008) states that aggregate should not have a sand equivalent index below 70 and 75, depending on environmental exposure. The prEN 12620 and NTC also contemplate this test. In fact, this index has been the focus of very few studies. Based on available research on the fines fraction (0/2 mm), RA was found to have a sand equivalent index ranging from 64.75 (González-Fonteboa and Martínez-Abella, 2005) to 93.6 (Gómez-Soberón, 2002).

11.4 Guidelines for measuring quality parameters of RAs

11.4.1 Physical characteristics of RAs

Table 11.7 shows the requirements in standards and guidelines regarding the physical properties of the RA described in the previous section.

Table 11.7 Requirements for physical properties in standards and guidelines

Property		RA	Limits	Most restrictive standard	Least restrictive standard
Density (kg/m^3)	Oven-dry density	RCA	≥1500–2500 CA	JIS A 5021	EN 12620; NTC
			≥2200 FA	JIS A 5022	JIS A 5022
		RMA	≥1500–2000	CUR	RILEM (Type I)
		MRA	≥1500–2200	DS 2426	DIN 4226-100 (Type 4); NB 26 (Type 1)
	Surface-dry density	RCA	≥2100	CSIRO Class 1A; NB 26 Type 2	CSIRO Class 1A; NB 26 Type 2
		MRA	≥1800	CSIRO Class 1B NB 26 Type 1	CSIRO Class 1B; NB 26 Type 1
	Bulk density	RCA	≥1200–2200	LNEC E 471	CSIRO Class 1A
		MRA	≥1000–2000	LNEC E 471	CSIRO Class 1A
	Specific gravity	RCA	≥2440–2500 CA	KS F2573	CSIRO Class 1A
			≥2200 FA	KS F2573	
Water absorption (%)		RCA	≤3–10 CA	KS F2573; JIS A 5021	DIN 4226-100 Type 1; DG/TJ07/008 Type II; WBTC 12; RILEM Type II; NB 26 Type 2
			≤3–13 FA	JIS A 5021	JIS A 5023
		RMA	≤9–20	PTV-406	DIN 4226-100 Type 3; RILEM Type I
		MRA	≤8–20	CSIRO Class 1B	NB 26 Type 1

Density

Density is an important quality parameter for RA, and thus it is included in most standards and guidelines. Table 11.7 shows the requirements regarding the physical properties of the aggregate. As can be observed, recommended values for oven-dry density, regardless of RA type, are 1500 to 2500. However, most of these values are higher than 2000 kg/m^3. In all likelihood, those standards that specify a minimum value (1500 kg/m^3) are following harmonized standard prEN 12620. This value is also included in other European standards (NTC, 2008; DIN 4226-100, 2002), which is a reflection of the incipient harmonization process. Only the JIS A 5022 establishes a minimum value of 2200 kg/m^3 for fine aggregate.

Surface-dry density is regarded as somewhat less crucial, and is contemplated only in the Australian (CSIRO, 1998) and Norwegian (NB 26, 2003) norms. Moreover, they only mention RCA and MRA. In both standards, values are more than 2100 kg/m^3 for RCA and ≥1800 kg/m^3 for MRA.

Similarly to surface-dry density, bulk density is also only mentioned in two standards (CSIRO, 1998; LNEC E 471, 2009), although they do not agree on the recommended limit values. For RCA, the CSIRO recommends a value of 1200 kg/m^3, whereas the value in the LNEC E 471 is 2200 kg/m^3. The recommendations in the two standards also significantly diverge with regard to MRA, for which the CSIRO recommends a value of 1000 kg/m^3 and the LNEC E 471, a value of 2000 kg/m^3.

Finally, specific gravity for coarse RCA is limited in the CSIRO (Class 1) and the KS F2573 with a minimum value of 2500 kg/m^3. The KS F2573 also establishes a minimum specific gravity value of 2200 kg/m^3 for fine RCA.

Water absorption

The absorption values for coarse RCA in Table 11.7 fall in two groups:

1. a more restrictive group that sets limits at 5 to 7% (DG/TJ07/008, 2007; EHE-08, 2008; JIS A 5022, 2006; JIS A 5023, 2007; NEN 5905, 2010; NBR 15116, 2005); and
2. a less restrictive group that sets limits at 9 to 10%. It is striking that the value established by Asian countries is 3%. Regarding RCA fines, the NBR 15116 and JIS A 5023 include values of 13%.

However, the JIS A norm recommends much lower values (3–7%), depending on the use of the concrete.

In reference to RMA, most of the standards agree on a less restrictive value (≤20%). Nevertheless, the Belgium standard specifies a value of 9%. The recommendations for MRA absorption show percentages ranging from 8% (CSIRO, 1998) to 20% (NB 26, 2003), with an intermediate percentage of 15% in the case of the DIN 4226-100. In this sense, the Brazilian standard is the only one that establishes values for fine MRA with a maximum value of 17%.

11.4.2 Mechanical behaviour of RAs

Table 11.8 lists the requirements in the standards and guidelines with regard to the mechanical properties of RA.

Table 11.8 Requirements for mechanical properties in standards and guidelines

Property	RA	Limits	Most restrictive standard	Least restrictive standard
Los Angeles abrasion coefficient	RCA	≤40–50	EHE-08; NEN 5905	LNEC E 471
10% fine value (kN)	RCA	>100	WBTC 12	
Soundness (%)	RCA	≤9–18 CA	CSIRO	DG/TJ07/008; EHE-08
		≤10 FA	KS F2573	

Resistance to fragmentation

Despite the fact that resistance to fragmentation is a crucial parameter for concrete, it is only contemplated in the EHE-08 and NEN 5905 with a value of 40 to 50 for RCA (Table 11.8). Nonetheless, as for other properties, the prEN 12620 states that the manufacturer should declare the category of the aggregate, depending on the value obtained in each test, when the standard does not include a limit value. Accordingly, other European standards with a view to harmonization with the prEN 12620 also establish this category assignment. In the specific case of the Los Angeles test, the manufacturer should declare a category for RCA (DIN 4226-100, 2002; prEN 12620, 2010; LNEC E 471, 2009; NTC, 2008; PTV 406, 2003) in the range of 10 to 60.

10% fine value

The TFV is a mechanical parameter that is only specified in the WBTC 12. As can be observed in Table 11.8, the RA should have a value greater than 100 kN.

Soundness or mass loss

According to Table 11.8, the mass loss of RA, when it is subject to the crystallization of salts, is 9 or 18% for coarse RCA. The only exception is the KS F2573, which specifies a limit of 12%. This same standard limits mass loss to 10% for fine RCA. Furthermore, as in the Los Angeles test, the manufacturer should declare the category of the aggregate when the soundness is in the range of 1 to 50% for RCA (prEN 12620, 2010; NTC, 2008) and for MRA (NTC, 2008).

11.4.3 Chemical suitability of RAs

Table 11.9 lists the requirements in the standards and guidelines related to the chemical properties of RA. The following section provides a detailed analysis of

Table 11.9 Requirements for chemical properties in standards and guidelines

Property	RA	Limits	Most restrictive standard	Least restrictive standard
Water-soluble sulfate (%)	RCA	≤0.2–1	LNEC E 471	RILEM (Type II); WBTC 12; NBR 15116
	RMA	<1	RILEM (Type I)	
	MRA	≤0.2–1	LNEC E 471	NBR 15116
Acid-soluble sulfate (%)	RCA	≤0.8–1	DIN 4226-100; EHE-08; LNEC E 471	BS 8500-2; BRE Digest 433; CUR; DG/TJ07/008 Type I; OT 70085
	RMA	≤0.8–1	DIN 4226-100	BS 8500-2; CUR
	MRA	≤0.8–1	DIN 4226-100; LNEC E 471	OT 70085
Total sulfur (%)	RCA	≤1	EHE 08; prEN 12620; NEN 5905; NTC; LNEC E 471	
	RMA	≤1	BRE Digest 433	
	MRA	≤1	NTC; LNEC E471; BRE Digest 433; SIA 2030	
Water-soluble chloride (%)	RCA	≤0.03–1	EHE-08	NBR 15116 Class A
Acid-soluble chloride (%)	RCA	≤0.03–0.25	OT 70085	DG/TJ07/008 Type I
	RMA	≤0.04–0.06	DIN 4226-100 Type 3	PTV-406
	MRA	≤0.04–1	DIN 4226-100 Type 2	NBR 15116 Class A
Total chloride (%)	RCA	≤0.03–0.15	EHE-08; SIA 2030	EHE-08
Lightweight particles (%)	RCA	≤0.1–1	CUR; NB 26 Type 2	EHE 08; LNEC E 471
	RMA	≤1	PTV-406; RILEM Type I; BS 8500-2	
	MRA	≤1	LNEC E 471	
Clay lumps (%)	RCA	≤0.2–2	KS F2573	NBR 15116
	MRA	≤2	NBR 15116	
Organic matter (%)	RCA	≤0.10–2	CUR; NEN 5905	NBR 15116 Class A
	RMA	≤0.5–1	PTV-406	RILEM Type I; CUR
	MRA	≤0.15–2	CSIRO Class 1B	LNEC E471; NBR 15116 Class A

each of them. As can be observed, there is no difference in the limit values established for different aggregate types.

Sulfur compounds

In order to analytically obtain sulfur compounds (e.g. water-soluble sulfate, acid-soluble sulfate and total sulfate compounds), the standards establish different tests, the values for which appear in Table 11.9. Water-soluble sulfates are limited in all of the standards and for all aggregate types to a maximum of 1%. The exception is the LNEC E 471, which gives a limit value of 0.2% for RCA and MRA. Furthermore, if the standard does not explicitly include a limit, the manufacturer should declare that water-soluble sulfate is within the range of 0.2 to 1.3% for RCA (prEN 12620, 2010; NTC, 2008) and for MRA (NTC, 2008).

Acid-soluble sulfate compounds should have values of 0.8 or 1%. Where the manufacturer is obliged to classify the RA with regard to this parameter, they should do so within the range of 0.2 to 1% for RCA (prEN 12620, 2010; NTC, 2008; PTV-406, 2003), RMA (PTV-406, 2003) and MRA (NTC, 2008; PTV-406, 2003).

Finally, for total sulfate compounds, regardless of the aggregate source, the standards unanimously establish a maximum value of 1%. In addition, it is necessary to specify RA classification in the PTV-406 within the range of 1 to 2%.

Chlorides

As for other compounds, chlorides are divided into water-soluble, acid-soluble and total chlorides. According to Table 11.9, the maximum value of water-soluble chloride in RCA generally ranges from 0.04% (JIS A 5021, 2005) to 0.05% (EHE-08, 2008; WBTC 12, 2002). It should be highlighted that the EHE-08 limits chlorides, depending on whether the concrete is mass concrete, reinforced concrete or pre-stressed concrete, to values of 0.15, 0.05 and 0.03%, respectively. This contrasts with the NBR 15116, which establishes a limit of up to 1% for the non-structural use of concrete. This limit is significantly less restrictive than the value of 0.15% in the Spanish standard.

With reference to acid-soluble chloride, for all RA, the standards establish limit values of 0.03 to 0.06%. However, for RCA, the DG/TJ07/008 recommends a more permissive value of 0.25%. Furthermore, for MRA, the standards recommend values that should not exceed 0.15% (DIN 4226-100, 2002) and 1% (NBR 15116, 2005). According to the LNEC E 471, the manufacturer should declare the category of the RA for RCA as well as for MRA.

Total chloride is limited in two standards (EHE-08, 2008; SIA 2030, 2010), although only for RCA. In the case of the EHE-08, the limit values are the same as for water-soluble chloride. However, the SIA 2030 is more restrictive in the case of mass concrete (0.12%) as well as in reinforced concrete (0.03%), which coincides with the value for pre-stressed concrete in the Spanish standard.

Damaging substances

Lightweight particles, soft particles and clay lumps are regarded as damaging substances. As shown in Table 11.9, only the soft particles are not included in any of the standards. For lightweight particles in RCA, there are three groups of standards with a wide range of values:

1. a first group with a less restrictive value of 1%;
2. a second group with an intermediate value of 0.5% (BS 8500-2, 2006; EHE-08, 2008; RILEM, 1994; WBTC 12, 2002); and
3. a third group with a more restrictive value of 0.1%.

In contrast, RMA and MRA are assigned a sole limit value of 1%.

Clay lumps are only mentioned in three of the standards and with very different values. For coarse RCA, values for this substance are limited to 0.2% (KS F2573, 2011), 0.6% (EHE-08, 2008) and 2% (NBR 15116, 2005). As can be observed, the Brazilian norm is extremely permissive. In fact, this same standard also limits clay lumps in MRA to 2%.

Organic matter

The most characteristic limit values for RCA content in organic matter in the standards consulted are 0.5% (DG/TJ07/008, 2007; JIS A 5021, 2005; RILEM, 1994) and 0.1%. The NBR 15116 has the least restrictive limit value of 2%. Similar limit values are established for RMA and MRA (Table 11.9).

11.4.4 Geometric properties of RAs

Table 11.10 lists the requirements in standards and guidelines for the geometric properties of the RA.

Maximum aggregate size

Maximum grain size is only contemplated in the KS F2573 and BS 8500-2, which specify the following similar values for RCA:

- 25/20 mm in the Korean standard; and
- 20 mm in the British standard.

In addition, the British standard (BS 8500-2, 2006) establishes the same value for RMA (Table 11.10).

Recycled sand content

With regard to recycled sand content, the standards that permit the use of recycled sand only sanction the use of sand from concrete. As shown in Table 11.10, the recycled sand content should generally be less than 5%. The RILEM recommendations in its Type II application permit the same content of sand from mixed aggregate.

Shape index

The shape index should have a maximum value of 35% for RCA as well as for MRA (Table 11.10). In this sense, for RCA, the European standard (prEN 12620, 2010),

Table 11.10 Requirements for RA geometric properties in standards and guidelines

Property	RA	Limits	Most restrictive standard	Least restrictive standard
Maximum aggregate size (mm)	RCA	≤20–25	BS 8500-2	KS F 2573
	RMA	≤20	BS 8500-2	
Recycled sand content (<4 mm) (%)	RCA	≤5	DG/TJ07/008 Type I; EHE 08; WBTC 12; RILEM Type II	
	RMA	≤5	RILEM Type II	
Shape index (%)	RCA	≤35	CSIRO	
	MRA	≤35	CSIRO	
Flakiness Index (%)	RCA	≤15–50	DG/TJ07/008	LNEC E 471
	MRA	≤50	LNEC E 471	
Crushing value (%)	RCA	≤30	CSIRO; DG/TJ07/008 Type I	
	MRA	≤30	CSIRO	
Sand equivalent index	RCA	>70–75	EHE-08	
Fines content (<0.063 mm) (%)	RCA	≤6–16 FA	EHE-08	
		≤1–10 CA	JIS A 5021; KS F 2573	NBR 15116
	RMA	10 FA	DIN 4226-100	
		3–5 CA	RILEM Type I; BS 8500-2	PTV-406
	MRA	≤10–20 FA	DIN 4226-100 Type 2	NBR 15116 Class A
		≤3–10 CA	LNEC E 741	NBR 15116 Class A
Shell content of coarse aggregates (%)	RCA	<10	PTV-406; NEN 5905	

the DIN 4226-100 and the NTC state that the producer should declare whether the shape index is in the range of 15 to 55%. This should be declared in accordance with the relevant category of the aggregate. The DIN 4226-100 also includes this requirement for RMA and MRA, whereas the NTC includes it for MRA.

Flakiness index

In the case of RCA, the flakiness index has widely different values (15–50%), depending on the standard or guideline. As shown in Table 11.10, the EHE-08 and the LNEC E 471 set a value of 35% and the WBTC 12, a value of 40%. For MRA, only the Portuguese standard has a limit value of 50%. In addition, similar to the shape index, the flakiness index should be declared by the producer in accordance with the relevant category, depending on the application or end use. This is made explicit for RCA (prEN 12620, 2010; NTC, 2008; PTV 406, 2003), for RMA (PTV 406, 2003) and for MRA (NTC, 2008; PTV 406, 2003) when the test is not required and when the flakiness index value is 10 to 50%.

Crushing value

The crushing value is the percentage of crushed particles in the RA. This value only appears in the Chinese norm for Type I RCA (Table 11.10). In this sense, the CSIRO has the same limit value (30%) for RCA and MRA.

Sand equivalent index

The sand equivalent index is only contemplated in the EHE-08 (Table 11.10). This standard states that fine RCA should not have an index lower than 70 to 75, depending on the general class or type of exposure of the concrete produced with this aggregate. As with the other properties, this value should be determined in order to assign the aggregate to the relevant category with an interval of 30 to 65 (prEN 12620, 2010).

Fines content

In all of the standards, the fines content is one of the most important geometrical properties of the aggregate. Requirements generally establish that the fines content of fine RCA should be 6 to 16% (Table 11.10). Approximately half of the norms set a value of 6%, whereas the other half places this value at 16%. An exception is the DIN 4226-100, which gives a maximum value of 10%. In the case of coarse RCA, all of the standards establish maximum values of 1 to 4%, except for the NBR 15116 with a maximum fines content of 10%. With regard to RMA and MRA, all of the standards have fines content values of 3 to 5% for coarse RA and 10% for fine RA. Nonetheless, for MRA, this fines content can have maximum values of 18 to 20% (DIN 4226-100, 2002; NBR 15116, 2005).

Shell content

The shell content in RA is only relevant for coarse aggregate. This test is only recommended in the Belgian standard (PTV-406, 2003) and the Netherlands standard (NEN 5905, 2010). It is probably no accident that the two countries are also geographical neighbours. In both documents, the shell content is limited to a maximum of 10% and only for RCA (Table 11.10).

11.5 Parameters affecting compliance with quality criteria

11.5.1 Physical properties

The density of RA (Table 11.3) is usually slightly lower than that of NA. Nevertheless, it should be relatively easy to comply with the limit values in the standards and guidelines analyzed (Table 11.7). Absorption capacity is one of the most important physical parameters for RA. The absorption capacity of RA is greater than that of NA. RA water absorption values can vary, depending on the type and size of RA (Angulo *et al.*, 2004; Becerral *et al.*, 2010; Miranda and Selmo, 2006; Poon and Chan, 2007; Tam and Le, 2007). The comparison of typical RA absorption capacity values with those in standards and recommendations (Table 11.7) indicate that it might be difficult for RA to comply with even the least restrictive values. The problem is greater in the case of fine RA than for coarse RA (Miranda and Selmo, 2006). An exception is RA from sanitary ceramic and electrical ceramic materials, whose typical absorption capacity is lower than all of the limit values in the standards because of the compactness of such material (Medina *et al.*, 2012; Senthamarai *et al.*, 2011).

Absorption capacity and density values are directly related to the cement mortar attached to aggregate particles. Consequently, typical values could be significantly reduced by improving the on-site selection of the CDW, as well as its processing at the treatment plant.

11.5.2 Mechanical properties

Mechanical properties of RA generally comply with the limit values in standards and guidelines. In the case of the Los Angeles coefficient and the TFV, compliance depends on the RA sample. In the case of soundness or mass loss, RA samples easily fulfill the limit values in the standards.

The typical Los Angeles coefficient values for RA in the references analyzed were found to be 17.4 and 53, whereas the standards and guidelines give limit values lower than 40 to 50 (Table 11.8). This means that certain types of RA (e.g. coarse RA) might have compliance problems. The presence of ceramic waste and cement mortar attached to the aggregate particles gives them a lower resistance to fragmentation (Chini *et al.*, 2001; Hansen and Narud, 1983; Sri Ravindrarajah *et al.*, 1987). Thus, RA selection and treatment processes need to be improved in order to improve fragmentation resistance. In the case of TFV, typical values for RA were found to be 61.36 to 189.38 kN (Tam and Le, 2007), although the limit values in standards and recommendations are all higher than 100 kN.

11.5.3 Chemical properties

A comparison of typical RA chemical composition values with those in the standards and guidelines (Table 11.9) reflect that the chemical properties of RA are one of its weakest aspects, because of its high content in sulfates and chlorine. Sulfate-based products, such as gypsum, are common contaminants in CDW (Tam *et al.*, 2008). Thus, sulfate-resistant cement should be seriously considered in situations where gypsum contamination is suspected. However, since a high percentage of the small particle-size fraction goes through the crushing process, the average sulfate content value in RA is 0 to 6%, depending on the characteristics and the processing of the CDW. These values exceed the limits in the standards and guidelines (Table 11.9).

The maximum chlorine content in RA ranges from 0.08 to 0.13%. Although some values are only slightly higher than the limit values for certain types of RA, they can still lead to the corrosion of the steel reinforcement embedded in concrete. This has an extremely negative impact on concrete durability (Tam *et al.*, 2008) and thus, it is crucial to control the presence of chlorine in concrete.

When the demolition process is selective and when there is a manual selection of waste before the crushing process, this facilitates the removal of gypsum, large impurities, clay, organic matter and lightweight particles. The quality of the RA can thus be enhanced so that the RA can more easily comply with the limit values for chemical compounds in the standards and guidelines. However, the presence of chlorine compounds in RA is directly related to its exposure to a marine environment as well as to the use of certain additives. As a result, chlorine is more difficult to control or reduce in RA. Certain authors suggest that this content can be decreased by immersing the aggregate in water (Debieb *et al.*, 2009).

11.5.4 Geometric properties

The geometric properties that are most severely limited are the presence of recycling sand and fines. All of them are strictly limited in standards and recommendations (Table 11.10) because of their negative effects on concrete quality. Properties related to aggregate shape, such as the shape index, flakiness index and crushing value, depend on the crushing and size selection equipment. Experience at recycling plants reflects that concrete tends to break into cube-shaped particles rather than elongated ones. Generally speaking, these characteristics should be controlled with suitable CDW treatment procedures to obtain RA in compliance with the limit values in the standards and guidelines.

11.6 Conclusions

The results of quality studies reflect that in comparison to NA, RA needs to improve certain parameters. For this reason, it was necessary to establish reference values. Nevertheless, these values often show significant variation, depending on the standard or guideline. This study shows that the standards and guidelines, elaborated by technical committees, research agencies and standards organizations, are endeavouring to encourage the use of aggregate from CDW in construction. For this reason, they all establish limit values concerning RA composition, properties and end use. The conclusions that can be derived from this study are:

- In certain cases, the limit values in standards and guidelines vary considerably. This means that the possible revalorization of CDW is significantly reduced in the most restrictive norms. Accordingly, what is needed is a set of harmonized regulations for RA, which would homogeneously specify its possible end uses, depending on its application. This would undoubtedly favour its wider use in construction.

- In relation to RA, its absorption as well as its chloride, sulfate and fines contents are the properties most in need of improvement, because of their potentially devastating impact on concrete strength and durability.
- Selective demolition and a more effective treatment of CDW at the plant are crucial, in order to control and improve the properties that lower aggregate quality.
- Finally, mixing RA and NA is a way of improving RA quality when certain of its properties make it unsuitable for a given application or end use.

11.7 Sources of further information and advice

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12
Properties of concrete with recycled aggregates

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Abstract: Concrete is widely used as a construction material in buildings and civil works. In the last few years intensive research activity has been carried out on the application of recycled aggregates (RA) to replace natural aggregates (NA) in concrete mixtures. Many investigations have been conducted using different percentages of coarse RA, studying the mechanical behaviour and durability of recycled concretes. It is possible to replace the coarse fraction of NA by RA, without affecting the final properties of concrete. The properties of recycled concretes manufactured with different types of RA (fine and coarse, concrete and mixed aggregates) are discussed.

Key words: recycled aggregates (RA), recycled concrete aggregates (RCA), recycled mixed aggregates (RMA), natural aggregates (NA), fine and coarse fraction, recycled concrete.

12.1 Introduction

Concrete is widely used as a construction material in buildings and civil infrastructure. It is not considered environmentally friendly because of the use of natural aggregates (NA) in its production. In the last two decades, a growing number of research studies have been conducted to make concrete production more sustainable, mainly focusing on reducing the consumption of natural resources. Significant research has been carried out on the application of recycled aggregates (RA) to replace NA in concrete mixtures (Poon *et al.*, 2002; Topçu and Sengel, 2004; Limbachiya *et al.*, 2000). This is important because approximately 80% of all concrete consists of aggregates.

In 1994, *Specifications for Concrete with Recycled Aggregates* (RILEM Recommendations for demolition and re-use of concrete and masonry) was published, being the first international recommendations for the application of RA in concrete production. In this publication, different categories of recycled coarse aggregates were defined:

- **Type I**: recycled aggregates from processing masonry rubble.
- **Type II**: recycled aggregates from processing concrete residues.
- **Type III**: blend of recycled and natural aggregates.

Even in this early publication, it was specially recommended that RA should come from crushed concrete, known as recycled concrete aggregate (RCA), due to its higher quality. Likewise, it was also advisable to replace only the coarse fraction of NA, so as not to affect the final properties of concrete. In this respect, many investigations were conducted using different percentages of coarse RCA, studying the mechanical behaviour and durability of recycled concretes.

In the last ten years, studies have focused on the application of RA coming from the treatment of debris mixed from concrete wastes and masonry. These recycled mixed aggregates (RMA) contain ceramic particles (crushed clay bricks), and also certain quantities of gypsum, asphalts, contaminants, and so on. Various studies applied these RMA in the manufacture of non-structural concretes (Correia *et al.*, 2006; Agrela *et al.*, 2011; Senthamarai *et al.*, 2005), concluding that these concretes had worse mechanical and durability properties than concretes with RCA. However, these RMA could be applied if their quality (e.g. assessed by means of their water absorption) is properly controlled (Mas *et al.*, 2012).

The most recent research has focused on concretes replacing natural sand by the fine fraction of the RA (Khatib, 2005; Evangelista and de Brito, 2007; Zega and di Maio, 2011). The higher water absorption capacity of these recycled materials, their greater acid soluble sulphates content, and the detrimental effect on workability, cause these sands to be not commonly used in concrete production. In this chapter, an updated review on the properties of recycled concretes manufactured with the different types of RA (fine and coarse, concrete and mixed aggregates) will be presented. Aspects related to their effect on fresh and hardened concrete properties will be examined.

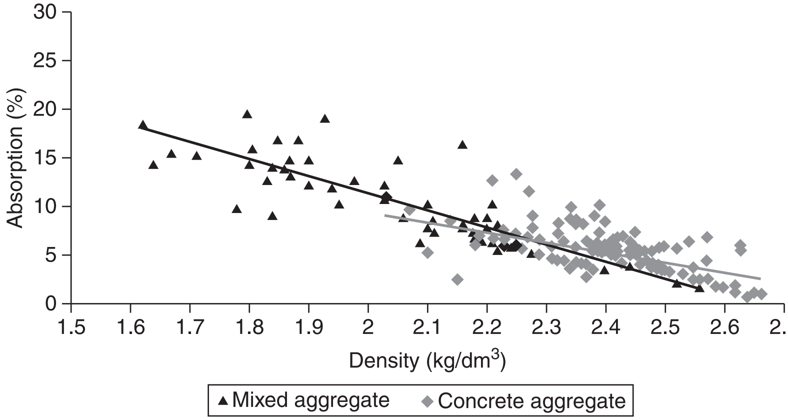
12.2 Properties of fresh concrete using recycled aggregates

The use of RA has an important influence on fresh concrete properties, particularly affecting its workability. The high absorption properties of these aggregates, in addition to their heterogeneity, make it harder to adjust and control the consistency of the concrete.

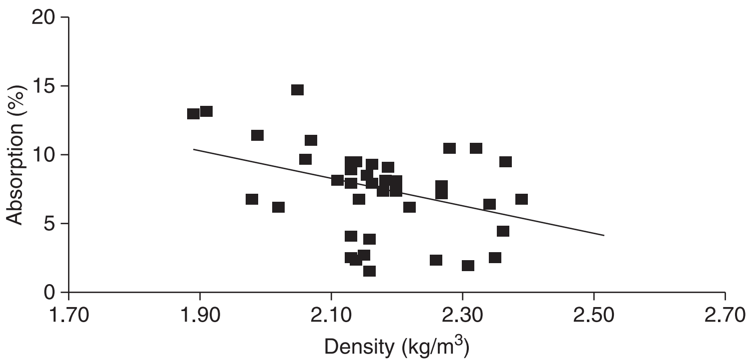
12.2.1 Influence of recycled aggregate (RA) absorption on the water demand of concrete

RA have higher absorption values than NA, for which the results are usually under 3%. The absorption properties of RA mainly depend on their nature, being lower for concrete coarse aggregates (in general under 12%), but in the case of mixed aggregates reaching up to 20% when they include a large quantity of ceramic particles (Fig. 12.1).

However, the fine fraction of RCA is composed of an important amount of porous mortar particles, produced by the crushing process, so they commonly present high absorption values too (Fig. 12.2).



12.1 Correlation between water absorption and density of coarse fraction of RCA and RMA (Geraldes, 2012).



12.2 Correlation between water absorption and density of fine fraction of RCA (Alaejos *et al*, 2010).

This significant and rapid absorption of the RA directly affects the water demand of concrete. The influence is negligible when only limited amounts of coarse concrete RA (20–30%) are used, so in this case the same concrete production process as for NA can be carried out. The rest of the time, extra water demand has to be considered in the production of recycled concrete, in which different alternatives can be applied:

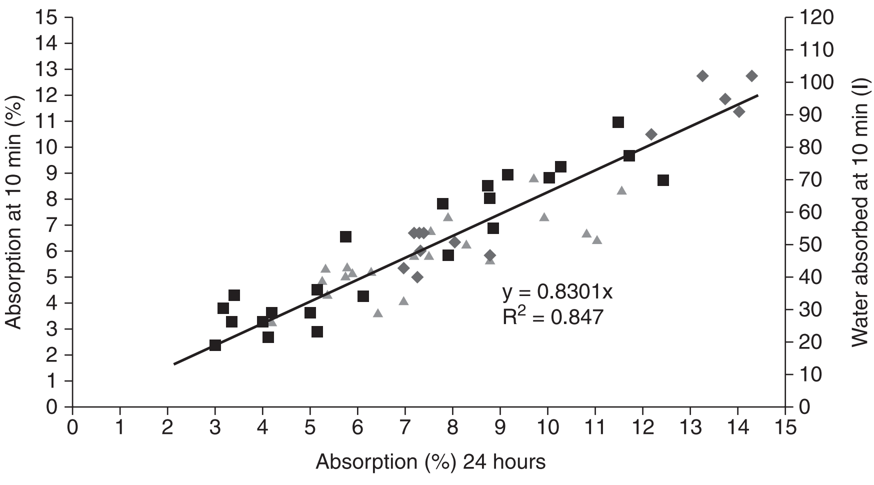
- a direct increase of the mixing water;
- use of chemical admixtures (plasticizers or superplasticizers);
- pre-saturation of the recycled aggregate.

Increase of the mixing water

In order to compensate for the absorption properties of the RA, when it is used in dry conditions, additional water content (w/c) has to be considered in the mix design. As the absorption of these materials takes place rapidly, the necessary amount is usually determined by means of the 10 min absorption rule. In order to obtain the experimental values for coarse RA, the standard test EN 1097 6:2001/A1:2006 can be applied, by immersing the dry aggregate sample for just 10 min in water, removing the surface humidity with an absorbent cloth and then weighing it. However, for fine aggregate, the standard test cannot be applied, as the superficial humidity of the wet sand is extremely high and cannot be as easily removed as it can for coarse aggregates.

The German Standard DIN 4226-100:2002 describes a specific method to obtain the 10 min absorption value for recycled sands. In the test, water is sprayed for 10 min over a dry sample of sand while stirring it. At the end of this period the sample is weighed. The procedure is quite subjective, so the standard points out that the final result is just an indicative value. Reference 16 describes a more precise procedure using a humidity analyser to determine the short period absorption of the recycled sands. The results of 10 min absorption are approximately 83% of the 24 h value for RCA and RMA (both coarse and fine aggregates) (Fig. 12.3).

According to the 10 min absorption values, the second *y*-axis in Fig. 12.3 shows, as an example, the corresponding additional w/c necessary in the mix, calculated for 100% coarse fraction of RA and assuming an average value of 800 kg/m^3 of gravel in the concrete mix. This extra w/c in Fig. 12.3 has been



12.3 Relationship between the 10 min and 24 h absorption for different types of recycled aggregates (Alaejos and Rueda, 2012) and the subsequent extra water content necessary in the mix (second *y*-axis).

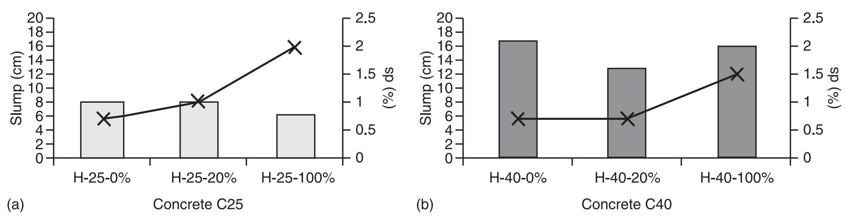
calculated on the basis that the RA are completely dry, although in practice their humidity will be variable. This means taking the risk of adding excess water if their humidity is not precisely determined. This method has an additional problem related to the variation in the water absorption capacity of the RA, due to these recycled materials presenting an inherent heterogeneity, and this means that the extra water is not a fixed value.

Use of chemical admixtures

When the RA are used in dry conditions, and in order not to add additional water, chemical admixtures can be used to compensate for the effect of water absorption and the consequent loss of workability of the fresh concrete. This alternative results in a mix with a lower effective water/cement ratio and subsequently a recycled concrete with improved durability and mechanical properties. The amount of chemical admixtures required to compensate for the effect of the RA increases as the aggregate content increases. If a limited amount of coarse concrete recycled aggregate is used (20%), no additional admixture will be needed, but a total replacement makes necessary a double amount of admixture to maintain the same consistency (Fig. 12.4 shows an example for coarse concrete recycled aggregate with 5.1% absorption).

For coarse RMA, the single use of a chemical admixture may not be enough to compensate for their detrimental effect on the consistency of fresh concrete. Taking into account that conventional superplasticizers allow for a water reduction of up to 30%, and assuming an average mix w/c of 190 l/m^3 in the concrete, this means a maximum water reduction of 57 litres. Again, if we look at the second *y*-axis in Fig. 12.3, where the theoretical 10 min water absorption rate of the aggregate is shown, we can see that the superplasticizer would be effective only for RA with a maximum absorption of 9%.

Due to the high absorption values of recycled fine aggregates, it is usually necessary to use a superplasticizer, in addition to extra w/c. One important aspect to consider is the water demand and subsequently the effect on consistency of the fine aggregates in the recycled sand. Pereira *et al.* (2012) show that the effectiveness of superplasticizers decreases when recycled sands are used, even previously



12.4 (a, b) Superplasticizer content to maintain the same slump in recycled concretes of two different strengths (Alaejos *et al.*, 2010).

compensating for their absorption, it being necessary to increase further the w/c compared to the conventional concretes: from a w/c = 0.45 to 0.49 and from w/c = 0.38 to 0.41 (0–100% recycled sand) to maintain the same slump, when two different SP are used.

Pre-saturation of the RA

For high absorption RA and also for high replacement values, pre-saturation is the easiest way to compensate for their effect on workability. Pre-saturation can be carried out in the Ready Mixed Concrete Plant, by watering the aggregates stockpiles or spraying water over the aggregates while they are being transported towards the mixer on the conveyor belts. Whatever the procedure applied, it is important to allow the excess water to properly drain and evacuate from the aggregate, in order not to increase the water/cement ratio of the mix. However, the pre-saturation procedure cannot be applied to recycled sands, as the particles retain a lot of water on their surfaces when wet, dramatically increasing the water/cement ratio of concrete.

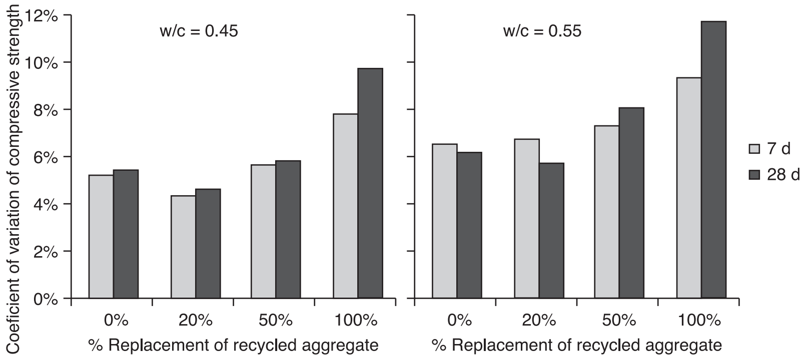
12.2.2 Influence of the heterogeneity of the RA in the properties of concrete

RA are heterogeneous materials as their properties vary depending on the nature and quality of the waste used in their production. As an example, Fig. 12.1 shows how a wide the range of results for absorption and density can be, even for aggregates of the same nature (pure concrete or mixed with ceramic). This heterogeneity is inherent to the RA, but of course can be diminished in recycling plants that apply the correct separation of wastes and proper processing. This dispersion of the aggregate quality can detrimentally influence the uniformity of the properties of concrete. In particular, coefficients of variation of compressive strength for recycled concretes are usually higher than for conventional concretes.

A study carried out on the production of recycled concretes obtained a coefficient of variation (based on ten consecutive batches with the same mix design) that was more than double when 60% of coarse RA was used, compared to 30% of RA (5 and 2% respectively). In the same way, Fig. 12.5 shows, for two different mixes (w/c = 0.45 to w/c = 0.55) (WRAP, 2007), the increase of the coefficient of variation of compressive strength when increasing amounts of coarse RA are used (Ulloa, 2012). In both concretes, a total replacement of natural gravel by RA gives a double value of the coefficient of variation.

12.2.3 Curing conditions

There is little information concerning the effect of different curing conditions on the mechanical properties of recycled concretes. A recent study (Fonseca *et al.*,



12.5 Influence of the percentage of recycled aggregate on the coefficient of variation of compressive strength (Ulloa, 2012).

2011) has compared results on concretes cured under four different conditions (laboratory, outer environment, wet chamber and water immersion) when using coarse concrete RA up to 100%. The final conclusion is that mechanical properties of recycled concretes (compressive strength, modulus of elasticity and splitting tensile strength) are no more affected than for conventional concretes.

12.3 Properties of hardened concrete using recycled aggregates

The mechanical and durability properties of recycled concretes have been extensively studied, mainly using the coarse fraction of RCA (Nixon, 1978; Hansen, 1992; Topçu and Sengel, 2004) and to a lesser extent using the RMA (Yang *et al.*, 2011) or the fine fraction (Zega and di Maio, 2011). The most important properties of hardened recycled concretes analysed in this section are:

- compressive strength;
- tensile splitting and flexural resistance;
- elastic modulus;
- density;
- shrinkage;
- creep.

12.3.1 Compressive strength

The use of coarse recycled aggregate in concrete causes a reduction in compressive strength (Xiao *et al.*, 2012; Sánchez de Juan, 2004), especially as the level of substitution increases and more significantly when RMA are used. This reduction is mainly due to the following causes:

- a worse aggregate quality reflected in a higher porosity, lower Los Angeles coefficient and presence of sulphates in some cases;
- higher water demand for mixing, affecting the w/c ration and subsequently the mechanical properties of concrete;
- presence of weak zones in the concrete, due to the mortar present in the RA (Sánchez de Juan and Alaejos, 2004);
- As the type and RA content are the main factors influencing the final compressive strength, three different cases will be analysed:
  - concretes with RCA replacement between 20 and 50% (low level content);
  - concretes with 100% RCA (high level content);
  - concretes with different contents of RMA.

In all cases, only the use of coarse RA is considered.

Concretes with recycled concrete aggregate (RCA) content between 20 and 50%

As a general state, when limited amounts of coarse RCA are used, the mechanical and durability properties of recycled concretes are hardly affected. However, an additional important aspect to consider is the quality of the RCA used, as higher strengths can be obtained when the aggregates come from crushed debris of a good concrete. In particular, the Spanish Structural Concrete Code (EHE-08) recommends processing debris from structural concretes (fc $\geq$25 MPa) if the RA are going to be used in structural concretes (EHE-08, 2008). Other requirements of RCA are shown in Table 12.1.

Regarding the mix design of recycled concretes, and due to the different densities of natural and RA, it should be taken into account that the replacement of natural aggregate by recycled aggregate must be calculated in volume, although it is usually expressed in weight. Included in Table 12.2 is an example of concrete mixtures, in which RCA are applied with density values of 2.44 gr/dm^3, and 2.65 gr/dm^3 in the NA used. Different substitution levels of NA by RCA are applied.

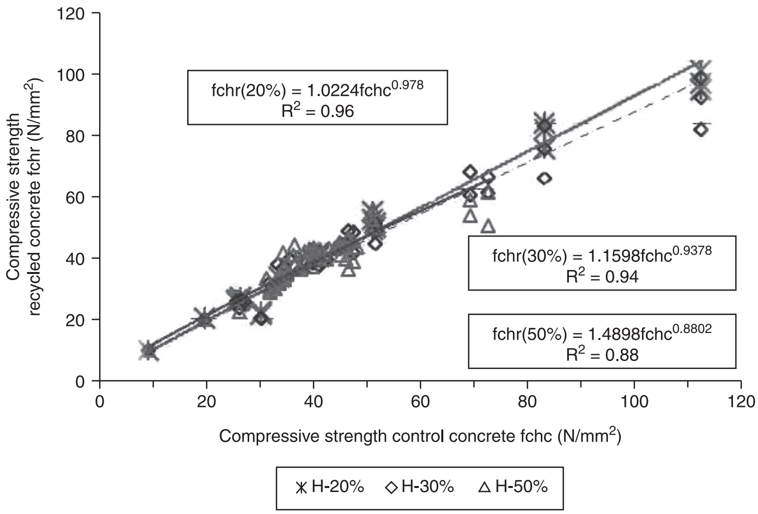
It can also be seen in the previous table that the weight of the different mixes are reduced according to the increasing degree of substitution. This is due to the lower density of recycled aggregate compared to natural.

When comparing the strength of recycled and conventional concretes, it is important to use the same effective w/c ratio for both (Katz, 2003). If the RA are used in dry conditions and their absorption is not compensated for in mix design, they will absorb water during the mixing process, reducing the effective w/c ratio and so improving the strength of the recycled concretes. This explains why, in some experimental studies, even greater strengths are obtained in recycled concretes than in conventional concretes.

When reference and recycled concretes are batched under similar conditions (same effective w/c ratio), a limited decrease in strength will be obtained (up to 50% of recycled aggregate) (Fig. 12.6) (Sánchez de Juan and Alaejos, 2004).

Table 12.1 Limit values in RCA to be used in concrete manufacture (EHE-08)

Property	Limit value (%)
Concrete particles + natural unbound aggregates	>90%
Ceramic particles	<5%
Asphalt particles	<1%
Impurities (gypsum + glass + others)	<1%
Water absorption capacity	<7%
Los Angeles abrasion coefficient	<40%
Flakiness index	<35%
Acid-soluble sulphate content (SO_3)	<0.8%
Total sulfur content (S)	<1%
Total organic matter content	<0.5%



12.6 Compressive strength of recycled concretes (20, 30 and 50% of coarse concrete aggregate) and reference concretes (Sánchez de Juan and Alaejos, 2004).

This figure shows that the reduction in compressive strength is negligible for concretes with a strength of less than 50 N/mm^2 and slightly increased for high strength concretes. A maximum strength loss of 6% could be expected for a substitution of up to 50% in concretes with less than 40 N/mm^2. This is not considered an important reduction, taking into account that these recycled concretes could be designed using almost the same amount of cement as conventional concretes.

Table 12.2 Example of concrete mix proportions applied with different degrees of substitution of coarse NA by RCA

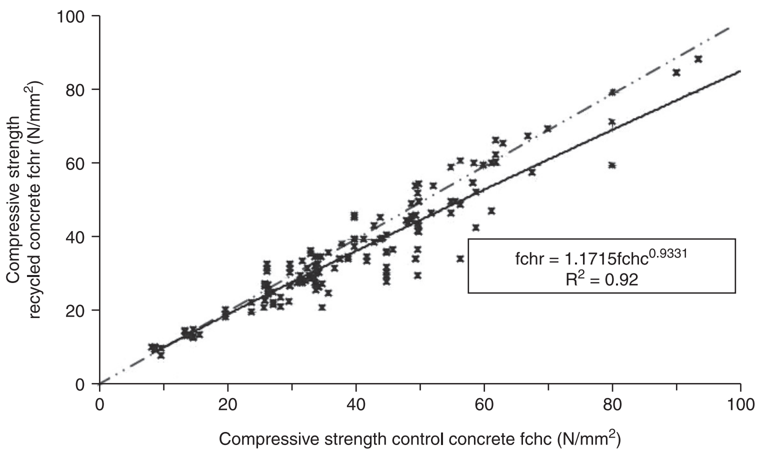
		Density (gr/dm^3)	NA-100%		RCA-20%		RCA-50%		RCA-100%	
			(kg)	(dm^3)	(kg)	(dm^3)	(kg)	(dm^3)	(kg)	(dm^3)
Concrete mix proportions	Cement	3.10	300.0	96.7	300.0	96.7	300.0	96.7	300.0	96.7
	Mixing water	1.00	180.0	180.0	180.0	180.0	180.0	180.0	180.0	180.0
	Coarse natural aggregate	2.65	1070.6	404.00	841.9	317.7	513.3	193.7	0.0	0.0
	Natural sand	2.60	895.00	344.23	895.0	344.2	895.0	344.2	895.0	344.2
	Coarse RCA	2.44	0	0	210.5	86.3	513.3	210.4	985.7	404.0
	Pre-saturating water	1.00	0	0	11.7	11.7	29.3	29.3	58.6	46.4
Water/cement ratio							0.6			
Mixture weigh (kg/m^3)			2445.6	1025	2427.4	1025	2401.6	1025	2360.7	1025

Recycled concretes with 100% RCA

There can be a significant decrease in compressive strength when 100% coarse recycled aggregate is used. Figure 12.7 (Sánchez de Juan and Alaejos, 2004) compiles a large amount of data collected from literature in many different research projects. It can be seen in the regression fitted to the data how the reduction is higher, the greater the compressive strength level. Moreover, many results are under this average curve, meaning that frequent lower values can appear, usually related to the use of low-quality RA, so reinforcing the importance of selecting high-quality RA for structural applications.

If we take into account both Figs 12.6 and 12.7, and the regressions fitted to the data, we can estimate the relative resistance for recycled concretes related to the strength and the replacement level (Table 12.3).

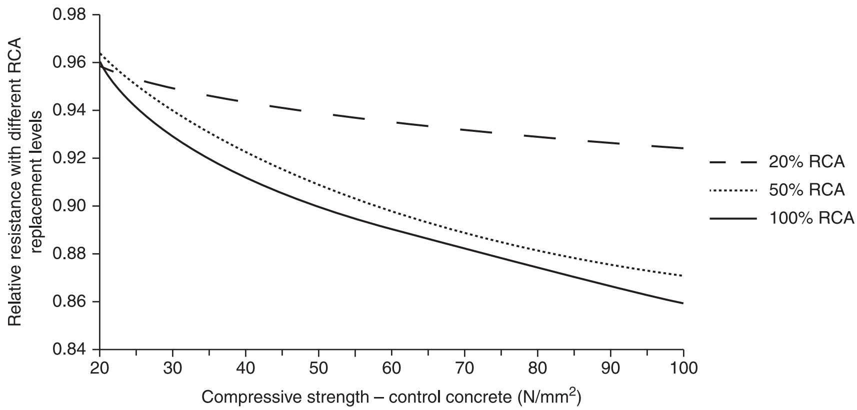
Figure 12.8 illustrates the relationship between the RCA replacement percentage in concretes with different compressive strength levels, and the relative resistance



12.7 Relationship between conventional and recycled concretes' compressive strength (100% coarse RCA) (Sánchez de Juan and Alaejos, 2004).

Table 12.3 Conversion factor for compressive strength on recycled concrete manufactured with RCA with different substitution levels

Compressive resistance in conventional concrete	Relative resistance recycled concrete Replacement 20%	Relative resistance recycled concrete Replacement 50%	Relative resistance recycled concrete Replacement 100%
20–50 N/mm^2	0.95	0.94	0.93
50–100 N/mm^2	0.93	0.89	0.88



12.8 Influence of RCA content and compressive strength on relative resistance of recycled concrete.

for recycled concretes. It can be seen how this factor decreases with a higher replacement level and the compressive strength value achieved.

The reduction of compressive strength in recycled concretes can be overcome by optimising their mix design (w/c ratio and cement content), although this method must be carried out under a rational point of view, as described in detail in Section 12.4.

Recycled concretes with a recycled mixed aggregate (RMA) content

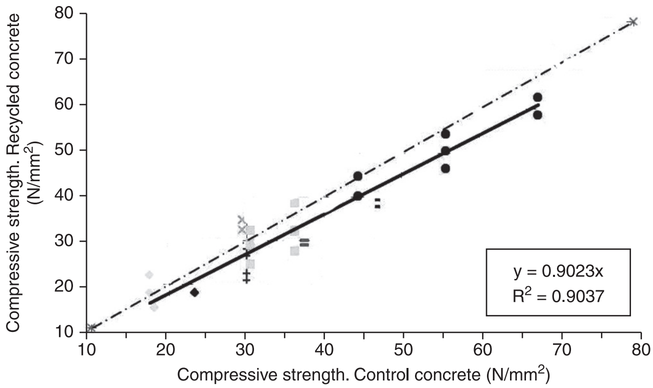
Most RMA come from processing wastes from building construction or demolition (Barbudo *et al.*, 2012). The aggregates obtained from these residues are composed of a mix of concrete, ceramics, NA, asphalt particles and impurities such as gypsum, plastics and wood. If the original wastes have not been properly separated, the amount of impurities increases. The composition of these aggregates can vary depending on the nature of the waste, but as an average it can be considered that 80% by weight is concrete or rock particles, with the remainder being ceramic or masonry (10–15%), asphalt particles (3–8%) and finally 2 to 6% of impurities (Agrela *et al.*, 2011). These aggregates are usually applied in the production of non-structural concrete. In order to obtain a minimum concrete quality, the aggregates must fulfil some requirements and Table 12.4 shows some recommended composition values and properties of RMA (IHOBE-CEDEX, 2011).

Figures 12.9 and 12.10 are taken from Geráldes (2012) and compile a number of results collected from literature, both for partial and total replacement of coarse aggregate. It can be seen that the detrimental effect on concrete compressive strength is higher than for RCA and how the reduction is higher the greater the compressive strength level.

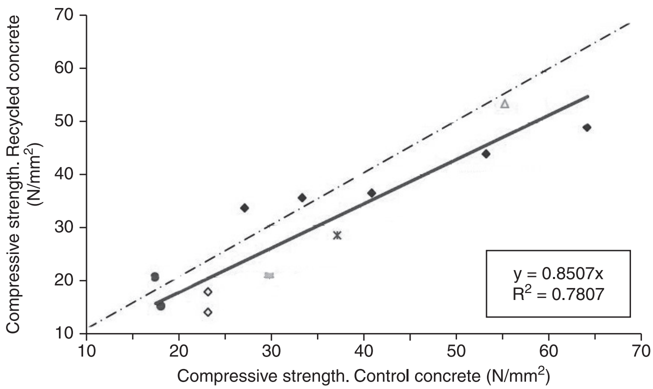
Table 12.4 Limit values in RMA to be used in concrete manufacture

Property		Upper Limit value (%)
Concrete particles + natural unbound aggregates		75%
Maximum percentage of ceramic particles + asphalt particles		25%
Asphalt particles		5%
Impurities	Total quantity	2%
	Gypsum	2%
	Glass	1%
	Others	1%
Acid-soluble sulphate content (SO_3)		1%
Water absorption capacity		12%

Source: Agrela *et al.*, 2011; IHOBE-CEDEX, 2011.



12.9 Influence of RMA on concrete compressive strength – replacement level up to 50% (Geraldes, 2012).



12.10 Influence of RMA on concrete compressive strength – replacement level 100% (Geraldes, 2012).

Table 12.5 Conversion factor for compressive strength of recycled concrete manufactured with RMA with different replacement percentages

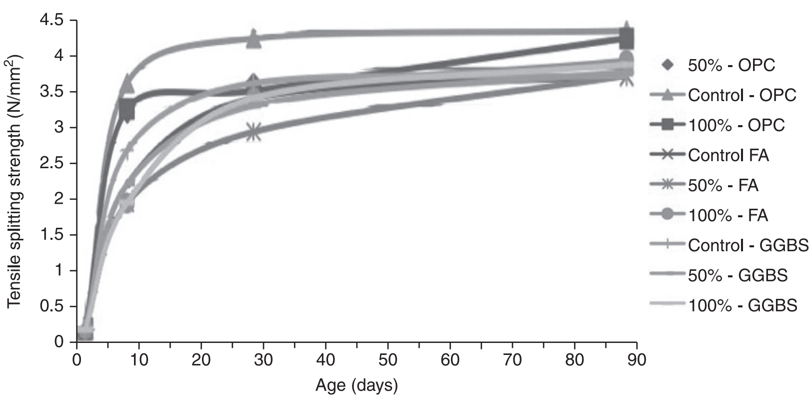
Compressive resistance in conventional concrete	Relative resistance recycled concrete Replacement 20%–50%	Relative resistance recycled concrete Replacement 100%
20–70 N/mm^2	0.90	0.86

These figures show that a partial replacement of coarse aggregate up to 50% leads to a 10% decrease in strength. A total replacement (100%) causes a strength loss equal to 15%. Table 12.5 shows the compressive strength conversion factor for recycled concretes manufactured with coarse RMA using different replacement levels.

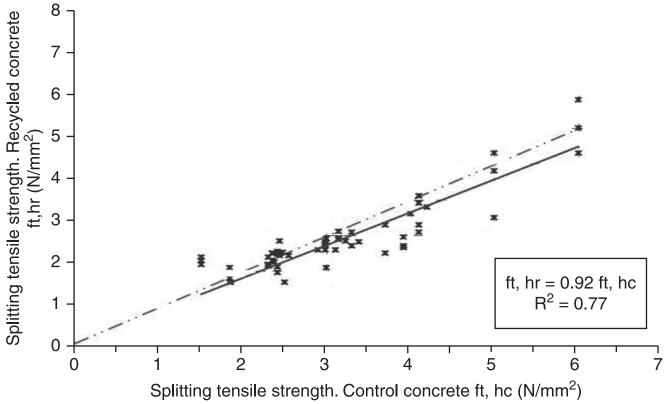
12.3.2 Tensile splitting and flexure strength

Both tensile splitting and flexure strength of recycled concretes exhibit much lower reductions compared to compressive strength. Figure 12.11 shows tensile splitting strengths obtained by Kou *et al.* (2011) with 50 and 100% replacements, also using fly ash in the mixes. It was observed that although initial resistances were higher for conventional concretes (recycled concretes exhibit a tensile strength reduction of 10% at 28 days), at later stages (90 days) the values were similar for all types of concretes.

Figure 12.12 show tensile splitting test results for concretes with 100% RCA (Sánchez de Juan and Alaejos, 2004), obtaining an average reduction of 8%. This value is in agreement with the decrease of 10% obtained at 28 days in the previous study.

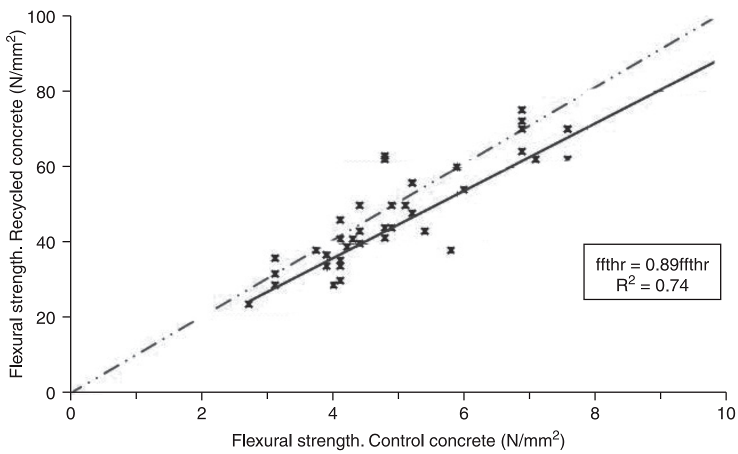


12.11 Tensile splitting strength of concrete mixtures manufactured with different mineral additions (Kou *et al.*, 2011).



12.12 Tensile splitting strength relationship between recycled and conventional concretes (Sánchez de Juan and Alaejos, 2004).

This improved behaviour of recycled concrete regarding its tensile strength can be explained by the good adherence paste-aggregate developed by the RA. These aspects were studied by Katz (2003) and Poon *et al.* (2006). Xiao *et al.* (2012) revealed that the splitting failure in concretes manufactured with RCA initiated not only from the interfaces between the RCA and new cement paste, but also from some of the RCA itself. Concerning flexural strength, again the differences observed between the recycled and conventional concretes are reduced. Figure 12.13 shows data from several studies, for which again a decrease of 10% is obtained for the recycled concretes.



12.13 Influence of RCA content on flexural strength (Sánchez de Juan and Alaejos, 2004).

12.3.3 Modulus of elasticity

Figure 12.14 shows the relationship between the modulus of elasticity of recycled concrete compared to conventional concrete, when 100% RCA is used. We can observe that the modulus of elasticity decreases by approximately 20% on average, so the conversion factor applied by international recommendations is 0.80 (RILEM, 1994). The decrease is due to the large amount of old mortar with comparatively lower modulus of elasticity, which is attached to original aggregate in RAC (Xiao *et al.*, 2012).

Several authors have reported cases where the elastic modulus is 30 to 45% lower than that of conventional concrete, as one important factor is the quality of the RCA. The previous figure contains many data located under the linear regression, representing cases where the modulus of elasticity was lower than the expected average value.

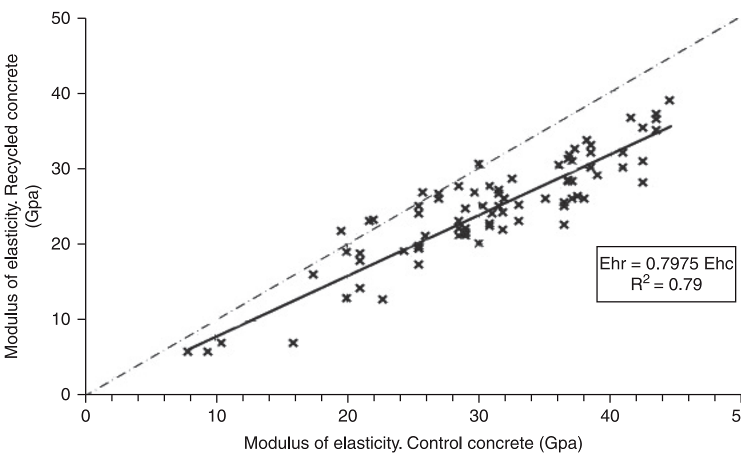
12.3.4 Density

NA have higher density than RA. This difference is not so great for RCA, and even when a total replacement of coarse aggregate is used, the final effect in the density of hardened concrete is minimal. Figure 12.15 shows some experimental results taken from the literature.

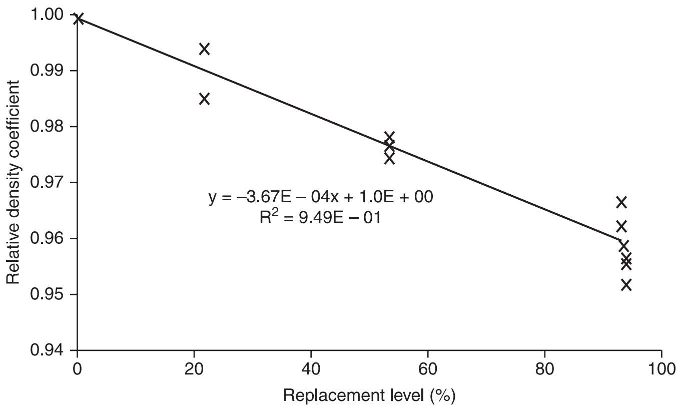
From a theoretical point of view, the expected decrease in recycled concrete density, when coarse RCA is used, can be easily estimated as:

$$d_{RC} = d_{CC} - V_{RA} \cdot V_{CA} \cdot (d_{NA} - d_{RA}) \qquad [12.1]$$

- d_{RC} = density of recycled concrete;



12.14 Influence of RCA content on modulus of elasticity (Sanchez de Juan and Alaejos, 2004).



12.15 Influence of RCA content on density of hardness concrete (Sanchez de Juan and Alaejos, 2004).

- d_{CC} = density of conventional concrete; normal value 2300 kg/m^3;
- V_{RA} = % recycled aggregate;
- V_{CA} = % coarse aggregate in concrete volume; normal value 40%;
- d_{NA} = density of NA; normal value 2550 kg/m^3;
- d_{RA} = density of concrete RA; normal value 2300 kg/m^3

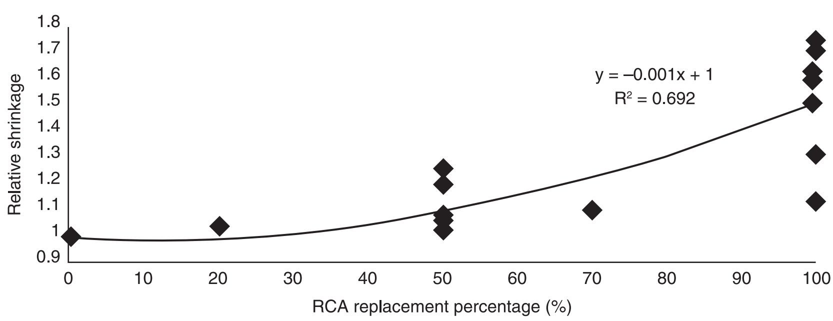
If we assume the normal values of the different parameters, then we obtain:

$$d_{R\,C}/d_{CC} = 1 - V_{RA}/100.0{,}4(2.55 - 2.3)/2.3 = 1 - 0.00043\ V_{RA} \qquad [12.2]$$

in agreement with the experimental relationship shown in Fig. 12.15.

12.3.5 Shrinkage

Shrinkage of recycled concretes with RCA is higher than conventional concretes, the increase depending on the replacement level and water to cement ratio (Xiao *et al.*, 2012). Regarding the recycled aggregate content, Fig. 12.16 includes data



12.16 Shrinkage as a function of RCA replacement percentage (Kou *et al*, 2011; Alaejos and Sanchez de Juan, 2012).

of shrinkage at 112 days from different studies. It can be seen that limited replacement of coarse aggregate (20%) has a negligible effect on shrinkage, a 50% of RCA increases shrinkage near 10% and if a total replacement of coarse aggregate is used, the conversion factor of 1.5. This last value has been adopted by international recommendations for 100% RCA (Rilem).

The increase of shrinkage is related to the lower restraining capacity of RCA particles compared to natural aggregate.

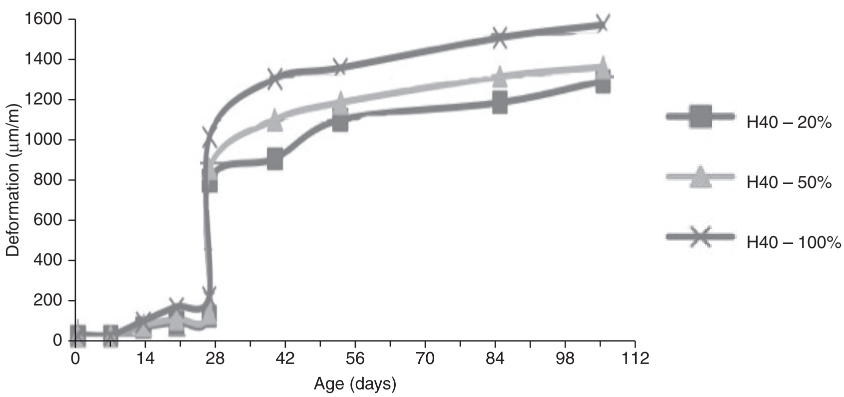
12.3.6 Creep

As in the case of shrinkage, creep of recycled concretes also increases corresponding to the higher replacement level of RCA used (Fig. 12.17) (Domingo-Cabo *et al.*, 2009). In this research, a creep at 180 days up to 70% higher for recycled concretes with 100% of RCA was obtained, but the values could be even greater, depending on the stress level applied (Xiao *et al.*, 2012).

The increased creep of recycled concretes is due to a greater proportion of cementous material included in the RCA (Limbachiya *et al.*, 2000).

12.3.7 Recycled concretes with a fine recycled concrete aggregate (F-RCA) content

Few studies have been carried out using the fine fraction of the RCA to produce recycled concretes. Evangelista and de Brito (2007) applied substitution levels of the F-RCA from 20 to 100% in concretes with different curing conditions. The compressive strength was reduced by 7.5% for total replacement of the natural sand by F-RCA. They concluded that when using good-quality recycled sand, the compressive strength is not affected up to 30% replacement ratios.

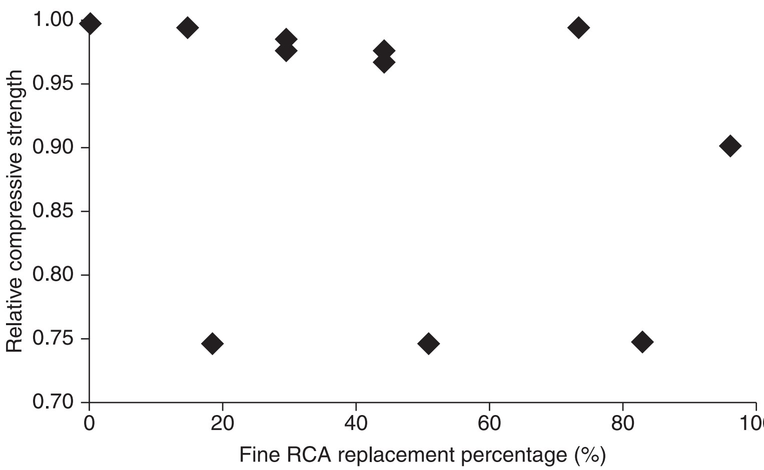


12.17 Creep as a function of RCA replacement percentage (Domingo-Cabo *et al*, 2009).

However, Khatib (2005) used sands originating from crushed brick, for which a compressive strength reduction of 15% was obtained, even when a limited substitution level was applied (25%). Figure 12.18 compares the results of relative compressive strength for both studies, when good- and poor-quality recycled sands are used.

With respect to other properties of concrete, Table 12.6 includes some results using high-quality fine RCA (Evangelista and de Brito, 2007; Zega and Di Maio, 2011).

As for compressive strength, the properties of recycled concretes produced with a limited percentage up to 30% of recycled fine aggregates present an appropriate mechanical and durable behaviour, but the reduction in mechanical properties is significant for those concretes with total replacement of natural sand by recycled concrete sand. Finally, Katz (2003) observed that a cementing capacity still remains in the finest fractions of F-RCA, when they have been obtained by crushing, and not by screening. These fine aggregates present certain hydrated and non-hydrated cement content, so a slight increase in the total amount of cement in



12.18 Compressive strength as a function of fine RCA replacement percentage.

Table 12.6 Properties of recycled concretes manufactured with fine RCA

Properties	Fine RCA 20%	Fine RCA 30%	Fine RCA 100%
Tensile splitting strength	0.98	0.93	0.77
Modulus of elasticity	0.98	0.95	0.81
Water penetration capacity	1.13	1.20	–
Shrinkage at 112 days	1.01	0.99	–

the recycled concrete mix is obtained, and this fact improves mechanical behaviour in recycled concretes.

12.4 Summary: using recycled aggregates successfully in concrete

The use of RA in the production of concrete causes a loss of compressive strength, higher when mixed aggregates are used, and increasing further with the aggregate replacement level.

12.4.1 Technical aspects

From a technical point of view, it is possible to produce high strength recycled concretes, increasing the cement content or reducing the water/cement ratio to compensate for this loss of strength.

For structural concretes, it must be taken into account that the use of RA also affects other mechanical properties such as the modulus of elasticity, creep or shrinkage, and the effect is even more important than that on compressive strength. For these other properties, the increase of cement will not compensate the loss. So for a rational use of these materials, it would be recommended to select good-quality RA (mainly concrete coarse aggregates) to produce structural concretes, as well as to limit the required maximum compressive strength, related to the percentage of RA used. These limitations are usually included in the international standards for structural recycled concretes (Table 12.7), although there are differences in the limits applied in different countries.

Most of the RA (both concrete and mix) can be used in the production of non-structural concretes, but in this case it is necessary to establish some technical requirements to avoid problems related to consistency variation, excessive shrinkage, durability, and so on. International standards include limitations to the compressive strength of concrete according to the percentage of RA, as a way to guarantee a minimum concrete quality (Table 12.8). Limits again vary from one country to another.

12.4.2 Environmental aspects

Another important point for a rational use of RA is the increase of cement sometimes necessary to compensate the loss of compressive strength, and its negative environmental effect (increase in CO_2 emissions). Figures 12.19 and 12.20 show the loss of compressive strength, both for 100% concrete and mixed coarse RA (Alaejos and Sánchez de Juan, 2012). The dotted line represents the theoretical reduction of w/c in the mix design of the recycled concrete that would be necessary to maintain the same strength as for conventional concrete.

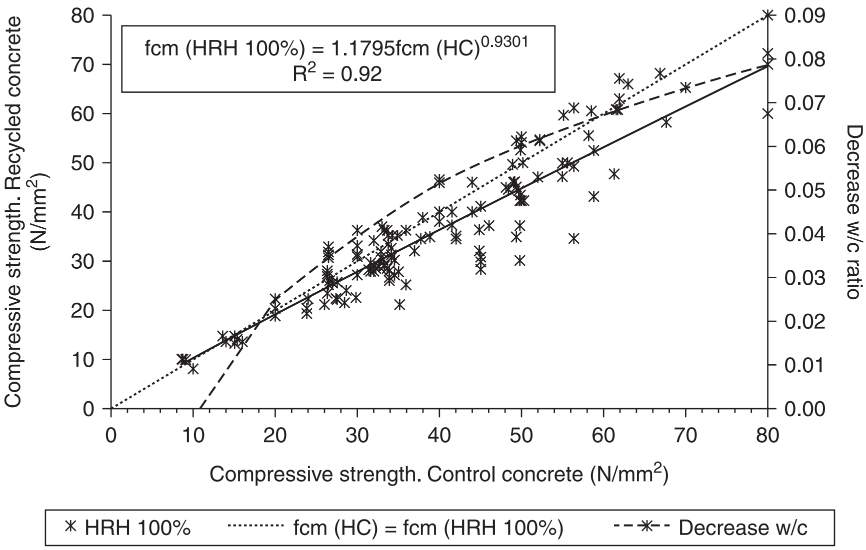
Table 12.7 Limits applied in different countries to the percentage of recycled aggregate and the compressive strength of concrete for structural application

Standard	Coarse recycled aggregate type and replacement level	Categoria resistente (N/mm^2)
Rilem	Type II (100% concrete aggregate) Type III (≤20% concrete aggregate)	50 No limit
Hong Kong	100% ≤ 20%	20 25–30
Belgium	100% type GBSB II (concrete aggregate) ≤ 20%	30 No limit
Holland	100% and 40 kg/m^3 increase of cement 100% and 50 kg/m^3 increase of cement ≤ 20%	17.5–22.5 (cubic) 27.5–45 (cubic) No limit
United Kingdom	100% (concrete aggregate) ≤ 20%	40 No limit
Japan (civil engineering)	I II III ≤ 20%	18–24 16–18 <16 –
Australia (non-structural)	100% (concrete aggregate) ≤ 20%	40 No limit

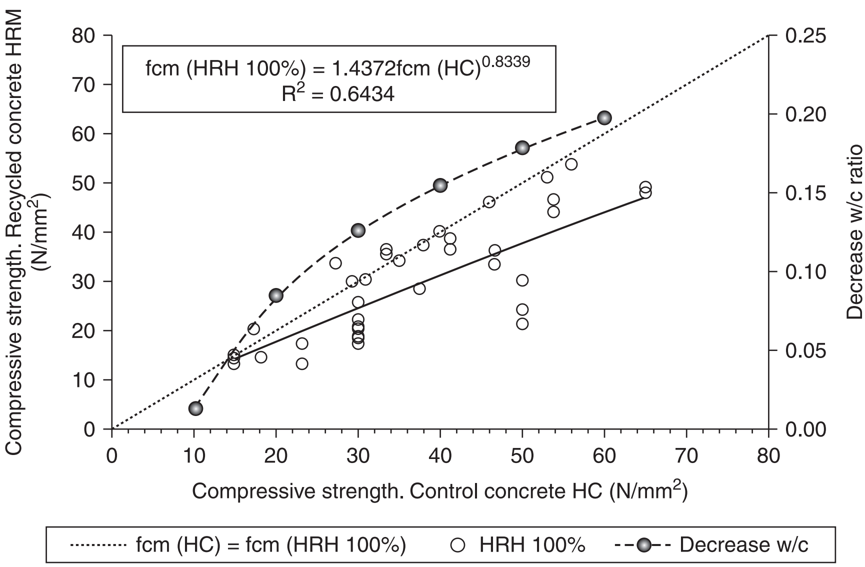
Source: Alaejos and Rueda, 2012

Table 12.8 Limits applied in different countries to the percentage of coarse recycled aggregate and the compressive strength of concrete for structural application

	Mixed recycled aggregate	Replacement (%)	Compressive strength (N/mm^2)
Rilem	Type I	100 100	16 N/mm^2(RA $d_{sss} \leq 2000$ kg/m^3) 30 N/mm^2(RA $d_{sss} > 2000$ kg/m^3)
Belgium	GBSB I	100	16 N/mm^2
Germany	Type 2 (%brick <30%)	35 25	≤ 25 N/mm^2 ≤ 35 N/mm^2
United Kingdom	RA	100 20	< 16 N/mm^2 > 20 N/mm^2
Holland	–	20	< 20 N/mm^2
Brazil	MRA	100	Non-structural concrete
Portugal	B2(%brick <30%)	20 100	≤ 35 N/mm^2 Reinforced concrete Mass concrete for fillings or levelling in non-aggressive environment
	RCA(%brick 30–90%)	100%	Mass concrete for fillings or levelling in non-aggressive environment



12.19 Relationship between compressive strength of recycled concrete with RCA, and conventional concrete and the necessary w/c decrease to compensate the loss of the recycled concrete (Alaejos and Sanchez de Juan, 2012).

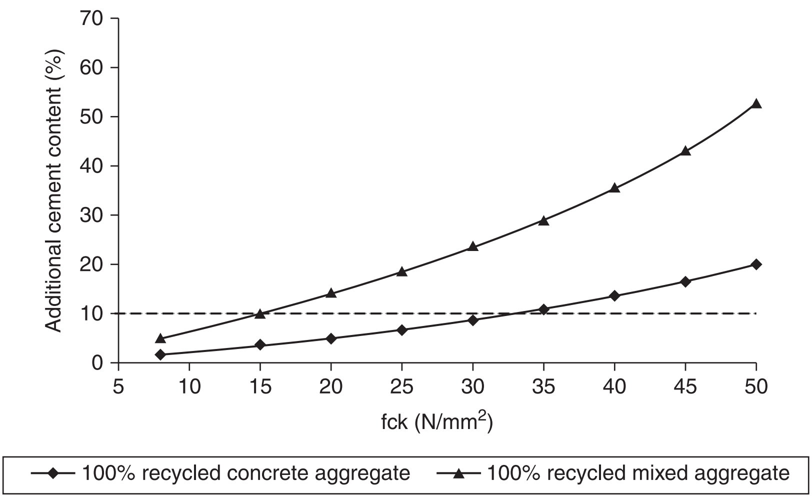


12.20 Relationship between compressive strength of recycled concrete with RMA, and conventional concrete and the necessary w/c decrease to compensate the loss of the recycled concrete (Alaejos and Sanchez de Juan, 2012).

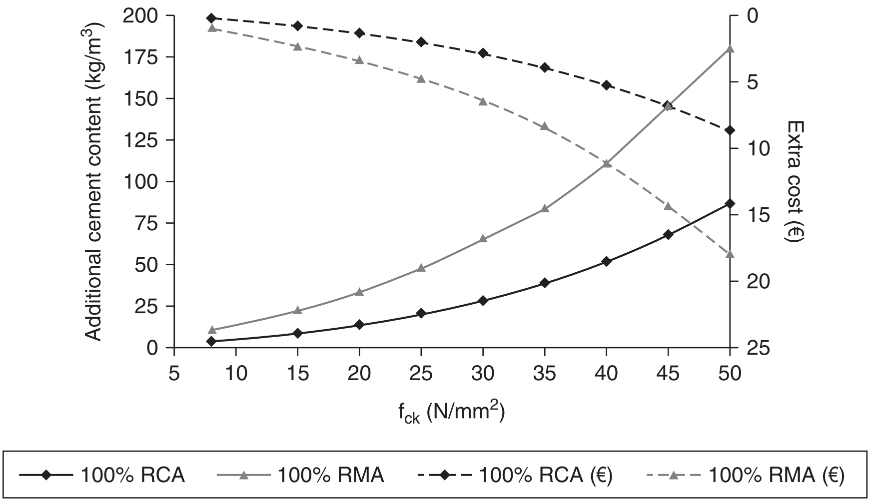
The reduction of w/c ratio can be achieved by increasing the cement content, then obtaining the values shown in Fig. 12.21 for concretes up to 50 MPa. This figure shows that an important additional cement content may be necessary to compensate the effect of the RA (up to 50% of additional cement for mixed aggregates).

From a rational point of view, if we admit a maximum of 10% of additional cement in the recycled concretes (Fig. 12.22), a maximum concrete strength category of C15 for mixed coarse aggregate and C30 for concrete coarse aggregate should be required. These recycled concrete categories (with limited extra cement consumption under 10%) coincide with those applied in Belgium (Tables 12.7 and 12.8). Other countries allow higher categories that are technically possible, but involve an excessive consumption of cement.

In the document 'A rational use of Recycled Aggregates' (Alaejos and Sánchez de Juan, 2012) is included a parallel study for partial replacement of coarse aggregate (20–50%), recommending in this case to limit the concrete strength category to C25 for mix aggregate and C40 for concrete aggregate (both related to a maximum 10% of extra cement consumption). These recommendable categories are near to those established by the German Standard (Tables 12.7 and 12.8). However, most international standards do not apply limits to the strength when RCAs are used, but again it must be taken into account that these recycled concretes will be produced with an excess of cement, diminishing their environmental advantages.



12.21 Extra cement content in recycled concretes with 100% of coarse RCA and RMA, to maintain the same strength as conventional concretes (Ulloa, 2012).



12.22 Extra cement content and associated cost in recycled concretes with coarse fraction of RCA and RMA, to maintain the same strength as conventional concretes (Alaejos and Sanchez de Juan, 2012).

12.4.3 Economical aspects

Figure 12.22 shows the extra cement content in kilograms per cubic metre of concrete, necessary in recycled concretes to maintain the same strength as conventional concretes. These figures can be directly translated to extra financial cost in the production of concrete, being higher or lower depending on the market cement price in each case. As an example, for a cement price of 100 €/t the figure shows the final impact in the concrete cost.

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13
Strength and durability of concrete using recycled aggregates (RAs)

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Abstract: There is growing pressure to produce and use more sustainable aggregates by reducing consumption of primary aggregates and switching to recycled or secondary aggregates. There are many advantages using this philosophy. The recycled materials are free from landfill tax, waste disposal charges and aggregate levy tax. This chapter examines the engineering qualities of concrete incorporating by products and recycled aggregates (RA). Conclusions are drawn that support the use of RA. The RA can provide suitable strength for structural concrete and with careful mix design, the same concrete can produce equal freeze/thaw durability if suitably protected with air entrainment or Type 1 polypropylene fibres.

Key words: recycled aggregate (RA), by product re-use, strength and durability.

13.1 Introduction: using recycled aggregates (RAs) in concrete

There is growing legislative and peer pressure on businesses, from government and competitors, to produce and use more sustainable aggregates by reducing consumption of primary aggregates and switching to recycled or secondary aggregates. Aggregate use in the United Kingdom is around 270 million tonnes annually and the construction industry accounts for 90% of this figure (Jones *et al.*, 2010). The use of recycled demolition waste as an aggregate in concrete has its drawbacks due to the heterogeneous nature of the material. The lack of consistency is a serious drawback for the designer and contractor when managing longer-term risks regarding durability; however, with robust control measures this can be ameliorated in most cases. For normal concrete, aggregates consist of sand, gravel and crushed stone and are a vital element in concrete.

In order to provide a solid, strong and uncontaminated mix, aggregates must be free from clay coatings and any other elements that could cause the concrete to weaken (Sustainable Concrete, 2009). BRMCA (2008) suggest that if demolition waste is used as an aggregate replacement, this will significantly increase the cement content and discredit the sustainability credentials of employing waste aggregate. This problem may be ameliorated with the use of a superplasticiser to reduce the water cement ratio and achieve similar compressive strengths. Richardson *et al.* (2009) and (2010) used unwashed and washed demolition waste as an aggregate and found an improved performance when the waste was pre-treated; however, there are time and cost implications when adopting this procedure. Hanson UK operate an aggregate plant that processes 65 000 tonnes of recycled aggregate (RA) per annum. The plant washes all aggregates using a water recovery system and removes all material such as lightweight aggregates, metal, paper, wood and plastic. The silt recovered from the process is compacted and used in restoration of construction sites (Cotton, 2011).

> Construction and demolition waste can be used as RA in construction. The more thoroughly the waste is treated, the higher the quality of the aggregate. However, high-quality aggregate is expensive, and thus, economically unviable in countries where natural aggregate is cheaply obtained. (Martín-Morales *et al.*, 2011)

Construction and demolition (C&D) waste is generated in small quantities at locations that can be widely separated. Therefore, portable equipment is needed, which can be used and set up close to a demolition site. Many contractors do not possess such machinery and this is a major barrier for 'newcomers' in the field of C&D waste management. Transporting waste over large distances makes the proposition of using C&D waste uneconomical and environmentally unsound. Commissioning of appropriately located recycling crusher units in a pilot plant can help in lowering barriers against recycling of C&D waste (Rao *et al.*, 2007). A concern to the author is the creation of a continuous and consistent supply chain, of RA as well as a consistent provision quality. The presence of hazardous materials in the C&D waste would negate any benefits from its use and this may be too costly to separate and make safe. A supply of C&D waste depends upon the redevelopment of buildings and infrastructure and this can be affected by the state of the national or international economy as well as fiscal policy.

One man's waste is another man's valuable resource, therefore it is important to adopt a culture of recycling and re-use to ensure that valuable resources are used to the full. There are many advantages to recycling and re-use and one of these is that once a material is not classed as waste, you do not have to follow waste controls such as using a registered waste carrier to transport your materials. This provides the constructor with greater options for distribution. Savings available to the contractor by using recycled demolition waste or similar materials are that they are free from landfill, waste disposal and aggregate levy taxes.

13.1.1 Standards used with RA

Generally in the UK, aggregates should conform to the standards listed in BS 8500–2:2006, 4.3. In making reference to aggregates conforming to these standards, there might be a need to specify or approve certain characteristics including size, grading, impurities, durability and other properties. BS EN 12620 and BS EN 13055–1 cover the use of natural, manufactured and RA. BS 8500–2 imposes additional requirements on RA. Neither of these standards uses the term 'secondary' aggregates, but such aggregates are covered by these standards, albeit under a different name (BS 8500:1, 2006).

BS EN 1744–6:2006 outlines tests for chemical properties of aggregates and determination of the influence of RA extract on the initial setting time of cement. BS EN 933–11:2009 outlines tests for geometrical properties of aggregates and a classification test for the constituents of coarse RA. BS EN 1260:2002+A1:2008 permits up to 90% replacement aggregate using crushed and graded concrete and mortar.

13.1.2 Aggregate types

RA are derived from reprocessing materials that have previously been used in construction and there are two methods of producing them. The best way to recycle is to create and use the aggregate at the site of the source, because the benefits of this are reduced transport costs and the environmental benefits of reducing transportation movement. The other way of dealing with RA is to produce them at a central plant. Examples of RA include crushed and graded recycled concrete from C&D waste material and railway ballast. RA passing a 5-mm sieve is not recommended for general use in concrete, because it usually has an adverse effect on water demand and may contain increased levels of contamination. In specific circumstances, where there is a high degree of control (e.g. fines from reclaimed product at a precast concrete works), 10% replacement of natural sand can be made without adverse effect on the product (BRE, 1998).

Secondary aggregates are usually the by-products of other industrial processes that have not previously been used in construction. Secondary aggregates can be further sub-divided into 'manufactured' and 'natural', depending on their source. Natural secondary aggregates include china clay stent and slate aggregate. Stent was successfully used in the construction of One Colman Place, London EC2 (finished 2007) and met the requirements of BS EN 12620 and PD 6682–1 for concrete aggregate. Stent is the term used to describe the waste granite rock material that has been separated from kaolin (china clay) by high-pressure water jets. For every tonne of china clay, approximately 4.5 tonnes of stent is produced along with other waste, which is usually tipped onto ever-growing surface spoil heaps. The use of this good-quality product saves landfill being used.

Quality control

Tests should be carried out on RA at monthly intervals to determine the quality of the product. These tests comprise 'Influence on initial setting time of cement' (EN 1744–6), 'Constituents of coarse RA' (prEN 933–11), 'Particle density and water absorption' (EN 1097–6) and 'Water-soluble sulphate' (EN 1744–1). Where the concrete is to contain RA, the alkali contribution from the RA shall be 0.20 kg Na_2O per 100 kg of RA; this will prevent damaging alkali silica reaction (ASR) (BS

8500–2:2002 complimentary standard to BS 206:1). ASR needs greater consideration where RCA is used. The mortar surrounding the aggregate particles will increase alkali levels and there is the possibility that the source concrete for the RCA includes reactive aggregates, although the risk of expansion would be reduced because of the higher porosity of RCA (*Design Manual for Roads and Bridges*, 2007).

Virgin sand replacement – crushed glass

One material that can be used as a sand replacement is crushed glass. Glass is simply heated sand and it is not a huge step to recycle it and use it in concrete. Glass has an elastic modulus of around 70 kN/mm^2, which is comparable to normal weight aggregate and it has zero water absorption, which is very desirable with regard to durability (Dhir *et al.*, 2005). Particle size is important when using crushed glass as a sand replacement or a pozzolanic material. The presence of finely ground glass (<1 mm) may cause ASR in the concrete. ASR is defined by a reaction between reactive forms of silica and sources of soluble sodium and potassium that form a gel internally, and this is capable of absorbing large volumes of water, leading to expansion and cracking of the concrete. If ground granulated blast furnace slag (GGBS) is used as partial cement replacement, there is a tendency to reduce the effects of ASR in the concrete. ASR occurs when 4.5 kg of alkali or more is present in 1 m^3 of concrete; the use of GGBS reduces the cement content and lowers the alkalinity. As a general rule, substituting CEM 1 with up to 70% of GGBS will act as a safety measure against ASR, and the other benefits of using GGBS are considerable.

An alternative to GGBS to control ASR is the use of lithium hydroxide monohydrate or lithium nitrate and this will limit the expansion to less than 0.6 mm/m, at which point the expansion is considered to be permissible. Lithium nitrate is the better option than lithium hydroxide, as the material is less corrosive and causes fewer handling problems, whilst suppressing the effects of ASR. Lithium does not prevent the reaction of ASR but it modifies the reaction so that the material is non-expansive, thus not causing any internal pressure from gel formation (Hooper *et al.*, 2001). Glass used above 1-mm sieve size tends to be less troublesome and reacts with the alkaline products of cement hydration leading to the formation of calcium silica hydrate gel, which contributes towards the compressive strength development. An alternative use of glass as an aggregate is to create lightweight granules (0.04–16 mm) from 100% recycled glass. The concrete produced with these granules has a density between 450 and 800 kg/m^3 and the product is mainly used for internal walls due to its low compressive strength (Brdlik and Poraver, 2010).

Virgin sand replacement materials (foundry sand)

Ferrous and non-ferrous metal-casting industries produce several million tons of sand by product worldwide (Singh and Siddique, 2011) and this can be used as a sand replacement in concrete. Soutsos *et al.* (2010) investigated the use of crushed and graded concrete and brick as fine aggregate replacement and found that if used as a partial sand replacement, a quality product could be manufactured.

13.1.3 RA classification

The *Design Manual for Roads and Bridges* (2007) defines three types of RA:

- **RCA (I)** defines the lowest quality material. It could have a relatively low strength and high levels of impurities, which might contain up to 100% brick or block masonry, or could comprise mainly concrete but with high levels of impurities.
- **RCA (II)** defines a relatively high-quality material comprising mainly crushed concrete with up to 10% brick by weight but low levels of impurities, less than 1.5% by weight (wood, asphalt, glass, plastics and metals). In some cases, it could contain an appreciable amount of natural aggregate (NA).
- **RCA (III)** defines a mixed material with up to 50% brick and high levels of impurities. BS 8500–2: 2006 also has a description of RCA, which defines the quality and impurities in RCA in a slightly different way to BRE (1998). This is also acceptable. However, asphalt impurities should be excluded from all concrete that is exposed, and the limit is set to less than 0.5% accordingly. In such circumstances, the limit of masonry impurities may be increased to less than 9.5% by mass, and for lightweight material (floating stony materials only), less than 1000 kg/m^3 should also be less than 0.1%, as allowed in BS EN 12620.

The total amount of contaminants should not exceed 11.5% of the aggregate and these are defined in BS 8500 and BRE Digest 433. The flakiness index shall not exceed 35 and the dry density should not be less than 2000 kg/m^3 to comply with RCA 111.

13.2 Factors affecting the durability of concrete

13.2.1 Durability and design working life

'Durability' as defined within Eurocode 1, DD ENV 1991–1; 1996, Section 2.5:

> . . . as an assumption in design, that the durability of a structure or part of it in its environment is such that it remains fit for use during the design working life given appropriate maintenance. The structure should be designed in such a way that deterioration should not impair the durability and performance of the structure having due regard to the anticipated level of performance.

Durability is further defined within Eurocode 2 (BS EN 1992–1–:2004), stating:

> A durable structure shall meet the requirements of serviceability, strength and stability throughout its intended working life, without significant loss of utility or excessive maintenance.

The satisfactory durability of CEM 1 (Ordinary Portland cement) concrete is a major reason why it is the world's most widely used construction material. Areas of concern that will adversely affect the durability of concrete are material limitations, which is particularly important when considering aggregate replacement, design and construction practices and severe exposure conditions that can cause concrete to deteriorate, which may result in aesthetic, functional or structural problems.

'Design working life' is defined within BS EN 1990: 2002+A1:2005:

> The design working life is the assumed period for which a structure is to be used for its intended purpose with anticipated maintenance but without major repair being necessary anticipated.

An indication of the required design working life is given in Table 13.1.

Concrete can deteriorate for a variety of reasons, and concrete damage is often the result of a combination of factors, starting with the initiation of cracks, which lead to processes that involve deleterious chemical reactions. The rate of crack propagation is controlled by ionic/molecular transport, producing micro-structural changes degrading the physical properties of the concrete and reducing the corrosion resistance of the steel reinforcement, if present in the concrete. Steel reinforcement generally suffers damage from water and air ingress due to oxidation of the steel and subsequent expansion of the steel, which causes surface spalling and structural damage. Concrete is subject to deterioration caused by absorption of moisture and thermal expansion and contraction. Extreme temperature ranges of both hot and cold can cause spalling. Moisture absorbed by the concrete expands and contracts with temperature changes and the resulting mechanical action can cause fractures and spalling.

Airborne pollutants, such as acid rain and carbon dioxide, can cause adverse chemical reactions, which can cause surface deterioration. This effect may be

Table 13.1 Indicative design working life

Required design working life in years	Example
1–5	Temporary structures
25	Replaceable structural parts, e.g. bearings
50	Building structures and other common structures
100	Monumental building structures, bridges and other civil engineering structures

Source: BS EN 1990: 2002+A1: 20051, Table NA.2.1.

exacerbated when C&D waste is used as an aggregate replacement due to the waste already containing deleterious materials from its first use. Without proper concrete design and production control, deteriorating concrete can compromise structural integrity, pose serious liability issues and create significant problems throughout a structure. Environmental factors such as seasonal temperature variations, cyclical freezing and thawing, rainfall and relative humidity changes, and concentration of deleterious chemicals in the atmosphere/water in contact with the concrete are the main causes of degradation. Geographical location is an important consideration with regard to durability, as are multiple, severe freeze/thaw cycles, which are worse for the destructive stresses applied within the concrete, than an extremely low constant temperature.

Design for the correct exposure class in accordance with BS EN 206 – 1/BS 8500 is critical to prevent breakdown and spalling of concrete due to poor design. The normal means of assessing durability is to comply with BS EN 206, which defines durability in terms of minimum cement content, water cement ratio and minimum cover. This simplistic approach is deemed unsatisfactory for modern durable structures; however, it is a good starting point.

Concrete cover to reinforcement is a critical aspect of design and production, with regard to durability. Whilst corrosion of reinforcement is not exclusively associated with the durability of concrete, there is an association between reinforcement corrosion and the condition of the concrete structure which often leads to localised areas of spalling due to the corrosion of the steel rebar. Poor positioning of steel during the construction phase is often carried out to tolerances less than the designer's requirements and when using C&D waste as an aggregate, the cover may be of variable quality.

13.2.2 Types of concrete deterioration

> Concrete deterioration is mainly related to its permeability. Most researchers believe that a well designed and manufactured concrete is originally watertight, containing discontinuous pores and micro cracks. When subjected to extreme loading or weathering, concrete deteriorates through a variety of physical and chemical processes, which result in cracking. Cracks in concrete generally interconnect flow paths and increase concrete permeability. The increase in concrete permeability due to crack progression allows more water or aggressive chemical ions to penetrate into the concrete, facilitating further deterioration. Such a chain reaction of deterioration – cracking – more permeable concrete – further deterioration, may eventually result in destructive deterioration of the concrete structure. (Richardson, 2010c)

Surface scaling is perhaps the most evident as well as the most common form of freeze/thaw damage. Tensile stresses occur at the surface due to the action of ice plug formation and the subsequent expansion of water. This action causes the loss of small particles, thus the aggregate is left exposed to the environment. The use of de-icing salts subjects the concrete to thermal shock by lowering the temperature of the surface and sub-surface. This aggravates the effects of concrete degradation. Care must be taken when using C&D waste to ensure the it is not contaminated with salts.

Pop out (type 1) deterioration is a term used to describe a saturated aggregate split in two due to the internal pressure exerted by the action of concrete freezing. Most virgin aggregates have a much greater tensile strength then Portland cement paste; however, it is possible to encounter virgin aggregates that are weaker than the surrounding cement paste and these lead to severe pop out breakdown of the concrete when subjected to freeze/thaw conditions.

Pop out (type 2) is identified by the mortar cover being broken up by the hydraulic pressure formed due to the freezing of concrete and the expansion of the entrapped water. 'D-line' cracking is defined within the *ACI Manual of Concrete Practice* (2000) as being a series of cracks in concrete near and roughly parallel to joints, edges and structure cracks. Hobbs (2002) suggests that surface cracking associated with freeze/thaw expansion creates cracks parallel to the exposed face, which decrease in intensity with the depth, changing to a random distribution of cracks often about 100 to 200 microns or more in width. D-line cracking develops along the joint or the edge of a concrete surface, because the concrete near the joints is weaker and more susceptible to freeze/thaw damage (Cordon, 1966).

In addition, stress concentrations at corners and edges of the concrete slab due to curling and warping contribute to D-line cracking. Stark and Klieger (1973) found that decreasing the maximum size of the coarse aggregate reduces the rate at which D-line cracking develops; however, it does not necessarily completely stop D-line cracking. Internal cracking can be caused if the concrete is subject to freezing prior to the initial set taking place. In such cases, the internal damage is often severe. In the case of fully cured concrete, internal cracking generally occurs when concrete is frozen when saturated. When selecting C&D waste aggregate, it is worth considering the aggregate size to reduce the tendency of D-line cracking.

The influence of environmental factors on the various deterioration mechanisms involved causes the micro cracks to propagate until they become continuous. In essence, the permeability of concrete influences the primary method of transport of moisture and aggressive ions into the concrete matrix and subsequently increases the permeation properties that are responsible for the increased rate of damage. Thereafter, crack growth accelerates the penetration of aggressive substances into the concrete and the spiral of deterioration continues.

Carbonation

Yoon *et al.* (2007) suggest global warming has increased the atmospheric CO_2 concentration and temperature. With regard to the use of concrete in the built environment, carbonation has been identified as a critical process when considering life-cycle cost and long-term durability. This view is supported by Pacheco Torgal *et al.* (2012), as they consider carbonation to be a major cause of concrete deterioration, which in turn leads to expensive maintenance.

Concrete carbonation is a process by which atmospheric CO_2 reacts with cement hydration products to form calcium carbonate. According to Yoon *et al.* (2007), the carbonation process reduces the pH value of the concrete from around 13 to 9. This has serious implications with regard to corrosion control of the rebar. Pacheco Torgal *et al.* (2012) suggest the alkalinity may be reduced to 8, which exposes the rebar to corrosion, as the optimum pH level is between 12 and 14 (Hobbs, 1988). The process by which corrosion occurs starts with CO_2 entering the concrete and reacting with the calcium hydroxide available in the pore solution and then with the CSH once the calcium hydroxide has been depleted. The carbonation rate is controlled by the concrete's pore system/relative humidity, which is then subject to diffusion. The optimum relative humidity for diffusion to occur is between 50 and 70% (Pacheco Torgal *et al.*, 2012; Yoon *et al.*, 2007). When RA are used in concrete, it is suggested by Levy (2004) that the carbonation depth increases with the RA content.

Chlorides

RA may have high levels of chloride contamination, depending upon their previous use. Salt used for de-icing and sea water can transmit high levels of chlorides into porous materials. BS 8500 requires that concrete should not exceed a maximum chloride content, depending upon where it is used and what it is made from. An aspect of concrete durability affecting gas/water permeability is chloride contamination. As a general rule, Darcy's law is used for both water and gas permeability measurements.

The onset of corrosion due to chlorides depends upon two factors, the rate at which the free chloride penetrates the concrete and the threshold value. The threshold level is the concentration of chloride ions tolerated before corrosion takes place. Oxygen and water are required before breakdown of rebar occurs due to oxidation. Chloride levels required to initiate corrosion of steel range between 0.17 and 2.5 m/m of chloride atoms by mass of cement.

Studies have shown that the rate of diffusion of chlorides through concrete is increased by the application of an electrical field (Nilsson and Tang, 1995). This would indicate that a material capable of high electrical conductivity is also capable of a higher rate of ion flow (Lu, 1997). This view is further corroborated:

> The electrical resistivity of the concrete affects the ionic flow . . . a higher concrete resistivity decreases the current flow . . . The electrical resistivity of concrete depends upon the capillary pore size, pore system complexity and moisture content.

> Chlorides have little direct effect on concrete resistivity as the hydroxyl ions from the cement dissolved in pore water outnumber the few chloride ions (Broomfield and Millard, 2002).

This statement as to the effect of chlorides in water is not consistent with the work of Castellote *et al.* (2001); they imply that chloride concentration is directly proportional to conductivity measured in mS/cm.

Abrasion resistance

Due to the variable nature of recycled aggregates, there may be concerns as to the fines content, which in turn may lead to larger than normal degrees of surface laitance. Abrasion resistance may be specified for structural elements where surface wear is a concern for functional performance, such as for industrial floor slabs. The specifier should identify elements of the structure to which abrasion resistance requirements apply. The specifier has the option to require a maximum abrasion loss and state the associated test method, and to state the age at which such values are to be measured. Multiple abrasion tests are available, but each varies in the type of wear pattern induced and thus each may have varying relevance to the wear anticipated in service for the subject structure. Tests available to test for abrasion are ASTM C418, ASTM C779, ASTM C944 and ASTM C1138 (Soutsos, 2010).

13.2.3 Cause and effect of concrete failure

The following interrelated factors shall be considered to ensure an adequately durable structure:

- the intended and possible future use of the structure;
- the required performance criteria;
- the expected environmental influences;
- the composition, properties and performance of the materials;
- the choice of the structural system;
- the shape of members and the structural detailing;
- the quality of workmanship and level of control;
- the particular protective measures; and
- the maintenance during the intended life.

The mechanisms of deterioration of concrete and their rate of deterioration are controlled by the environment, the microstructure and the fracture strength of the concrete, which will be affected by the use of C&D waste. According to Martín-Morales *et al.* (2011):

> Elongated and slab particles lower the concrete quality. They decrease its workability, since they demand a greater quantity of water and sand. This reduces the strength of the concrete, and also requires an excessive amount of cement. Consequently, the particles break more easily.

The flakiness index needs to be established to determine the degree of elongation and particle shape when using C&D waste as an aggregate. Natural aggregate has a higher flakiness index than RA (i.e. 5–9%). In the case of RA, particle shape is determined largely by the crushing equipment. Impact mills used in the recycling plants produce cube-shaped aggregates. Experience at the recycling plant has shown that concrete tends to break into small blocks without generating slabs (Martín-Morales *et al.*, 2011).

The factors affecting durability in concrete, detailing durability in terms of input, transformation and output, are shown in Table 13.2.

Table 13.2 Cause and effect model for deterioration of concrete

External environment		Internal environment
Water →	Holistic model for deterioration of concrete	Compressive, tensile and flexural strength
		Water cement ratio
Chlorides →	**'Change'**	Air content
		Cement type
Loading type and rate →		Pozzolanas and micro fillers
		Covercrete composition
		Heartcrete composition
Temperature →		Bulk density
		Mix proportions
		Sorptivity
Pollutants →		Moisture content
		Adsorption
Chemical reactions →		Ion flow
		Resistivity, conductivity, capacitance
Gases →		Tortuosity of ionically conducting fluid
		Thermal conductivity
		Internal temperature gradients
		Sulphates and C-S-H including Ettringite formation
		Alkali silica reaction
		Capillary formation
		Pore formation
		Gel pore spacing and formation
		Crack control – micro and macro cracks
		Additives
		Fibre types
		Water reducing mixtures
		Stability of materials used in the life cycle of the product
		Carbonation
		Flakiness of the aggregate
		Viscosity modifying admixtures
Input		**Output (Deterioration)**

13.2.4 Summary

For construction demolition waste to be used effectively as a virgin aggregate replacement, care must be taken to use a sustainable source of C&D waste, RA or secondary aggregates that are of a guaranteed quality and are largely free from deleterious material.

13.3 Strength and durability of concrete using RAs

Previous research (Richardson *et al.*, 2009) suggests that the use of RA in concrete produce a reduction in compressive strength when compared with virgin aggregates. Meyer (2009) found that most reductions in strength for concrete made with recycled coarse aggregate were in the range from 5 to 24%, compared with concrete made with virgin aggregate. When both coarse and fine aggregate were obtained from recycled concrete, the strength reductions ranged from 15 to 40%, compared with concrete made with only naturally occurring materials. However, RA usage is not advisable without consideration of the pertinent differences its inclusion brings to concrete mix design. Sagoe and Brown (2002) have concluded that the density of recycled concrete aggregate (RCA) is lower than that of virgin aggregate concrete, due to the occurrence of porous residual mortar lumps within the demolished material.

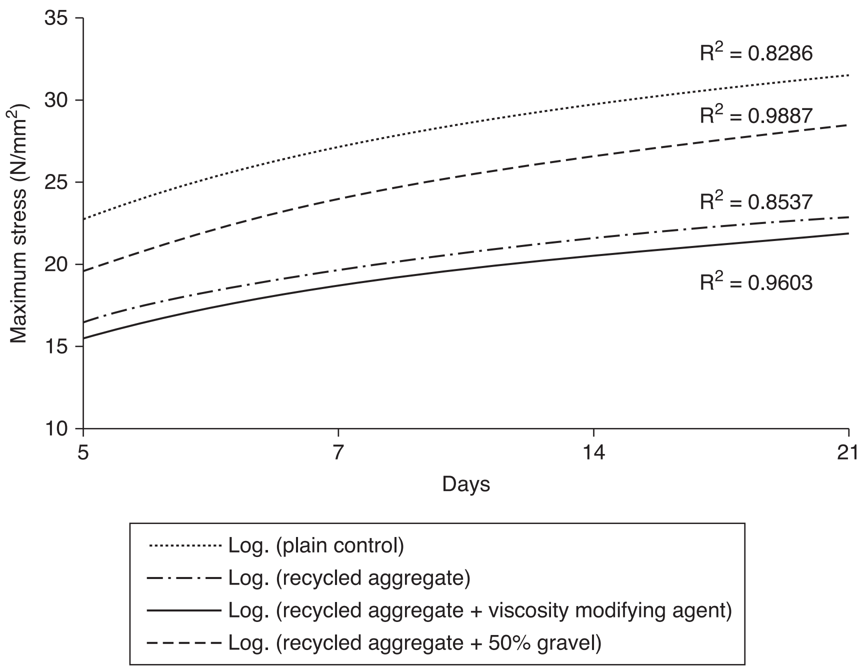
Collins (1994) states that as the structure of RA may contain voids and therefore it is usually the case that a higher water content is required to achieve a good standard or workability. Jones *et al.* (2010) state that mixed construction demolition waste has an absorption of 9.5%, whereas natural gravel has a value of 1.2%. Poon *et al.* (2008) verify the reduced workability and dimensional stability exhibited by concrete formed from recycled fine aggregates (<5 mm), which results from higher water absorption (>10%). Rao *et al.* (2007) suggest that the use of C&D waste as an aggregate induces large shrinkage due to the high absorption values of the aggregate.

Zaharieva *et al.* (2004) found that the high absorption rate of RA is the main obstacle in concrete manufacturing, as the freshly mixed RA concrete (RAC) quickly loses the initial workability, even when superplasticizers are used. To prevent the suction of the mixing water by RA, it is necessary to pre-soak them. The properties of concrete made using RA are also debatable, as the high porosity of the RA can mainly be attributed to the residue of mortar adhering to the original aggregate (Rao *et al.*, 2007) and highly porous aggregates are not desirable with regard to qualities of durability. Rao *et al.* (2007) suggest that the inclusion of fly ash, silica fume, etc. can improve the durability of the RA concrete mix.

Earlier work by Richardson *et al.* (2009) showed ungraded recycled demolition waste had a 54% strength reduction when compared to the plain concrete control sample. Washing/pre-treatment of C&D waste aggregate, as described by

Richardson *et al.* (2010a), showed that the action of grading the RA and washing out of the fine material reduced the strength loss of the recycled demolition waste aggregate concrete to 28% when compared to the plain control sample manufactured with virgin aggregates. This is a significant strength reduction using RA when compared to virgin aggregates. Washing and grading the C&D waste aggregates improved the compressive strength performance by 26% when the two tests were compared.

Etxeberria *et al.* (2007) suggested that a 25% coarse aggregate replacement with C&D waste does not adversely affect the shear strength of the concrete. Figure 13.1 shows the compressive strength development of various design mixes of concrete cubes compared to a plain control sample. Research and experience suggest that replacement of 20% of NA with RCA for reinforced concrete should have minimal effect on concrete properties or design issues and no special additional measures need be undertaken. Prior to completing the mix design, it is strongly recommended that trial mixes are undertaken when the use of RCA is proposed, and this may include the construction of trial panels to check on finishes and methods of placement. Depending upon the intended use of the concrete, it may be necessary for such panels to be tested by coring to verify adequate



13.1 Strength development of concrete types (Richardson *et al.*, 2010a).

compaction, and integrity of concrete surrounding reinforcement (*Design Manual for Roads and Bridges*, 2007).

A 50% C&D waste aggregate replacement shows that when comparing the RA concrete mixes against the plain control sample, the observed strength reductions were 28% for the washed graded recycled demolition waste, 30% for the same mix with the viscosity modifying agent, but only 11% for the mix with 50% gravel replacement. When compared to the control batch, the RA mix using 50% of RA was the most similar to the plain control batch.

13.3.1 Equivalent mortar volume (EMV) method of mix design

RCA contains a percentage of material containing mortar and if this is taken into account when designing a concrete mix, there is a likelihood that equal or better compressive strength can be achieved using RCA when compared to NA. Abbas *et al.* (2009) show that not only can compressive strength be achieved to an equal standard of concrete with virgin aggregates, but it also has a higher resistance to freeze/thaw action, chloride penetration and carbonation. These mixes were designed with the EMV method using RCA compared to mixes designed by conventional methods with virgin aggregates. The main design change is that the EMV method takes account of the residual mortar attached to the RCA as part of the total mortar volume requirement. The virgin aggregate volume of the RCA concrete was increased by an amount equal to the volume of the residual mortar of RCA.

The mix design is complex in respect that the residual mortar attached to the RCA has to be accurately determined experimentally. Once this has been determined, a proportion of RCA and virgin aggregate can be used in the concrete mix design. The derivations for the equations for the EMV method are available through Carleton University (Fathifazel, 2008). In support of using existing mortar contained in the C&D waste, Mymrin and Correa (2007) suggest that it is possible to produce new concrete from concrete waste, without requiring new cement, by adding 11% fly ash to the 79% of C&D waste aggregate.

13.3.2 Freeze/thaw durability using RAs

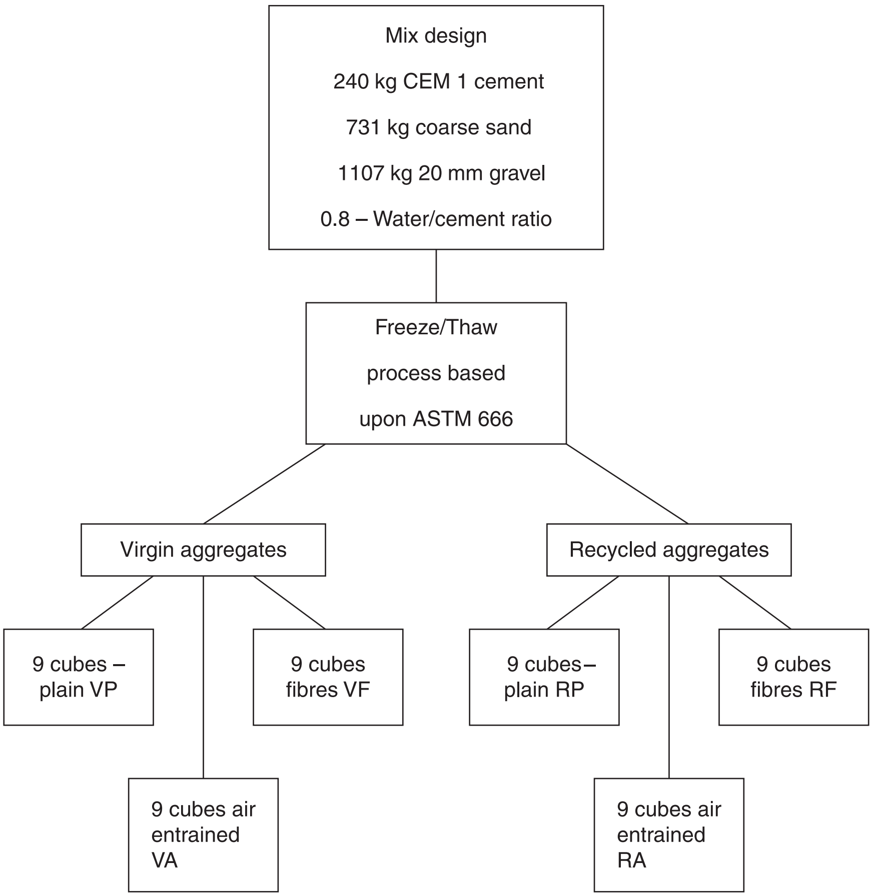
Richardson *et al.* (2010b) used three cubes from each of the six concrete mixes, which were used to determine the initial strength of concretes prior to any exposure to the freeze/thaw cycles (Table 13.3).

Figure 13.2 shows the batching and production range of the concrete as produced for freeze/thaw testing.

The freeze/thaw testing program was based on ASTM 666, where weight loss was examined and the freezing was carried out at −18°C in air and thawing was undertaken in water at 20°C until the core temperature of the test cubes reached

Table 13.3 Cube production numbers (Richardson *et al.*, 2010c)

Concrete	Mix description	Number of test cubes
Virgin aggregate	VP – Plain mix	9
	VA – Air entrainment added	9
	VP – Polypropylene fibres added	9
Re-cycled aggregate	RP – Plain mix	9
	RA – Air entrainment added	9
	RP – Polypropylene fibres added	9

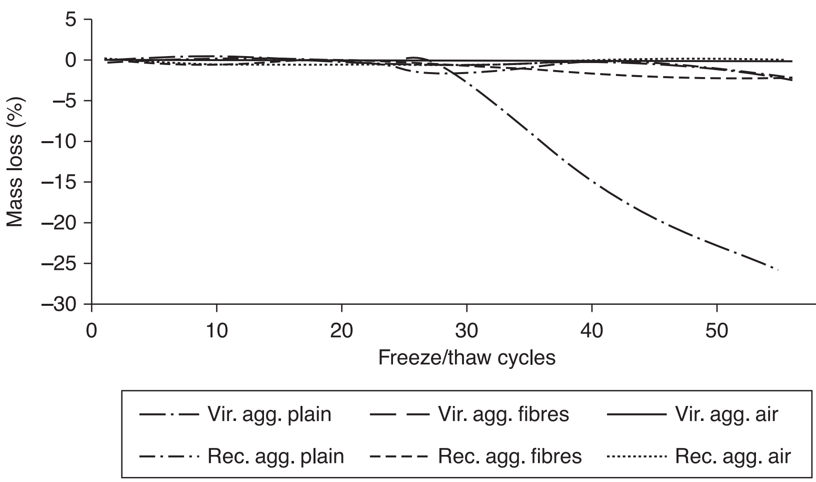


13.2 Manufacturing chart (Richardson *et al.*, 2010a).

6°C. BS 15177:2006 was used to inform the duration of the test, which was limited to 56 cycles. The remaining three concrete cubes from each mix were tested at the end of the freeze/thaw program to provide a strength comparison between the freeze/thaw cubes and the control cubes.

Richardson *et al.* (2010a) found that RA produced concretes of the highest density when compared to the plain concrete manufactured with rounded marine dredged gravel. The angular RA combined with the proportion of blue brick present in the RA mix was thought to produce a higher particle packing effect, which was observed within the relative compressive strength and density values. RA concrete was found to be of at least equal freeze/thaw durability to concrete manufactured with virgin aggregates. This was due to careful selection of the replacement aggregate and treatment prior to batching. Gokce *et al.* (2004) found that if RCA was used in an air entrained concrete mix subjected to freeze/thaw cycles, then concrete aggregate that was previously air entrained provided a full air entrainment system in the newly formed concrete; whereas plain RCA included in an air entrained concrete mix created an improper air void system that resulted in a poor standard of freeze/thaw resistance. As a general rule, Gokce *et al.* (2004) found that removing adhered mortar from RA was beneficial to the freeze/thaw performance of concrete manufactured.

Figure 13.3 shows the comparative performance of plain concrete, concrete with polypropylene fibres and concrete with air entrainment, manufactured with virgin aggregates and again with RA. The percentage mass loss trend was over a 56 freeze/thaw cycle duration, recorded every 7 freeze/thaw cycles. The very slight weight gain at the start of the test was due to water absorption and crack propagation.



13.3 Percent mass loss for freeze/thaw cubes (Richardson *et al.*, 2010b).

The results in Figure 13.3 show that the concrete cubes made with RA were satisfactory when compared with concrete made with virgin aggregate. The use of air entrainment and polypropylene fibres in concrete made with RA have been shown to be equally effective for providing freeze/thaw durability when compared with concrete made with virgin aggregates with air entrainment and polypropylene fibres. The results show that the concrete cubes made with RA were slightly more durable than those made with virgin aggregate.

13.3.3 Summary

Reusing concrete waste in concrete is an obvious choice, as the materials can be washed and returned to near their original condition. This has the added advantage of providing a recycled aggregate that has few of the problems associated with the use of ungraded and variable aggregates, such as construction demolition waste.

In the UK, the demand for aggregates for all uses is approximately 270 million tonnes per year, with 70 million tonnes of this demand coming from secondary and RA (BRMCA, 2008). If 70 million tonnes of material is being recycled per year, there needs to be a suitable supply chain to ensure continuity of supply and price stability in the RA market. Ensuring a supply chain of RA affords many potential gains, achieved through reducing the material volume transported to already over-burdened landfill sites, possible cost reductions to the contractor/client when considering the landfill tax saved and the potential for lower cost aggregate replacements, a reduction in the environmental impact of quarrying and the saving of depleting NA resources.

13.4 Conclusions

The key conclusions drawn from this investigation with regard to the use of RA, secondary aggregate and construction demolition waste as coarse aggregate in concrete are:

- Ensure a sustainable supply of suitable material.
- Ensure a repeatable quality of material free from any deleterious material.
- Take full account of the required characteristic strength and design the mix to match, taking into account different absorption qualities of the material used.
- Durability is dependant upon many design factors, so ensure the material is fit for the purpose.
- The use of RA can provide suitable strength for structural concrete, with a careful mix design and the same concrete can produce equal freeze/thaw durability if suitably protected with air entrainment or Type 1 polypropylene fibres.

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14
Recycled aggregates (RAs) for roads

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Abstract: This chapter presents the main physico-mechanical and chemical properties of recycled aggregates (RA) from construction and demolition waste (CDW), which are feasible for use as roadbed, sub-base and base unbound materials and cement-treated materials in road construction. The limiting properties for each of these uses are identified and recommendations for their use are given. Practical issues, environmental performance and bearing capacity and roughness measurements in the field are included, after which conclusions and commentary on future trends in the use of this material for roads are made.

Key words: recycled aggregates (RA), road, unbound section, cement-treated materials, environmental performance.

14.1 Introduction

Recycled aggregates (RA) composed of construction and demolition waste are an alternative to natural aggregates (NA) in road construction, which is the most common use for these materials. Their physico-mechanical and chemical properties mainly depend on the origin of the waste, demolition technique and treatment process in a recycling plant. These materials are usually classified into two groups according to their composition: recycled concrete aggregates (RCA) and mixed RA. The first is composed mainly of concrete particles and unbound aggregates and the second also contains ceramics, mortar, asphalt and harmful impurities such as gypsum. The first parts of the chapter present the main physico-mechanical and chemical properties of RA used for roads, excluding asphalt wastes.

The standardisation of the RA used in roads is currently underway. Most countries are implementing standardised tests and setting significant limits on the use of NA. These limits have been widely discussed and are necessary to the development of specific regulations for the use of RA in medium- and low-traffic-intensity roads. The second part of this chapter describes the appropriate use of RA as an unbound material in roadbeds, sub-bases and bases, as well as in cement-treated materials. The limiting properties for each of these uses are identified and recommendations for RA use in each situation are given.

An important step in creating technical specification requirements is the development of experimental sections. The third part of this chapter describes practical issues and the results of recent studies relating to the bearing capacity and roughness measurements of RA in field conditions.

Characterising the components that can leach into soil and cause potential risks to the environment is important from an environmental standpoint. The amount of harmful substances such as organic compounds, anions and metals should be low in RA. The fourth part of this chapter examines possible leaching analysis techniques and classifications for inert, non-hazardous and hazardous materials according to the European Directive, after which conclusions and commentaries on future trends in RA use are made.

14.2 Physico-mechanical characterisation of recycled aggregates (RAs) for roads

Aggregates that are recycled from construction and demolition waste are composed mainly of particles of concrete and mortar (Rc), ceramics (Rb), asphalt (Ra), fragments of stone and unbound aggregates without mortar attached (Ru), light particles (RL) and other materials that can be considered impurities, such as gypsum, wood, glass, plastic and metals (X). The components of recycled coarse aggregates that are greater than 4 mm in diameter can be analysed in accordance with EN 933-11:2009.

14.2.1 Composition

The composition of RA depends mainly of the source of the waste (structural concrete, masonry, roads, etc.), the demolition technique used and the treatment process in the recycling plant, and varies significantly between countries. Bricks are the main construction material in some regions, whereas concrete represents the majority in others; wood is a major construction material in northern countries such as Finland and Sweden. With the exception of excavated materials, cement-based materials are the predominant components found in the EU-27, followed by ceramics, unbound aggregates and asphalt (European Commission DG ENV, 2011).

Depending on the composition, RA are classified as RCA, recycled mixed aggregate (RMA) or recycled mixed ceramic aggregates (RMCA). Table 14.1

Table 14.1 Classification of RA according to composition

Type/Category	RCA	RMA	RMCA	Unclassified RA
Composition (EN 933-11)	$Rc+Ru \geq 90\%$	$Rc+Ru+Ra \geq 70\%$	$Rc+Ru+Ra \leq 70\%$	–
	$Rb \leq 10\%$	$Rb \leq 30\%$	$Rb \geq 30\%$	–
	$Ra \leq 5\%$	$Ra \leq 15\%$	$Ra \leq 15\%$	–
	$Xo+Xg \leq 1\%$	$Xo+Xg \leq 1.5\%$	$Xo+Xg \leq 1.5\%$	$Xo+Xg \geq 1.5\%$
	$Xg \leq 0.5\%$	$Xg \leq 1\%$	$Xg \leq 1\%$	$Xg \geq 1\%$
	$FL \leq 0.2\%$	$FL \leq 0.5\%$	$FL \leq 0.5\%$	$FL \geq 0.5\%$

Note: Xo = wood, glass, plastic, metals and other impurities; Xg = gypsum.

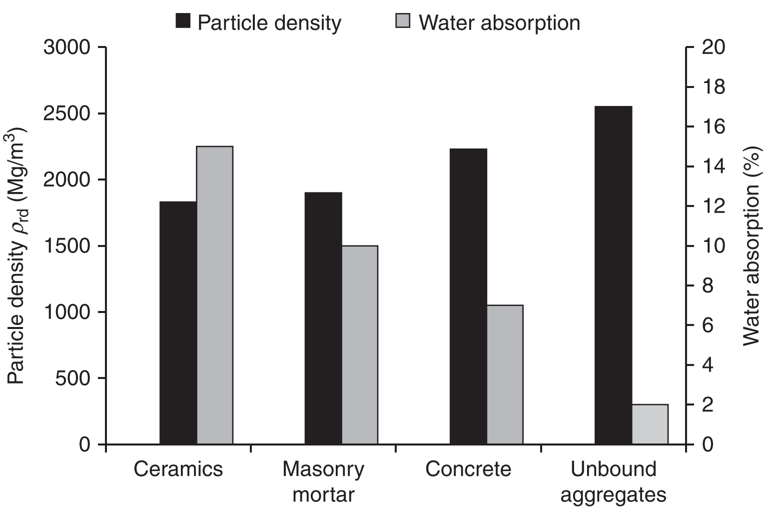
shows the typical composition of each category. A fourth category could be included for recycled materials with impurities that do not meet the composition requirements of the other categories.

The most commonly used classification system for RA is based on their composition, but an improved classification system that facilitates the use of RA in road construction should consider their physico-mechanical and chemical properties. Recent studies have indicated the need to establish sub-categories within each category that tabulate the specific amount of each component present in RA, including concrete and masonry mortar particles and the amount of red ceramic particles from brick and glazed ceramic from tiles, because the physico-mechanical properties of each component are very different.

14.2.2 Density and water absorption

The density and water absorption of particles can be determined with the European Standard EN 1097-6:2000. If the RA are composed of different particle sizes, it is necessary to separate the sample into the following fractions: 0.063/4, 4/31.5 and 31.5/63 mm.

Density and water absorption are related to the composition of RA. Figure 14.1 shows the mean values for the oven-dry density (ρ_{rd}) and water absorption of the main components in RA used for roads, and shows that the ceramic particles have the lowest density and highest water absorption, followed by masonry mortar and concrete particles. These values justify the density and water absorption values obtained for the different types of RA described in Table 14.1.



14.1 Particle oven-dry density and water absorption of RA components (8/31.5 mm fraction).

The presence of mortar in RCA gives these RA a rough and porous appearance, resulting in lower density and higher water absorption than that of crushed natural aggregates (CNA). Typical oven-dry density values range from 2240 to 2430 Mg/m^3 for the coarse fraction and 2130 to 2410 Mg/m^3 for the fine fraction. The water absorption values range between 4 to 7% for the coarse fraction and 6 to 9% for the fine fraction. These parameters are also related to the quality of the original concrete in the aggregate.

The coarse fraction of RMA has a lower density than that of RCA, due mainly to the presence of red ceramic particles. Typical RMA density values range from 2010 to 2290 Mg/m^3 for the coarse fraction and 2090 to 2400 Mg/m^3 for the fine fraction. RMA have higher water absorption than RCA, with typical values ranging from 6 to 11% for the coarse fraction and 4 to 10% for the fine fraction. These differences are more pronounced in RMCA, with typical density values ranging from 1870 to 2030 Mg/m^3 for the coarse fraction and 2100 to 2330 Mg/m^3 for the fine fraction. Typical water absorption values range from 11 to 15% for the coarse fraction and 5 to 10% for the fine fraction.

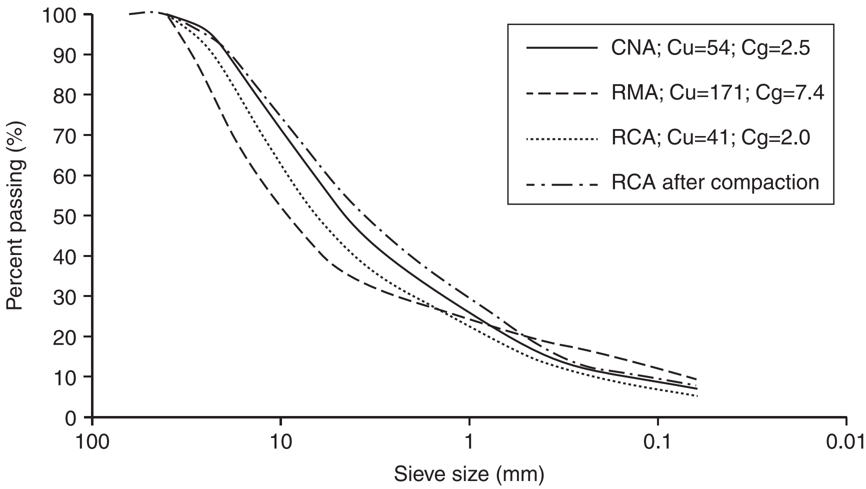
As a reference, the CNA used in road pavements have typical density values ranging from 2550 to 2680 Mg/m^3 and typical water absorptions ranging from 0.5 to 1.8 %, with very similar results for both the coarse and fine fractions. Recent studies (Agrela *et al.*, 2011; Barbudo *et al.*, 2012) have shown that the particle density of the coarse fraction in RA decreases almost linearly with the percentage of ceramic particles, while the water absorption behaves otherwise. However, no relationship has been identified between the results of composition tests and the density and water absorption of the fine fraction.

14.2.3 Particle size distribution

The particle size distribution curves (PSD) of RA can be analysed in accordance with the European Standard EN 933-1:2006. RA have continuous curves, indicating greater opportunities for interactions between particles and the ability for a greater degree of compaction. Figure 14.2 shows the continuous PSD, the coefficients of uniformity (Cu) and gradation (Cg) of typical RCA, RMA and CNA used as granular material in road construction. The fine fraction (< 0.063 mm) is usually less than 10% of the total. The graph also shows that the initial PSD of RCA is modified after the compaction process due to the breakage of the less resistant particles in the material. This breakage is a key issue during construction work.

14.2.4 Particle shape

The particle shape of the coarse fraction in RA is determined by the flakiness index and percentage of crushed particles. The flakiness index can be determined in accordance with the European Standard EN 933-3: 1997. This method determines the percentage of particles that have a thickness (smallest dimension) of less



14.2 Particle size distribution curves of RCA, RMA and CNA.

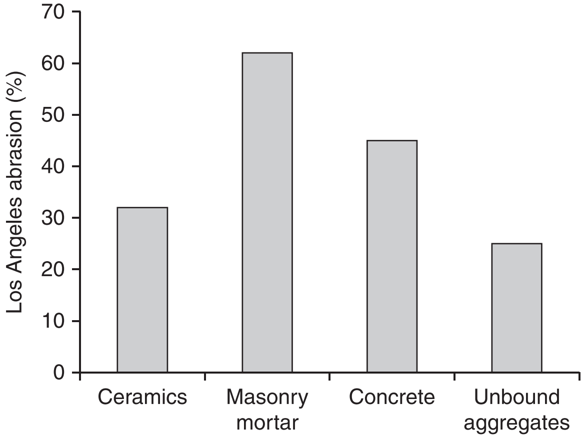
than one-half the nominal size. Flat and elongated particles have a tendency to fracture along their narrow dimension during compaction, and this fracturing can affect the PSD curve (Fig. 14.2). These flat and elongated particles also tend to impede compaction and make it more difficult to achieve a specified material density.

RCA have a greater number of rounded particles than RMA. Typical values on the flakiness index range from 4 to 9% in RCA, 10% to 20% in RMA and 15 to 30% in RMCA. The flakiness index increases with the amount of crushed ceramic particles from bricks or tiles, but the RMA is very well compacted, because elongated particles are broken and the flakiness index decreases during its compaction process. This behaviour is not a limiting property; even in RMCA, the flakiness index before compaction is approximately 40%.

The percentage of crushed particles in an aggregate can be analysed in accordance with the European Standard EN 933-5:1999. RA are obtained by crushing larger pieces of rubble, so they contain a large number of fracture faces. This property results in a high angle of internal friction and consequently a high stress ratio at failure. RA contain more than 70%, and in RCA this value depends on whether the NA used in the manufacture of the original concrete is crushed or rounded aggregate. However, this parameter is not a limiting property.

14.2.5 Fragmentation resistance

The fragmentation resistance of the coarse fraction in RA is determined by the Los Angeles (LA) abrasion coefficient, which can be found in accordance with the European Standard EN 1097-2:1999. The LA abrasion coefficient depends on



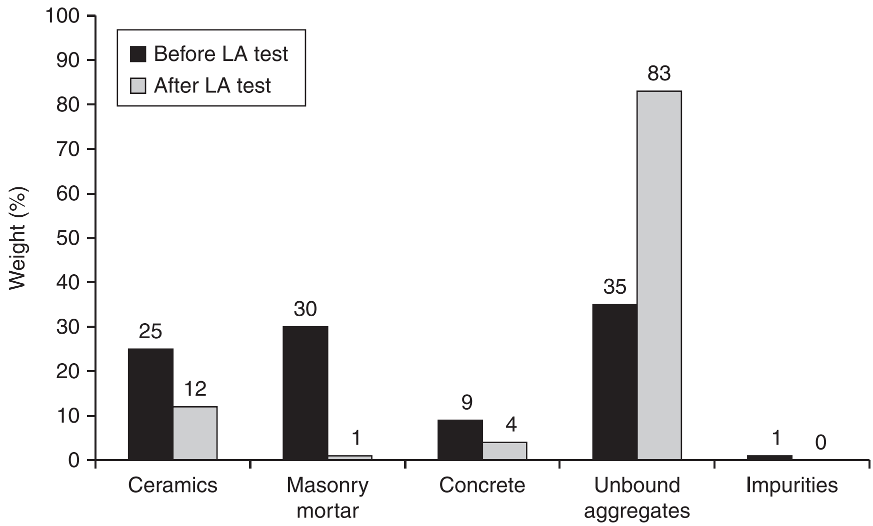
14.3 Los Angeles abrasion coefficients of the different components of RA (10/14 fraction).

the composition of the RA. Figure 14.3 shows the mean LA coefficient values of the main components in RA. The masonry mortar has the highest value, followed by concrete and ceramic particles, because all masonry mortar and mortar attached to concrete particles is powdered during the abrasion test. The unbound aggregate has the lowest value. This graph justifies the LA coefficient values of the different RA varieties.

Most RCA have a LA coefficient value of less than 40%, so most countries allow the use of RCA as a structural layer material in pavements. However, the LA coefficient increases when the percentage of attached mortar is high. Samples with attached mortar contents lower than 44% fulfil the 40% limit, and these RCA have a bulk specific density greater than 2160 Mg/m^3 for the coarse fraction and water absorption of less than 8% (Sánchez and Alaejos, 2009).

Most authors have found that the LA coefficient increases linearly with the percentage of ceramic particles in RMA (Vegas *et al.*, 2011; Barbudo *et al.*, 2012). However, ceramic particles have a lower LA coefficient than concrete particles, so this increase is attributable to other components bound to ceramic particles such as masonry mortar, which has the highest LA coefficient value of all the materials considered.

Figure 14.4 shows how the composition of the 10/14 fraction in RMA is modified after an LA abrasion test. The percentages of ceramic, masonry mortar and concrete particles decrease, while the percentage of unbound aggregates increases significantly. This property has been identified as limiting to RMA in the literature, because most RMA have a LA value equal to or greater than 35 to 40%, which is the limit set by some European Standards for use in pavement structural layers. Furthermore, these values are well above those of the CNA normally used in structural layers, whose LA coefficient values range from 22 to 25%. Table 14.2



14.4 Composition of a mixed recycled aggregate before and after the Los Angeles abrasion test (10/14 fraction).

Table 14.2 Range of LA abrasion coefficient values

	RMCA	RMA	RCA	CNA
Vegas *et al.* (2008)	28–31%[1]	41–43%	37–38%	–
Sánchez and Alaejos (2009)	–	–	35–42%	
Jiménez *et al.* (2011)	–	31–41%	33–34%	20%
Vegas *et al.* (2011)	39–41%	29–39%	35%	
Barbudo *et al.* (2012)	30–43%	31–45%	32–34%	17–26%

Note: [1]100% ceramics.

shows the variation of the LA coefficient in the different RA used for roads. The LA coefficient decreases significantly in compacted materials, because softer particles are broken during the compaction process, as stated above. As we shall see in later sections, this decrease increases the feasibility of using RA with high LA coefficients (Jiménez *et al.*, 2007b).

14.2.6 Plasticity, susceptibility to expansion, sand equivalent and clean coefficient

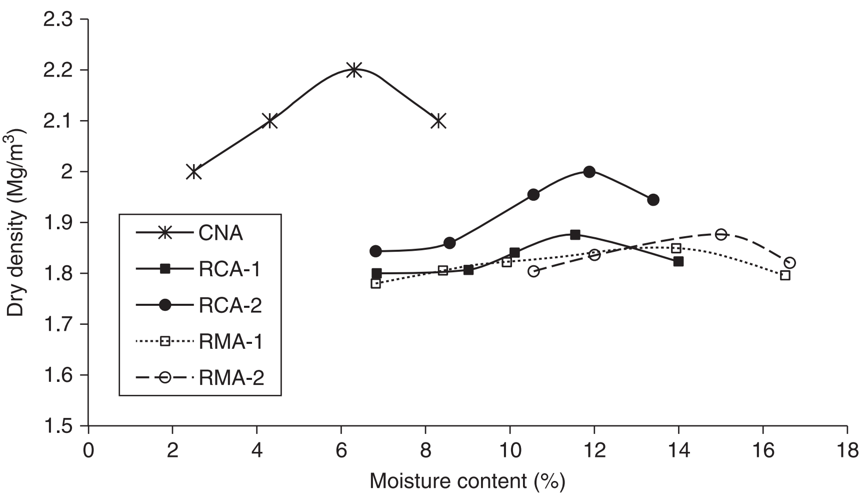
According to ASTM D4318, the components of RA are non-plastic and unsusceptible to expansion, according to the European Standard EN 1744-1:1998. The results of free swelling in an oedometer determined in accordance with UNE 103601:1996 are negligible, and the recycled materials do not collapse after soaking in an oedometer (NLT-254). However, there are no conclusive studies on the effect of sulphates contained in RA on the dimensional stability of pavement structural layers, and this topic is a focus of current research.

The sand equivalent determined in accordance with EN 933-8:2000 meets the specified limit of 35%, with values ranging from 55 to 65%. There is a demonstrated relationship between the percentage of concrete and mortar particles and sand equivalents; the higher the quantity of concrete and mortar, the higher the sand equivalents. The clean coefficient, determined according to EN 146130:2000, rarely exceeds the specified limit of 2%. These properties have not been identified as limiting properties.

14.2.7 Moisture-density relationship (compaction)

The moisture-density relationship of RA is obtained according to either the Standard or Modified Proctor Test (ASTM D698 and ASTM D1557). The peak point of the compaction curve corresponds to the maximum dry density and optimum moisture content. These curves are an indication of the sensitivity of the density to variations in the materials' moisture contents.

Figure 14.5 shows the typical shape of the curves obtained from Modified Proctor Tests for CNA, RCA and RMA, and indicates that CNA have a higher dry density and lower optimum moisture content than both RCA and RMA (Poon and Chan, 2006; Jiménez *et al.*, 2011). The differences in maximum dry density and optimum moisture content are attributable to the particles' different densities and water absorption. CNA and RCA usually have curves that are more sensitive to changes in moisture content than RMA, which have flat curves.



14.5 Modified compaction test curves for crushed limestone and aggregates recycled from CDW.

Determining maximum dry density value and optimum moisture content is more difficult in RA than in CNA, due to the greater heterogeneity of the components of RA as well as the breakage of softer particles during compaction, which modifies the PSD curves and water absorption of the particles. When performing Proctor tests on these materials, it is important that the water be mixed 1 h before compaction due to the high water absorption of RA. The saturation of RA occurs after 30 min of soaking, so 1 h is sufficient for the water to fill the accessible pores of the aggregates. Laboratory tests have indicated a higher dry density in previously saturated RA.

14.2.8 California bearing ratio (bearing capacity)

The California bearing ratio (CBR) is an index test for soils, specifically for the purpose of pavement design. Although it was developed by the California State Highways Department in the 1930s, it is still in common use worldwide (Thom, 2008). In most cases, materials are compacted at their respective optimum moisture contents and subjected to the CBR test under soaking conditions for 4 days. RA have a high bearing capacity, with a CBR soaked value greater than 40%, which is above that required by most standards for base and sub-base materials, so this bearing capacity is not a limiting property (Table 14.3). CNA usually have greater CBR values than RA, and RCA have CBR values greater than those of RMA (Poon and Chan, 2006; Jiménez *et al*, 2011). These differences are most likely due to the lower intrinsic particle strengths of concrete and ceramic. Recent studies have showed than the use of crushed clay brick lowers the CBR value, and sub-bases that use crushed clay brick as a fine aggregate have a lower CBR value than those using RCA as a fine aggregate (Poon *et al.*, 2006).

RA have exhibited an improvement in bearing capacity over time. Their CBR values are greater after 90 days than after 4 days in soaked conditions. This increase in RA CBR values has been attributed to the pozzolanic reactions occurring between the silica and alumina of the ceramic fines and the hydrated portlandite of the cement, or to certain hydraulic properties that remain in the cement of the concrete and mortar (Vegas *et al.*, 2011). RA show negligible swelling after 4 days of soaking, ensuring the stability of structural layers in roads.

Table 14.3 Range of CBR index values

	RMCA	RMA	RCA	CNA
Vegas *et al.* (2008)	64–91%[1]	69–90%	82–107%	–
Jiménez *et al.* (2011)	–	62–94%	97–138%	152%
Vegas *et al.* (2011)	80–130%	90–111%	–	–
Barbudo *et al.* (2012)	45–155%	40–110%	55–109%	50–152%

Note: [1]100% ceramics.

14.3 Chemical characterisation of RAs for road construction

The chemical characterisation of RA depends on the composition of the original material, making them very heterogeneous.

14.3.1 Chemical and mineralogical analysis

The chemical composition of concrete waste depends largely on the nature of the aggregates used in its manufacture (i.e. limestone or siliceous aggregates).

Table 14.4 shows the chemical characterisation of different RA. In all chemical characterisation tests, the coarse and fine fractions must be ground together to guarantee fraction sizes of less than 2 mm. This limit is established to ensure that the entire PSD (not just that of the fine fraction) is used to determine the chemical characteristics of the RA (Vegas *et al.*, 2011).

The main mineral phases identified in RA using X-ray diffraction (XRD) techniques are calcite and quartz. The abundance of theses minerals masks the presence of other mineral phases. Table 14.5 indicates the different mineral phases identified in RA in the literature. Recycled materials used in roads are of special interest, because the different phases of sulphates identified in the mortar attached to concrete particles include gypsum, ettringite and portlandite (Vegas *et al.*, 2011).

Table 14.4 Chemical characterisation of different recycled aggregates

	RCA – siliceous	RCA – limestone	RMA and RMCA
SiO_2	45–60%	4–5%	40–50%
Al_2O_3	15–20%	1–2%	6–8%
Fe_2O_3	2–5%	1–2%	2–4%
CaO	5–7%	52–54%	20–28%
MgO	0.5–1.5%	0.2–0.8%	0–1%

Source: Adapted from CEDEX (2010).

Table 14.5 Mineral phases present in different RA

Mineral phase	RCA	MRA and CRA
Calcite: $CaCO_3$	xxxxx	xxxxx
Quartz: SiO_2	xxx	xxx
Dolomite: $CaMg(CO_3)_2$	x	x
Plagioclase: $(Ca,Na)(Si,Al)_4O_8$	–	x
Feldspar: $(K, Na)AlSi_3O_8$	x	xx
Muscovite: $KAl_2(Si_3Al)O_{10}(OH)_2$	–	xx
Gypsum: $CaSO_4*2H_2O$	x	x
Ettringite: $Ca_6Al_2(SO_4)3(OH)_{12}*26H_2O$	x	x
Portlandite: $Ca(OH)_2$	x	x

14.3.2 Organic matter

The test for determining organic matter is based on an oxidant mass balance such as potassium permanganate (UNE 103204), which is reduced when reacting with other oxidizable species. Organic matter is one of these species, but not the only one present in RA (Vegas *et al.*, 2008). Recent studies have shown that the potassium permanganate test is a non-selective method for determining the presence of organic impurities such as wood, plastic or paper, being the composition test (EN 933-11) that is the most effective method for determining these types of impurities. Other techniques, such as loss on ignition at 500 °C, are not capable of determining the amount of organic matter in RA, because other phases present in RA are affected (Vegas *et al.*, 2011).

The typical quantities of organic matter in RA range from 0.42 to 0.82% in RCA, 0.43 to 0.9 % in MRA and 0.45 to 0.60% in RMCA, all similar values that rarely exceed 1%. This chemical property has not been identified as limiting to road applications in the literature, although the presence of asphalt particles can raise the organic matter content.

14.3.3 Sulphur compounds

The quantity of sulphur compounds is limited in road aggregates to ensure the dimensional stability of the section and avoid potential pathologies in adjacent concrete structures or cement-treated pavement layers. Sulphur compounds can be determined by calculating the total sulphur compound content (%SO_3) in accordance with EN 1744-1:1998, water-soluble sulphates in accordance with UNE 103201 (ASTM C1580) or gypsum content in accordance with NLT-115. In cement-treated applications, it is necessary to determine the acid-soluble sulphate content (%SO_3) in accordance with EN 1744-1:1998.

Typical gypsum contents (NLT-115) range from 0.41 to 1.34% in RMA, while total sulphur contents (EN 1744-1:1998) range from 0.7 to 1.2% in RCA and 1.0 to 2.5% in RMA. Water-soluble sulphate contents (UNE 103201) range from 0.2 to 0.4% in RCA, 0.2 to 1.5% in RMA and 0.2 to 0.9% in RMCA. This chemical property has been identified in the literature as limiting, especially in RMA, where the total sulphur compound content exceeds the specified limit of 1% for pavement structural layer materials. However, these limits have been established for NA and not for recycled materials, so they have been widely discussed in the literature and established as being necessary to the development of specific regulations for the use of RA in road construction (Jiménez *et al.*, 2007a; Vegas *et al.*, 2011). These results indicate that recycled materials should not be placed in contact with concrete structures.

The soluble sulphates in RA come mainly from gypsum, and there is a linear relationship between the total sulphur content measured according to EN 1744-1 and the water-soluble content determined according to UNE 103201. Similarly, there is a linear relationship between the total sulphur content and the gypsum content measured according to NLT-115 (Vegas *et al.*, 2011). Section 10.2 of the

European Standard EN 1744-1 includes a specific method for determining the water-soluble content (%SO_4) of RA by weight, which is the best option for chemically characterising RA in unbound applications. Pre-screening to remove the fine fraction during the RMA recycling process reduces the total sulphur content and improves its quality. By contrast, it is unnecessary to perform such a pre-screening process when recycling concrete wastes (Jiménez *et al.*, 2011).

14.3.4 Soluble salts

The soluble salt content of embankments and sub-bases must be determined in unbound applications to ensure the dimensional stability of the sections. The soluble salt content is determined in accordance with NLT-114 (ASTM D4542), and the soluble salts present in RCA include calcium hydroxide (portlandite), sulphates and carbonates, while the predominant types in RMA are sodium and magnesium sulphates with some carbonates, bicarbonates and sodium chloride. Sulphates represent 20% of the total ionic species, meaning that soluble salts can also be determined from the water-soluble sulphates, and there is a linear relationship between both variables (Vegas *et al.*, 2011).

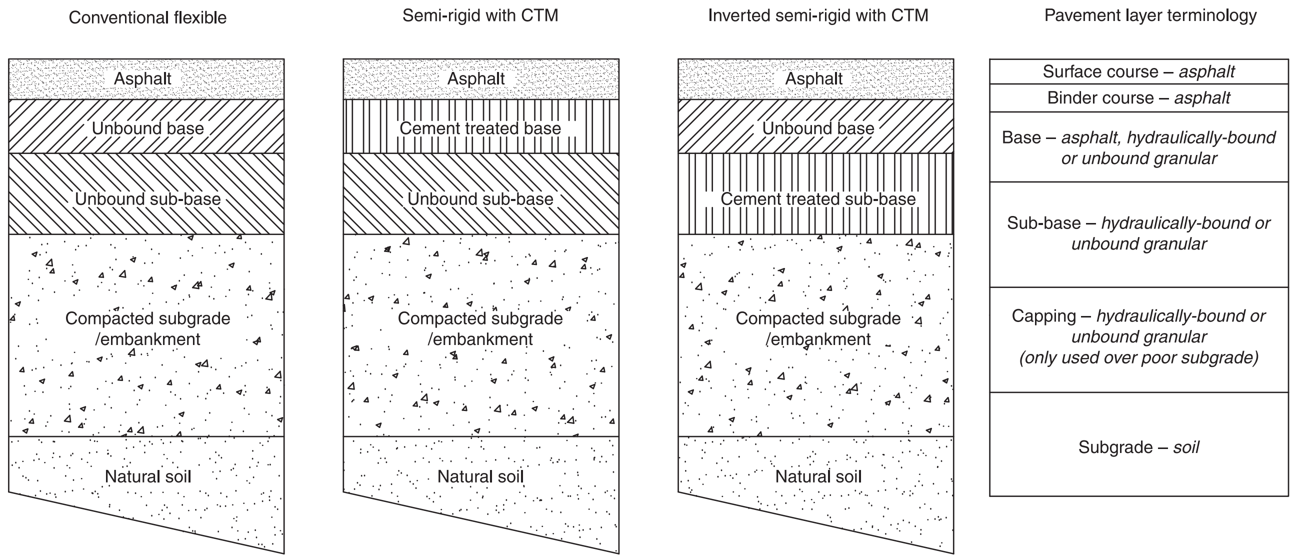
As a reference, the typical soluble salt values range between 0.7 and 2.5% in RCA and 0.9 and 3.3% in RMA. These values greatly exceed the limits set for natural materials, 1% for natural soils used in roadbeds and 0.2% for NA used as sub-bases. This property has been identified as limiting in the literature, although these limits have been widely criticised because it is not unreasonable to require the same limits for RA (Jiménez *et al.*, 2007a; Vegas *et al.*, 2011).

14.4 RAs from construction and demolition waste (CDW) in pavement sections

A pavement must provide a regular surface over which vehicles can circulate in safe and comfortable conditions throughout its lifespan. A variety of pavement systems have been developed to perform this function, and the most common classification method for these systems is according to structural type: flexible, rigid, composite and unpaved.

14.4.1 Typical road pavement sections

A rigid pavement has a Portland cement concrete (PCC) slab as its main structural layer, while a flexible pavement consists entirely of unbound materials and asphalt. A composite pavement combines elements of both flexible and rigid pavement systems and usually consists of an asphaltic concrete surface over PCC or cement-treated materials. This last system is also called a semi-rigid pavement. An unpaved road is simply not paved. Figure 14.6 illustrates the basic components of conventional flexible and semi-rigid pavement sections.



14.6 Basic components of conventional flexible and semi-rigid pavement sections.

The sub-grade is the top surface of a roadbed upon which the pavement structure is constructed, the purpose of which is to provide a platform for the construction of the structural layers and support the pavement without undue deflections. The sub-grade of pavements constructed on cuts is the natural *in-situ* soil at the site, while the sub-grade for pavements constructed on embankment fills is a compacted borrowed material (compacted sub-grade). When it is necessary to treat a sub-grade soil with lime, cement or another hydraulic binder to improve its quality, this layer is called a capping or lower sub-base.

The sub-base is a layer placed on a sub-grade to support a base course. The materials used in sub-bases are high-quality granular materials, although they are of lower quality than the material in the base layer. Sometimes sub-base materials are treated with Portland cement to increase their strength and stiffness. The sub-base is more than just a fill layer. The base layer is placed on a sub-base or sub-grade (if a sub-base is not used) to provide uniform and stable support for the binder and surface courses, providing a significant portion of a flexible pavement's structural capacity. The base materials must be of high quality, and can include limestone or siliceous CNA.

The surface course is the top layer and may be accompanied by other layers (a binder course or base course) designed to accommodate the traffic load. This course is made of asphalt in flexible or semi-rigid pavements and PCC in rigid pavements, and is usually constructed on top of an unbound coarse aggregate base layer, but often placed directly on the prepared sub-grade in low-volume roads.

Replacing natural materials with RA is a viable alternative for roads. RA can be used in roadbeds or structural layers (sub-bases and bases) as unbound or cement-treated materials based on their composition and quality. The recycled materials are not susceptible to any weathering or appreciable physical or chemical changes over their life spans, regardless of their application, and RA do not produce water solutions that can damage concrete structures and cement-treated pavement layers or contaminate soils and water currents. The following sections describe feasible uses of RA in pavement road sections.

14.4.2 RAs as unbound roadbed materials

Unclassified RA and RMCA (Table 14.1) and materials from the pre-screening process in recycling plants can be used as roadbed materials for roads with medium and low average daily traffic (ADT) volumes (<800 heavy vehicles/day). This use generates low value added, but consumes large quantities of recycled materials with low embodied energy. However, it is necessary to limit the amount of impurities in these materials, especially the percentage of gypsum and light particles. The PSD must meet either of the following conditions: more than 70% by weight passing through a 20-mm sieve or more than 35% by weight passing through a 0.080-mm sieve.

Although there are no specific international regulations for such materials, it is recommended that two categories be established for recycled materials being used as roadbeds: Class I for materials used in the highest layers of embankments (lower sub-bases) and Class II for materials used in the cores of embankment fills. The material quality required for each use may be different, and the following recommendations are established as a guideline. The percentage of gypsum particles calculated according to EN 933-11 must be less than 1% for Class I recycled materials and can be as much as 2% for Class II materials. The percentage of light particles and impurities must be less than 1% in Class I and 2% in Class II materials.

Recycled materials containing a percentage of gypsum greater than 2% are often linked to non-selective demolition techniques and simple treatment processes in recycling plants. The best use for these materials with low embodied energy is in embankment fill construction for low-ADT volume or unpaved rural roads, provided special precautions are taken and laboratory studies are performed to ensure the dimensional stability of the section. For aesthetic reasons, the percentage of light particles must be limited to 0.5% in recycled materials used as surface courses in unpaved rural roads. Recycled materials containing more than 5% gypsum particles should never be used in roads. Organic matter is not a limiting property, so the limits established for roadbed natural soils are 1% in Class I recycled materials and 2% in Class II materials.

In terms of the water sulphate content (%SO_4) calculated according to section 10.2 of the European Standard EN 1744-1, some recommendations establish a limit of 0.7% in Class I recycled materials and 1% in Class II materials. These limits are undergoing revision by researchers. Recent studies have shown that a soluble salts content of approximately 4% does not affect the dimensional stability of sections (Vegas *et al.*, 2011, Jiménez *et al.*, 2012b). Because the percentage of gypsum particles is limited, the soluble salts content can be as much as 2% in Class I recycled materials and 3% in Class II materials, without endangering the dimensional stability of the section.

Because the recycled materials used in roadbeds may contain large quantities of excavated soils, the plasticity, free-swelling and collapse in an oedometer and the CBR index of these materials must be determined, despite the fact that these parameters have not been identified as limiting properties in RA. The limits for these properties are the same as for natural soils.

14.4.3 RAs as unbound sub-base materials

RCA and RMA with ceramic contents of up to 30% by weight (Table 14.1) can be used as unbound sub-base materials in roads with ADT volumes of less than 800 heavy vehicles/day, but always under a granular base. Selected RCA must be used in the higher structural layers. Based on the current state of knowledge, the use of RMA with more than 20% of ceramic particles in contact with the surface course (asphalt or concrete) is not recommended. Although there are no specific international regulations on this application of RA, the following recommendations can serve as a guide.

The percentages of gypsum and light particles must be less than 1% and 0.5%, respectively, both calculated in accordance with EN 933-11. The RA must be free of clay lumps or foreign matter that may affect the durability of the section, and the sand equivalent must be greater than 30. For RA to be useful as granular materials for sub-bases, their PSD must fall within the grading envelopes specified in Table 14.6. Most RA meet this condition, although it is advisable to check the material's susceptibility to compaction and verify that it meets the condition after the compaction process. The LA abrasion coefficient must be less than 45% for roads with an ADT volume between 200 and 800 heavy vehicles/day, but this limit may be increased to 50% for minor traffic intensities (GIASA, 2010). Most RMA meet this limit.

Using recycled materials in sub-bases containing excavation soils is not recommended, so the RA must be non-plastic for this application. The organic matter determined with the potassium permanganate method must be less than 1%, although this limit may increase if the RA contain significant quantities of asphalt (GIASA, 2010). Recent studies have shown that the maximum total sulphur content (%SO_3) calculated according to EN 1744-1 can be as much as 1.3%, without affecting the dimensional stability of the section (Vegas *et al.*, 2011). No reliable studies exist that set a limit on the allowable water-soluble sulphates calculated in accordance with Section 10.2 of the EN 1744-1.

14.4.4 RAs as unbound base materials

RCA and RMA with ceramic contents of up to 20% by weight (Table 14.1) can be used as unbound base materials in roads with an ADT volume of less than 800

Table 14.6 Grading envelopes of materials to be used as sub-bases and bases

Recycled material		Particle size distribution (mm) based on EN 933-2/percent passing (%)												
		56	45	40	32	22	16	11.2	8	4	2	0.5	0.25	0.063
RA 40/0	UL	100	–	99	–	87	–	–	63	46	35	23	18	9
	LL	100	–	75	–	58	–	–	35	22	15	7	4	0
RA 32/0 (F)	UL	–	100	–	99	–	84	–	68	51	40	26	20	11
	LL	–	100	–	75	–	56	–	40	27	20	7	4	0
RA 22/0 (F)	UL	–	–	–	100	99	–	82	75	61	50	32	24	11
	LL	–	–	–	100	75	–	52	45	32	25	10	5	0

Note: UL: Upper limit; LL: Lower limit. Adapted from GIASA (2010).

heavy vehicles/day. Materials used for this application must have a CE marking, although there are again no specific international regulations on this topic. The following recommendations can be made as a guide. The percentage of gypsum particles must be less than 0.5% in RCA and 1% in RMA and the total amount of impurities, including light particles, must be less than 1% in both materials. All are measured in accordance with the EN 933-11.

The recycled materials must contain no clay lumps, foreign matter or excavated soils, and the clean coefficient must be less than 2% and the sand equivalent greater than 35. The materials must also be non-plastic. The flakiness index must be less than 35 and the percentage of crushed particles must be greater than 75% for roads with an ADT volume between 200 and 800 heavy vehicles/day and greater than 50% for roads with minor traffic intensities. These properties are not limiting for RCA and RMA that contain less than 20% ceramic particles.

The LA abrasion coefficient of these aggregates must be less than 45%, but this value can be exceeded by RCA produced from low-strength concrete with large quantities of attached mortar or by RMA with large quantities of masonry mortar. If this value is exceeded, the LA test can be performed again on compacted material in the laboratory using the modified proctor energy test. The coefficient values improve significantly with this test (Jiménez *et al.*, 2007b), and the LA coefficient of compacted materials must then be less than 40%. For RCA and MRA to be useful as granular materials in bases, their PSD must fall within the grading envelopes set in Tables 14.6 or 14.7. It is advisable to check the aggregate's susceptibility to compaction and verify that the materials meet this condition after the compaction process (GIASA, 2010). The total sulphur content (%SO_3) calculated according to EN 1744-1 may be as much as 1.3%, and no reliable studies exist that set a limit on the allowable water-soluble sulphates calculated in accordance with Section 10.2 of the EN 1744-1.

Table 14.7 Grading envelopes of materials to be used in bases

Recycled material		Particle size distribution (mm) based on EN 933-2/percent passing (%)												
		56	45	40	32	22	16	11.2	8	4	2	0.5	0.25	0.063
RA 32/0 (G)	UL	–	100	–	99	–	84	–	63	45	32	21	16	9
	LL	–	100	–	75	–	57	–	40	26	15	7	4	0
RA 22/0 (G)	UL	–	100	–	–	99	–	82	73	54	40	24	18	9
	LL	–	100	–	–	75	–	54	45	31	20	9	5	0
RA 22/0 (D)	UL	–	–	–	100	99	–	70	58	37	15	6	4	2
	LL	–	–	–	100	75	–	42	30	14	0	0	0	0

Note: UL: Upper limit; LL: Lower limit. Adapted from GIASA (2010).

14.4.5 RAs as cement-treated materials

Pavement materials that use cement as a binder are known as cement-treated materials (CTM). These materials are dosed to achieve a dry mix consistency and are suitable for roller compaction. CTM have been widely used as a semi-rigid base course for both flexible and rigid pavements, and must provide most of the pavement strength in semi-rigid pavements. RA have been used as CTM in recent years and this application is one of the best ways to recycle these materials, because it does not have a complex treatment process in recycling plants and gives high added value to the RA.

The mechanical properties of CTM are influenced by a number of variables, primarily the cement content, cement type, moisture content, curing time, degree of compaction and aggregate type (Xuan *et al*., 2012a). When RA are used, the mechanical properties are also determined by the ratio of crushed masonry content to crushed concrete content (Xuan *et al.*, 2012b). Dosage studies in the laboratory must be performed prior to use. Traditionally, the CTM mix design is based on the optimum moisture content, maximum dry density, workability and confined compressive strength.

RA can be used in two ways: Option I, where all materials are recycled, and Option II, where only the coarse fraction is recycled and the fine fraction is composed of natural sand. RCA and RMA with a ceramic content of up to 20% can be used in Option I. As with the other uses of RA in roads, there are no specific international regulations, but the following recommendations can be made. The percentage of gypsum particles in a composition test (EN 933-11) must be less than 0.5% by weight in RCA and 1% in RMA, while the percentage of impurities, including gypsum and light particles, must be less than 1% in both materials. The organic matter determined by the potassium permanganate method must be less than 1%. The presence of organic matter may delay the setting reaction, but this is usually not a limiting property.

Sulphates can react with hydrated cement in the presence of water, resulting in ettringite. This compound occupies a much larger volume than the initial material, causing expansion and dimensional instability of the section. No reliable studies exist that set a limit to the allowable water-soluble sulphates (%SO_4) determined according to Section 10.2 of EN 1744-1, so the existing recommendations are very limiting, setting a value of 0.2%. The total sulphur content and acid-soluble content (%SO_3) determined according to EN 1744-1 can be as much as 1%. An alternative to this limit is the use of sulphur-resistant cements (SR). It is also necessary to study the alkali-silica, alkali-silicate and alkali-carbonate reactions in the material.

As when using RA in the manufacture of concrete and mortars, greater water absorption shortens the mix's workability, so it may be advisable to use additives that delay setting. For RCA and MRA to be useful as CTM in Option I, their PSD must fall within the grading envelopes specified in Table 14.8. The cement content

Table 14.8 Grading envelopes of materials used as cement-treated materials, Option I

Recycled material		Particle size distribution (mm) based on EN 933-2/percent passing (%)									
		50	40	25	20	12.5	8	4	2	0.5	0.063
CTRAM-40	UL	100	100	100	100	100	89	65	52	37	20
	LL	100	80	67	62	53	45	30	17	5	2
CTRAM-20	UL	–	–	100	100	100	100	100	94	65	35
	LL	–	–	100	92	76	63	48	36	18	2

Note: CTRAM: cement-treated recycled aggregate material.

should be the minimum that can achieve an unconfined compressive strength between 2.5 and 4.5 MPa at a curing time of 7 days. The cement content must be no less than 3% and the workability no less than 180 min, although execution by the band technique can extend this time to 240 min.

Only the coarse fraction of RCA is recommended for use in Option II. There are again no specific international regulations for its use, but the following recommendations can be made for the coarse recycled fraction. The percentage of gypsum particles in a composition test (EN 933-11) must be less than 0.5% by weight and the percentage of total impurities less than 1%. The organic matter determined by the potassium permanganate method must be less than 1% and the total sulphur content and acid-soluble content (%SO_3) determined according to EN 1744-1 must be less than 1%. The coarse fractions of RA show no potential reactivity.

In terms of physico-mechanical properties, the flakiness index must be less than 35, the percentage of crushed particles greater than 50% and the LA abrasion coefficient less than 40. The fine fraction is composed of crushed stone or gravel. For coarse RCA to be useful as CTM in Option II, their PSD must fall within the grading envelopes specified in Table 14.9. The cement content should be the

Table 14.9 Grading envelopes of materials used as cement-treated materials, Option II

Recycled material		Particle size distribution (mm) based on EN 933-2/percent passing (%)									
		45	32	22	16	11.2	8	4	2	0.5	0.063
CTRCAM-32	UL	100	99	–	83	–	63	48	37	21	7
	LL	100	75	–	53	–	38	25	16	6	1
CTRCAM-22	UL	–	100	99	–	79	68	51	39	22	7
	LL	–	100	75	–	52	44	28	19	7	1

Note: CTRCAM: cement-treated recycled concrete aggregate material.

minimum that can achieve an unconfined compressive strength between 4.5 and 7 MPa at a curing time of 7 days. The cement content must be no less than 3.5% and the workability limits the same as for Option I.

14.5 Assessing the use of RAs in practice

The construction of experimental sections is an important step in developing new technical specifications and increasing the knowledge of construction techniques using RA. The number of publications on experimental sections built with recycled materials is limited and information on this application is dispersed. However, recent studies have shown that RA are a viable alternative to NA in road construction (Jiménez *et al.*, 2012a,b; Agrela *et al.*, 2012; Herrador *et al.*, 2012).

14.5.1 Practical issues

RA must come from recycling plants that are authorised for the management of CDW and have a certificate issued by the manufacturer to ensure their physico-mechanical and chemical properties. Aggregates used as sub-base and base materials must have a CE marking. RA can also come from authorised mobile plants associated with the construction work. This option reduces transportation costs and CO_2 emissions, but it is more difficult to control the quality of RA coming from mobile plants. Before commencing work, the origin of the RA must be verified through documentary checks, visits to the recycling plants and testing on material stockpiles for quality control.

Construction procedures for RA are similar to those used for NA, except for the moisture content prior to compaction. RA have greater water absorption than NA, so pre-wetting is essential to achieving proper compaction. The RA should be wetted on the stockpiles or in several stages after work has commenced, maintaining a minimum interval of 1 h so that the water fills the particle pores.

14.5.2 Bearing capacity in field conditions

The material's bearing capacity in the field is measured by Young's modulus, Ev2, obtained during the second cycle of the static plate load test (PLT). The results of a PLT are presented as curves of applied contact pressure versus settlement, and Young's modulus, Ev1, can be obtained from the slope of the secant through the points corresponding to $0.3\,\sigma_{max}$ and $0.7\,\sigma_{min}$, where σ_{max} is the maximum pressure applied. The Ev2 value depends on the structural layer and the category of heavy vehicles on the road (Table 14.10). As a general rule, the limit to the ratio Ev2/Ev1 is 2.2.

When using RA as a structural layer material, it is easy to obtain the Ev2 values from Table 14.10. Young's moduli in sections built with RA are similar to those in sections built with CNA (Jiménez *et al.*, 2012a), but unlike CAN, RA bearing

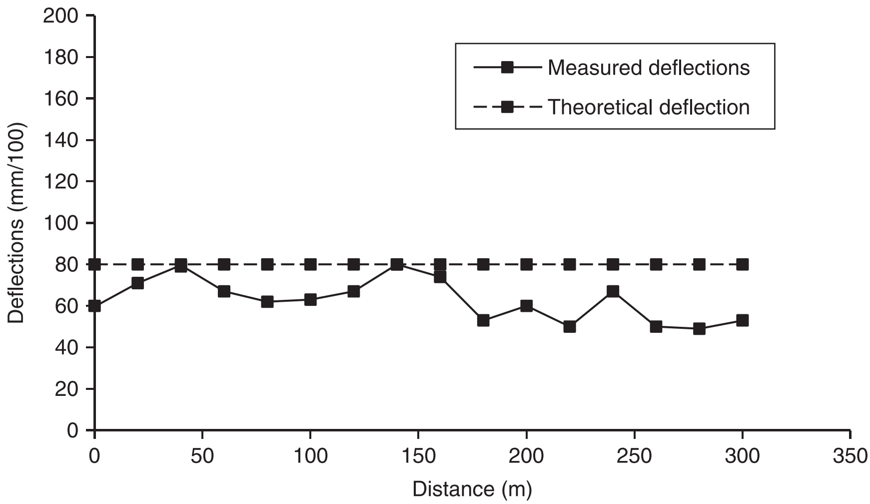
Table 14.10 Minimum value of Ev2 based on traffic category

Structural layer	ADT (heavy vehicles/day)		
	200–800 (T2)	50–200 (T3)	<50 (T4)
Unbound sub-base	100 MPa	80 MPa	60 MPa
Unbound base	150 MPa	100 MPa	80 MPa

capacities increase over time (Jiménez *et al.*, 2012b). The Ev2/Ev1 ratio is often greater than 2.2 in both RA and CNA, so an Ev2 value or Ev1 value greater than 60% of the value required for Ev2 cannot be considered as this limitation.

Surface deflections can be measured in accordance with ASTM D4694 (2003) to assess the strength of a section. The use of pavement deflection as an indicator of a pavement's structural performance is generally accepted. Figure 14.7 shows a falling weight deflectometer (FWD) mounted on a trailer typically used in field testing. With the FWD, an impulse load is applied to the road surface by dropping a steel bearing plate with a diameter of 450 mm. The drop must ensure a dynamic load of 68.6 kN, and the surface deflection is measured by 7 geophones located directly under the loaded area and at several off-set positions, the maximum of which is 2.5 m from the centre. Temperature can influence this measurement on asphalt layers.

14.7 Falling weight deflectometer mounted on a trailer.



14.8 Measured and theoretical deflections on an unbound base in an experimental section with RA.

If RA are properly set in place, the deflection measurements are usually lower than their theoretical counterparts, so the unit work can be accepted. Figure 14.8 shows the results of the measured and theoretical deflections on an unbound base in an experimental section built using 1 m of RMA as unbound sub-base and 0.3 m of RCA as unbound base. The standard deviation/mean (SD/M) ratio must be less than 0.2 to indicate that the section is uniform.

The dynamic equivalent modulus can be back-calculated according to the function proposed by Brown (1996):

$$Ev = \frac{2pa(1-\upsilon^2)}{d} \qquad [14.1]$$

where p is the contact pressure below the plate (0.431 MPa), α is the plate radius (225 mm), υ is Poisson's ratio (0.35 in unbound granular materials and 0.25 in granular treated materials), and d is the measured plate deflection in mm.

14.5.3 Road roughness measurement

The roughness of a road surface strongly influences its operation costs and is closely related to the road's quality and condition. An analysis of the evolution of a road's roughness is necessary to evaluate the experimental section built using RA over time, and the International Roughness Index (IRI) is the index most commonly used for this purpose worldwide.

14.9 IRI equipment.

The IRI is measured using a commercial device that is mounted on a vehicle equipped with an accelerometer on the rear axle to measure the vehicle's movement and two laser sensors on the front axle to measure its displacement (Fig. 14.9). The data are automatically recorded on a computer every 2.5 cm and the IRI values are subsequently obtained from the measured longitudinal profiles. The IRI is calculated using a quarter-car vehicle math model (ASTM E867-06: 2006), which has a cumulative response. The unit of the IRI is given as a slope (dm/Hm).

The experimental sections built using RCA show slightly higher IRI values than those for sections built with CNA. However, the IRI can be lower if RMA are used, which could be related to the modification in the PSD curve that occurs during the compaction process of RA due to an increase in the percentage of fines. The sections built with CNA deteriorate more rapidly than those built with RA, even in unpaved rural roads (Jiménez *et al.*, 2012a).

14.6 Environmental performance

If they are not separated at the source, CDW can contain small amounts of hazardous wastes, such as adhesives, lead-based paints (LBP), phenols, formaldehyde resins, polychlorinated biphenyls (PCB), polycyclic aromatic hydrocarbons (PAH) and others, each of which can pose particular risks to the environment. When RA are used in roads, rainwater or infiltration can leach these harmful elements (including organic compounds, anions and metals). Such leaching represents a potential risk to the environment (Chowdhury *et al.*, 2010).

The environmental performance of a material can be assessed using the European Standard EN 12457-3:2002 leaching test. As in the chemical characterisation tests, the coarse and fine fractions are ground together to guarantee fraction sizes of less than 4 mm. This standard consists of a 2-step batch leaching test on 175-g solutions of dry material samples at liquid-to-solid ratios (L/S) of 2 and 10 l/kg, respectively. Different L/S represent different amounts of water in contact with the product during the test and can simulate short- and long-term exposure scenarios. The quantities of several major and trace elements are determined in the laboratory using inductively coupled plasma mass spectrometry (ICP-MS). The leached concentration (mg/kg) for L/S = 2 and L/S = 10 are compared with the waste acceptance criteria for landfilling, which are stated in Annex 2 of the 2003/33/CE Council Decision. The recycled materials are then classified as inert, non-hazardous or hazardous (Table 14.11).

Most RA are classified as inert materials. Although the literature has identified some elements that exceed the limits of the specifications for an inert material, such as chromium (Cr), or the sulphate ions detected in leachates and caused mainly by the high amount of gypsum in some CDW. The RA containing these materials have been classified as non-hazardous (Galvín *et al.*, 2012).

Table 14.11 Acceptance criteria

Parameter	Leached concentrations (mg/kg) depending on landfill class					
	Inert		Non-hazardous		Hazardous	
	L/S = 2	L/S = 10	L/S = 2	L/S = 10	L/S = 2	L/S = 10
Cr	0.2	0.5	4	10	25	70
Ni	0.2	0.4	5	10	20	40
Cu	0.9	2	25	50	50	100
Zn	2	4	25	50	90	200
As	0.1	0.5	0.4	2	6	25
Se	0.06	0.1	0.3	0.5	4	7
Mo	0.3	0.5	5	10	20	30
Cd	0.03	0.04	0.6	1	3	5
Sb	0.02	0.06	0.2	0.7	2	5
Ba	7	20	30	100	100	300
Hg	0.003	0.01	0.05	0.2	0.5	2
Pb	0.2	0.5	5	10	25	50
Cl^- (mg/l)	550	800	10 000	15 000	17 000	25 000
F^- (mg/l)	4	10	60	150	200	500
$SO_4^=$ (mg/l)	560	1000	10 000	20 000	25 000	50 000

Source: WAC; ER Council Decision 2003/33/EC.

The translation from an L/S scale to a timescale can be assumed using the following equation:

$$t = \frac{L}{S} \cdot d \cdot \frac{H}{I} \qquad [14.2]$$

where *t* is the time required to reach the concentration simulated in the laboratory, *d* is the bulk density in kg/m3, *H* is the layer thickness in m, and *I* is the infiltration rate in mm/y. Other methods have been widely applied in the literature, including the maximum expected release measured according to the NEN 7341 availability test developed by the Dutch procedure and the cumulative release for different exposure conditions measured using the NEN 7343 percolation test. Therefore, the RA used for roads do not pose a greater leaching risk than NA, except for sulphates, which should be carefully analysed (Jiménez *et al.*, 2012a,b; Engelsen *et al.*, 2012).

14.7 Conclusions and future trends

RA are a viable alternative to NA in roads with medium and low traffic intensities. Depending on their composition and quality, RA can be used as roadbed, sub-base or base materials as well as unbound or cement-treated materials. Their physico-mechanical and chemical properties differ from those of NA, requiring the development of specific regulations that favour the use of these materials and allow an increased recycled material rate in road construction, which is still very low in some developed countries.

Fragmentation resistance, sulphur compounds and soluble salts have been identified as limiting properties in the literature, but the test methods and limits set for NA have been declared inappropriate for RA. Researchers are working to establish the effects of the total sulphur compounds and soluble salts content, mainly from gypsum particles, on the dimensional stability and bearing capacity of sections in the short and long term. The goal of this research is to set appropriate limits for water-soluble sulphates in unbound applications and acid-soluble sulphates in cement-treated recycled materials that ensure the dimensional stability and durability of sections.

Susceptibility to compaction and the effect of RA components on fragmentation resistance are additional focuses of current research activity. The stress-strain behaviour of unbound and bound RA must be known for pavement design, especially their elastic moduli and Poisson's ratios. These values are important indices used in linear-elastic multi-layer pavement design systems, but few studies on these parameters exist. The construction of experimental sections is an important step in developing new technical specifications and increasing knowledge of construction techniques using RA. There remains a lack of such studies in the literature.

The best way to make use of lower-quality materials with low embodied energies is to employ them in the construction of rural or low traffic intensity roads, while selected recycled materials can be used as sub-base and base structural layers in medium traffic intensity roads. In general, the leachates from RA are not an environmental risk, and RA are classified as inert or non-hazardous. The best use of selected types of RA is in cement-treated materials, because they do not require a complex treatment process and give high added value to the CDW. Other uses, such as the inclusion of RA in compacted concrete or grave-emulsion as a base, are currently being studied.

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15
Recycled aggregates (RAs) for asphalt materials

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Abstract: In this chapter, we gather the results of the most innovative studies about recycled aggregates (RA) from construction and demolition waste (CDW), in an attempt to better understand the physical and mechanical performance of these materials, which are more economical, sustainable and environmentally friendlier, as an integral part of bituminous mixtures.

Key words: construction and demolition waste (CDW), recycled aggregates (RA), asphalt mixtures properties.

15.1 Introduction

Around the fifth millennium BC (in the late Neolithic period), one of the most important technological advances in history occurred: the wheel. Although it is widely believed that Mesopotamians initially conceived it as a potter's wheel, its use in transportation developed rapidly, as the need to supply food to the cities of these first civilizations grew and trading developed. Increased transport traffic between cities led to the need to build infrastructure. The engineering and construction of tracks and roads began.

Since then, during the past 7000 years, different civilizations have contributed to the development of roads and pavement, as they are known today. These developments included the solid Roman stone carriageways and the sophisticated road network of the Islamic Empire, which was already built with asphalt back in 700 BC; as well as the first modern British road systems of the 18th and 19th centuries, which led to the first toll roads, or the acclaimed German 'Autobahn' from the 1930s, precursor of modern freeways and high-performance roads of our times. Looking back into the past, it can be observed that, over and over again, different people have tried to build more solid and durable roads by more efficient and economical means. With the growing concerns about climate change, a new factor has recently become unavoidable: sustainability.

One means of achieving more sustainable pavements followed by current engineers is in the substitution of natural aggregates (NA) for recycled aggregates (RA), obtained, for example, from the construction and demolition of former buildings and infrastructures. Using materials which might otherwise be disposed of in garbage dumps and reducing the need to open new quarries, will help to ensure more sustainable pavements and combat global warming. Unfortunately, construction and demolition waste (CDW) from concrete structures can be contaminated with materials such as brick, gypsum, glass, bitumen, wood and rubber, among others (Ravindrarajah, 1996; Dumitru *et al*., 1996; Sagoe-Crentsil *et al*., 1996; Shayan *et al*., 1997; Arm, 2001; Gómez-Soberón, 2002; Chen *et al*., 2011a; Zaharieva *et al*., 2003). Therefore, results drawn from research focused on their use have led to scattered, unstable and sometimes contradictory conclusions.

Little research has been conducted concerning the mechanical performance, durability and economic impact of these new materials and, what is even more important, concerning their performance with regard to road safety. For these reasons, their application within bituminous mixtures constitutes one of the most important leading lines of investigation in highway engineering, although their use is still one step behind the application to other construction materials such as concrete.

This chapter aims to present the latest findings on the physico-mechanical performance of bituminous mixtures containing these types of aggregates, as well as outlining predictions about the effects of their multiple applications. As may be observed, many steps have already been taken and many satisfactory results have been obtained. However, this is just the starting point and, perhaps, as in the case of previous societies, the beginning of a new era. It cannot be denied that there is still a long way to go.

15.2 Volumetric properties

A sample prepared in the laboratory can be analysed to determine its probable performance inside the structure of a given pavement. Basic preliminary analysis, aimed at simply characterizing the materials in order to measure the mixtures afterwards, are mainly based on characteristics such as the specific gravity, amount of air voids, voids in the aggregates (filled either with air or bitumen) and absorption. The addition of RA to bituminous mixtures especially has an impact on two of them: the specific gravity and the absorption.

15.2.1 Air voids

Voids are the spaces inside a bituminous mixture that, once it has been manufactured and compacted, are not occupied by any aggregate and may be filled with air or bitumen. In general terms, RA from CDW are more porous than NA and they also show a high rate of binder absorption into the grains, thus leading to reductions in the thickness of the film that coats the aggregate. Thus, when they are added to bituminous mixtures, the volume of voids filled with air (Percent Voids Total Mix – VTM) increases, whereas the volume of voids filled with asphalt (Voids Filled with Asphalt – VFA) and of spaces between the aggregate particles (Voids in the Mineral Aggregate – VMA) tend to decrease

(Paranavithana and Mohajerani, 2006; Pérez *et al*., 2010a; Shen and Du, 2005; Wong *et al*., 2007).

15.2.2 Specific gravity

The specific gravity of a compacted mixture is the ratio between the weight and a given specific volume. Low values are generally associated with low-quality aggregates, that may lead to poor performance once they have been added to a given pavement. Nevertheless, the addition of recycled materials to bituminous mixtures also leads, as a general rule, to slightly lower specific gravities when compared with those mixtures formed solely by NA. According to Poon and Chan (2006), NA show slightly higher specific gravities than recycled concrete aggregates and these, in turn, show higher specific gravities than those of clay-based aggregates.

Gómez-Soberón (2002) established that the specific gravity of RA is between 9 and 14% lower than that of reference NA. Li (2004) also gave evidence of the great importance of the mortar adhered to recycled concrete aggregates, since the author considers that it may be the cause of many differences in performance between NA and RA. For example, the bulk dry specific gravities of the RA were found to be between 1.290 and 1.470, and the bulk Saturated Surface Dry Basis (SSD) specific gravities were between 2.310 and 2.620; placing them in between the typical densities of a natural rock aggregate and a lightweight aggregate. Moreover, there are many additional publications, such as Huang *et al*. (2002), Paranavithana and Mohajerani (2006), Pérez *et al*. (2007), Melbouci (2009), Mills-Beale and You (2010) and Gokce *et al*. (2011), that agree with the above-mentioned trend.

15.2.3 Absorption rate

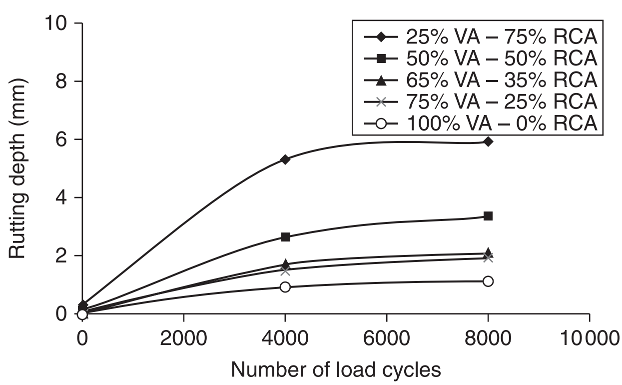
RA tend to be, as well as lightweight, very porous and absorbent, basically due to the presence of a high content of materials such as cement mortar, brick and, sometimes, wood particles. If a regular absorption rate is normally lower than 1.5% for natural quarry aggregates, some authors, such as those mentioned above, found values of 10.3 and 30.9% for fine concrete and clay-based aggregates, respectively (Poon and Chan, 2006); between 4 and 9.5% (Li, 2004); up to 8.8% (Shen and Du, 2005); and 8.16% (Gómez-Soberón, 2002), etc. This phenomenon must be taken into serious consideration when working with mixtures that require the addition of great amounts of water or, rather, with those requiring a precise water dosage, as is the case in cold bituminous mixtures made with asphalt emulsions.

15.3 Rutting

The rutting phenomenon may be motivated by a low shear strength of the bituminous mixtures used on the surfaces of flexible pavements; although the resistance to permanent deformations (PD) is also commonly attributed to the content of binder, the degree of compaction and to factors associated with the aggregates such as angularity, particle size distribution, grain-shape and roughness. However, in some cases, the substitution of NA for RA produces almost unpredictable effects and so a very controlled application is required. This is the case of RA from CDW, which contain fractions of concrete, brick, plaster, glass, bitumen, rubber and other materials. According to the wheel tracking tests carried out by Shen and Du (2005), hot mixtures containing 50% of NA and 50% of RA showed a minimum PD, compared with all the results obtained from other mixtures containing different proportions or even 100% of NA, when conducted at 25 °C; but a maximum PD when tested at 60 °C. Therefore, they reported and classified such aggregate substitutions as unsuitable.

Pérez *et al*. (2012), after having conducted the same experiment on hot asphalt mixtures with RA substitution rates of 0, 20, 40 and 60% containing cement or limestone fillers, concluded that all of them showed a good performance regarding PD, since in the 8 cases studied, the PD-rate (average deformation velocity of 3 specimens at the interval between 105 and 120 min from the start in a deformation vs. time graph) was lower than the maximum established by the Spanish standard PG-3 (Direccion General de Carreteras (2004)). However, other research has shown adverse effects, such as the study conducted by Mills-Beale and You (2010), who, when using an asphalt pavement analyser at a temperature of 52 °C, concluded that, for each number of load cycles, as the content of recycled concrete in the mixture increases, the PD also increases (Fig. 15.1).

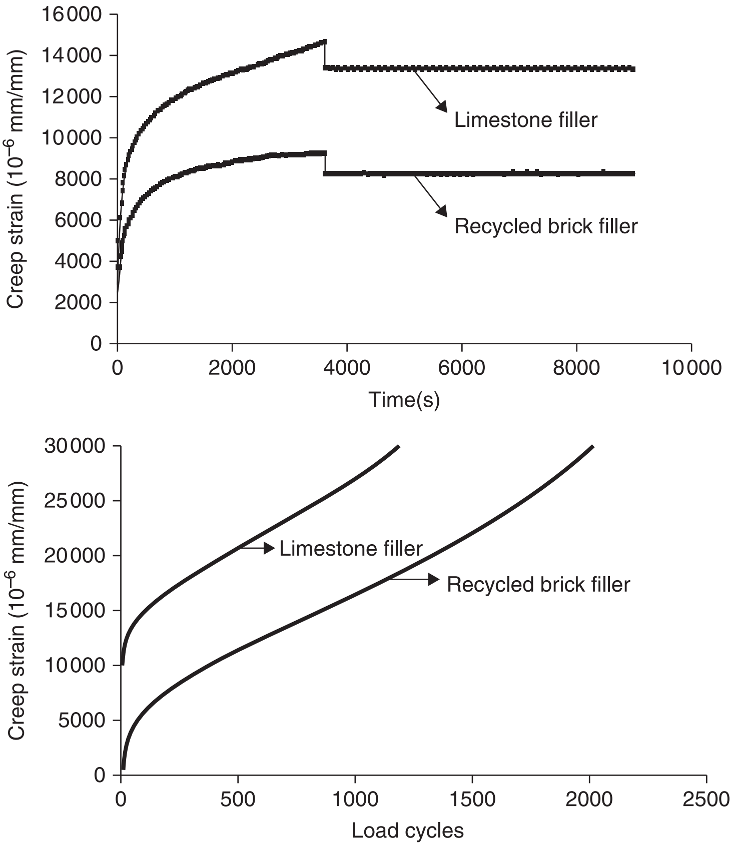
However, other studies have concluded that mixtures containing coarse aggregates from CDW showed a similar performance to control mixes containing exclusively NA (Paranavithana and Mohajerani, 2006); which could be considered as a favourable result. This trend has been reaffirmed by Wong *et al*. (2007),



15.1 Permanent deformation for mixtures with different content of recycled concrete aggregate (RCA) and virgin aggregate (VA), ranging from 0% (○) up to 75% (◆) (Mills–Beale and You, 2010).

substituting the fines and the natural filler for recycled concrete in small proportions (6% of the total). They have proved that the results (in accordance with the Australian standard AS 2891.12.1, 1995) were considerably better in the case of greater substitutions (up to 45%), since the number of load cycles had to be increased by up to 2 to 3 times to achieve the same deformation. However, it has also been found that greater porosity and high absorption substitution materials, such as the brick powder used as filler, provide the mixtures with better performance at high temperatures and lower PD (Chen *et al.*, 2011a). In this study, both static and dynamic tests were conducted at 60 °C and, in all these tests, the deformations of mixtures containing brick powder were significantly lower than those shown by control mixes (containing 100% of limestone) (Fig. 15.2).

Finally, there is evidence of studies conducted on hot mixtures containing different substitution materials such as Silestone® decorative stone debris which,



15.2 Static (upper) and dynamic (lower) creep strain of asphalt mixtures containing limestone filler and filler from brick powder (Chen *et al.*, 2011a).

despite their grain-shape being less favourable than that of the NA used in control samples (flakiness index of 18% instead of 5%), showed very similar results in wheel tracking tests (Rubio *et al*., 2011).

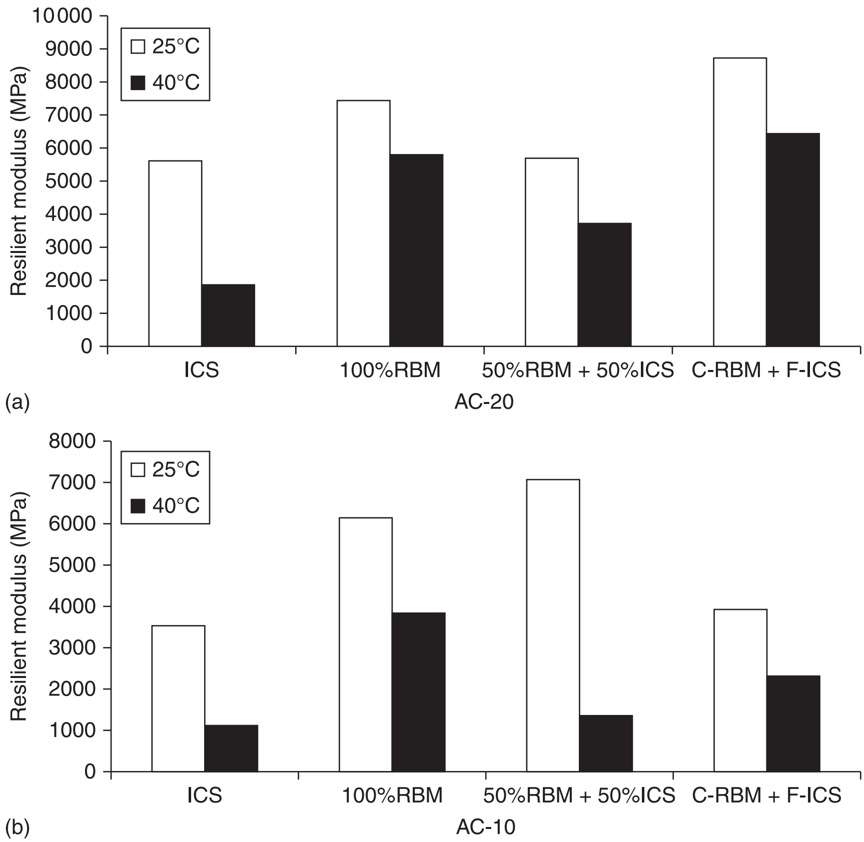
15.4 Stiffness

The stiffness of bituminous mixtures is a key factor to analyse and design flexible pavements, since it is directly related to the capacity of the material to distribute the loads and also serves as a synthetic indicator of the structural properties of the mixtures (Pasetto and Baldo, 2011). Thus, some authors define the resilient modulus (or stiffness modulus) as an estimate of the elastic modulus of the material, based on the measurement of the stress and strain experienced by a specimen of bituminous material under high frequency cyclic loads, similar to those experienced by pavements under heavy traffic. Tests aimed at estimating this modulus are normally conducted in accordance with the American standard ASTM D 4123-82 (1995) or its analogues from other countries, such as the Australian standard AS 2891.12.1. In Europe, there is a reference standard, EN 12697 26 (2006) (Annex C), on indirect tensile strength testing on cylindrical specimens.

However, when materials show visco-elastic behaviour (as is the case of bituminous mixtures) the peaks of stress and strain due to cyclic loads do not occur simultaneously. Therefore, there would not be much sense in calculating a modulus that simply associates a given stress with a given strain at a given point in time. For this reason, the dynamic modulus (E^*) is defined as the absolute value of the complex modulus calculated through the ratio of the maximum stress (stress peak) and the maximum axial strain (strain peak) for each sinusoidal stress cycle.

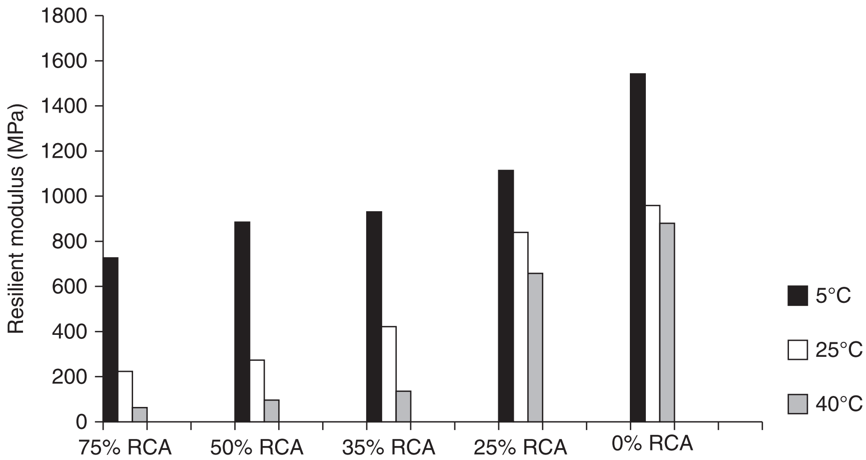
In hot asphalt mixtures, the substitution of NA for RA of different natures may produce various and, sometimes, even beneficial effects. For instance, it has been observed that mixtures containing brick powder filler show a higher modulus than mixtures containing NA at temperatures of 5 and 40 °C; but this value is lower at 25 °C. This factor may indicate a better fatigue performance than in other mixtures, since this type of failure normally appears at the latter temperature (Chen *et al*., 2011a).

Construction debris, which contains fractions of concrete, brick, plaster and other materials such as plastic and wood, may bring about almost unpredictable effects, as happened in the case of PD. For example, Shen and Du (2005) achieved the highest modulus for stiff binders, in accordance with the standard ASTM D 4123–1995, by substituting only the fraction of coarse aggregate and preserving the natural fines; whereas for less stiff binders, the highest modulus was achieved by substituting 50% of the aggregate (in all its fractions) when the test was conducted at 25 °C. However, when the test was conducted at 40°C, and with the same aggregate substitution rate, the lowest results were obtained. Surprisingly, the highest modulus at 40 °C was achieved with a 100% substitution rate (Fig. 15.3).



15.3 Resilient modulus at different temperatures and for different substitution percentages (0, 100 and 50% and containing recycled material only in the coarse fraction) using a Pen 60–70 bitumen AC-20 (a); and Pen 85–100 AC-10 (b) (Shen and Du, 2005, with permission from ASCE).

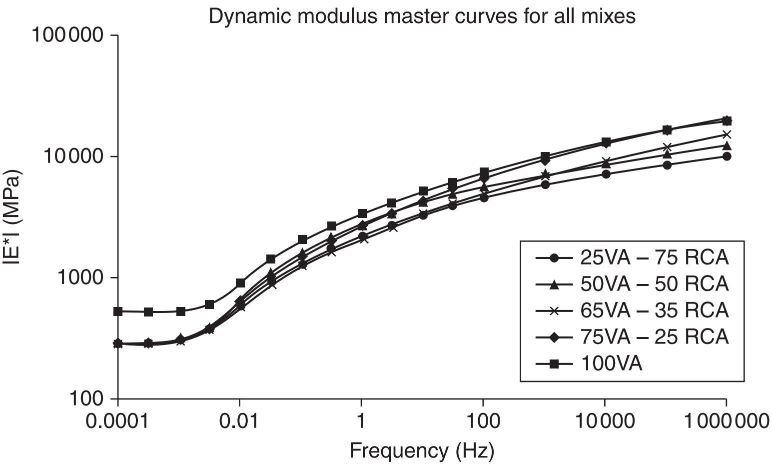
Despite the instability of these results, in general, the resilient modulus tends to decrease when this type of material is added to the mixture and as the bitumen content increases, which may be attributed to the use of a low strength mortar together with a relatively lower quality aggregate (Paranavithana and Mohajerani, 2006). Moreover, the stiffness modulus also tends to decrease as the temperature increases (Wong *et al*., 2007) although, in this case, the results obtained were completely comparable (and even higher) to those samples containing solely NA. However, the tests conducted by Mills-Beale and You (2010) agreed on a decrease of the resilient modulus as the test temperature increased, but they also showed an opposing trend, thus achieving a higher value as the aggregate substitution percentage decreased (Fig. 15.4).



15.4 Resilient modulus test results (ASTM D 4123–82) at different temperatures and for different recycled aggregate percentages (Mills-Beale and You, 2010).

With regard to the dynamic modulus (E*), no publications have been found on how hot and cold asphalt mixtures are affected by the addition of recycled materials as a substitute for NA. Nevertheless, studies carried out by Pérez *et al*. (2010a) proved that RA from construction and demolition debris, containing 50% of natural and 50% of RA in all its fractions, positively affect hot mixtures, thus slightly increasing their dynamic modulus in the case of coarse mixtures (without fines) and even having a more pronounced effect on semi-dense mixtures. Similarly, Mills-Beale and You (2010) studied the dynamic modulus of mixtures containing different percentages of recycled concrete. However, Pérez *et al*., 2010b), who applied the Spanish standard NLT-350 (1990) (3-point-flexion test), in turn conducted the tests in accordance with the American standard AASHTO T 62–03 (2004) (4-point-flexion test) at 13.2, 21.3 and 39.2 °C and at stress frequencies of 25, 10.5, 1 and 0.1. Consequently, they drew master curves (Fig. 15.5) that showed an opposite trend to Pérez *et al*. (2010a); that is, the higher the content of recycled concrete added to the mixture, the lower its dynamic modulus.

From these master curves, some other important conclusions can be drawn. It can be observed that, at low frequencies (Fig. 15.5, left), all mixtures containing RA perform in the same way, regardless of content, always showing a lower dynamic modulus than that of the sample containing 100% of NA. This indicates that, at high temperatures, mixtures containing RA will not be as stiff as mixtures containing NA. Therefore, they will be more prone to develop PDs. However, at high frequencies (Fig. 15.5, right), it can be observed how the dynamic modulus decreases as the content of RA increases. This means that, at low temperatures, mixtures with a higher content of RA will not be so stiff; thus being less prone to cracking, making them perfectly suitable for cold climates.

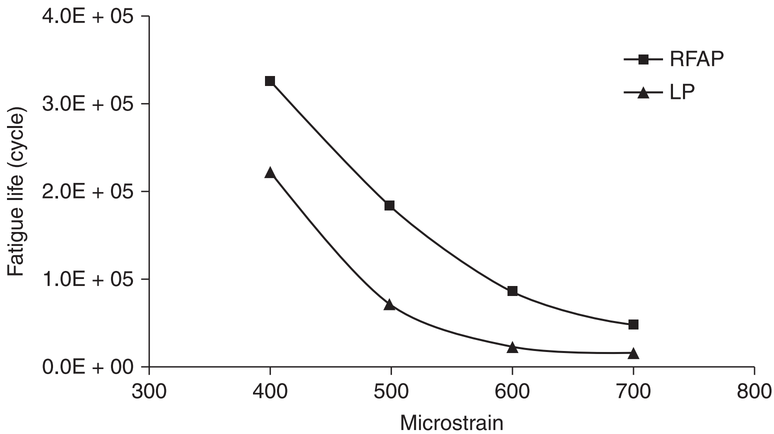


15.5 Dynamic modulus master curves for different contents of recycled concrete, ranging from 0% (line with squares) up to 75% (line with circles) (Mills–Beale and You, 2010).

15.5 Fatigue

Fatigue is defined as the reduction in the strength of a material when subjected to repeated loading, in comparison to the strength when subjected to an individual loading. Bituminous mixtures are devised to be affected mainly by these types of loads throughout their useful life. Therefore, it is crucial factor that should be taken into account, even at the early stages of their design, in order to avoid the appearance of frequent pathologies such as the well-known 'alligator cracking'. Thus, for more than half a century, a wide range of methods aimed at measuring and quantifying fatigue have been developed. Nowadays, the current versions of those first tests, such as the four-point bending test included in the American standard AASHTO T-321 (2003), are still being used. The EN 12697-24 (2007) standard regarding bituminous mixtures, *Bituminous Mixtures – Test Methods for Hot Mix Asphalt. Part 24: Resistance to Fatigue*, prevails in Europe.

Several authors, such as Kim *et al*. (1985), indicated that the fatigue performance of hot bituminous mixtures is closely linked to the fracture characteristics of the bitumen matrix. It has also been proved that, in general terms, the higher the stiffness modulus of a mixture, the better fatigue performance achieved (Pasetto and Baldo, 2011). At the same level of stress, an increase in the stiffness of the mixture will imply the development of fewer deformations and so the risk of the appearance of cracks will decrease. However, several factors, such as the substitution of the whole or part of the NA for other types of aggregate, may affect the aforementioned law. As a general rule, CDW tend to affect fatigue laws negatively (Pérez *et al*., 2010a), although the registered effect was more



15.6 Cycle number to failure versus strain level of hot bituminous mixtures containing recycled fines (RFAP) or limestone aggregates (LP) (Chen *et al.*, 2011b).

pronounced on mixtures containing a vast amount of fines (such as semi-dense mixtures) than on coarse mixtures, whose values did not differ from those corresponding to reference mixtures without RA.

Chen *et al*. (2011b) established that mixtures containing recycled fines showed a better fatigue performance than aggregates containing limestone fines in any of the deformations studied (between 400 and 700 $\mu\varepsilon$) (Fig. 15.6). Specifically, the brick powder used in the mixtures as filler provides them with a higher fatigue life in comparison with control mixes without recycled filler; being 1.73 and 1.75 times higher for 500 and 600 $\mu\varepsilon$, respectively (Chen *et al*., 2011a).

15.6 Stripping and durability

Stripping is a phenomenon that reduces the adhesive bond between the binder and the aggregate or by a softening of the cohesive bonds within the asphalt binder itself, normally under the loads of traffic and with the presence of moisture content inside the mixture. It usually begins at the bottom of a sealed layer and progresses upward, thus turning stripping into a phenomenon not easily detectable until the damage caused becomes evident and harmful. For this reason, the potential sensitivity of the mixtures to the presence of water is studied in laboratories (CTRE, 2005).

There are numerous factors that enhance their appearance, such as the type and use of the bituminous mixture, the characteristics of the mixtures and the aggregates, environmental factors during and after their placing, the use or not of anti-stripping additives, etc. The aforementioned causes trigger one of the two types of rupture mechanisms that have been studied until now: the failure of the aggregate-binder adhesion (adhesion failure) and/or the cohesion failure among the particles of its own binder (cohesion failure). With regard to both mechanisms, the former is the most widely accepted and, in the past, it was regarded by many authors as the only mechanism that could cause stripping (Kennedy, 1982; Majidzadeh and Brovold, 1968; Tunnicliff and Root, 1982).

In order to find a way to measure the water susceptibility of bituminous mixtures, since the 1930, many authors have proposed several test methods (Terrel and Shute, 1989). According to Hicks (1991), the process that leads to the collapse of the pavement due to water is divided in two phases: during the first phase, the aforementioned failures related to adhesion and cohesion appear; during the second phase, the mechanical failure appears under traffic loads, as a logical continuation of the first stage. There are, for instance, tests focused on the first phase, such as visual inspections, which consist of immersing specimens in hot or ambient temperature water to observe, after a certain period of time, the amount of uncoated aggregate. Laboratory tests, based on the second phase, where generally specimens are subjected to conditions similar to the real ones, subsequently compare their parameters (resilient modulus, indirect tensile strength, etc.) with those corresponding to unconditioned specimens. Finally, there is a third generation of tests, such as loaded wheel tests which, although designed to determine other pathologies such as rutting, are very useful in analysing water susceptibility as well.

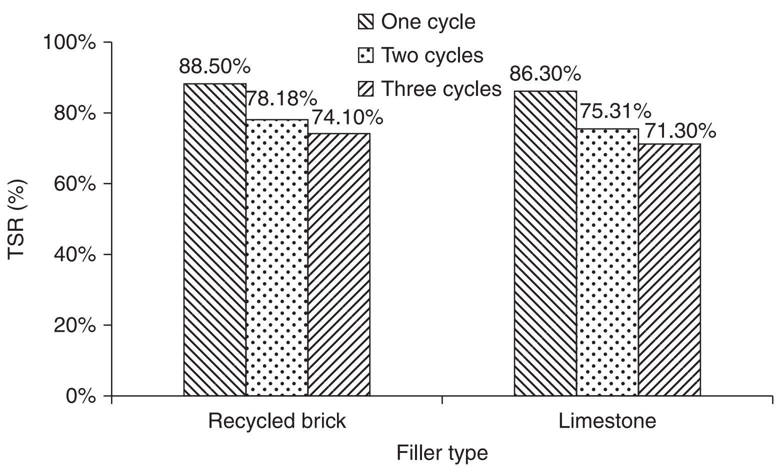
In recent years, the study of water susceptibility of hot bituminous mixtures containing RA has progressed in different directions. Similar to other parameters previously covered in this text, the huge heterogeneity of RA from CDW justifies the fact that several authors have found noticeably different results, although there is a general trend toward the deterioration in the resistance to stripping. According to Mills-Beale and You (2010), water susceptibility increases as the content of RA in the mixture increases. This trend can be clearly observed through an increase in the Tensile Strength Ratio (TSR) as the amount of recycled content decreases, although it is true that only those samples containing 75% or more of RA fall below the minimum level of 75%.

This same trend was confirmed by Pérez *et al*. (2010a), who concluded that some of the mixtures examined, such as coarse mixtures containing 50% of recycled content, obtained Retained Strength Index (RSI) values of 50%, well below the minimum of 75% established by the Spanish Standard PG-3 (Direccion General de Carreteras, 2004). In this research, it has also been pointed out that voids do not excessively grow when RA are added, thus proving that failure is not due to a higher water intake inside the mixture. Therefore, bad results may be attributed to the nature of the RA, which contains adhered mortar that would contribute to water retention inside the mixture, thus displacing the bitumen from the surface of the grains. Shen and Du (2005) concluded that the substitution of NA for aggregates from construction wastes decreased the TSR (determined by the conditioned Marshall stability divided by the control Marshall stability) when compared to those mixtures containing 100% NA. Nevertheless, in this case, drops were not so sharp, since a minimum result of 87.71% was obtained in mixtures containing 100% of RA. In mixtures where only coarse fractions were substituted, very small reductions occurred, approximately 1% in comparison with reference mixtures.

Pérez *et al*. (2012), after examining hot mixtures with RA substitution rates of 0, 20, 40 and 60% containing cement or limestone filler, also concluded that generally these mixtures showed a poor stripping performance, since only in 2 out of the 8 studied cases, the minimum required by the Spanish standard PG-3 was surpassed (Direccion General de Carreteras, (2004)). Moreover, Pérez *et al*. (2010b) also verified the existence of drops in the TSR when RA were added, which were attributed to the poor adhesion between the aforementioned aggregates and the binder. Paranavithana and Mohajerani (2006) determined that the stripping potential (up to 34%) was significantly higher than the one allowed (10%). Moreover, they also verified that mixtures containing recycled concrete showed a very different performance under dry and wet conditions, owing to the high absorption of water and strippability of the mortar under mixing and compacting conditions.

Nevertheless, there are other materials that not only do not damage adhesiveness to bitumen, but also improve it. For example, Silestone® decorative stone debris shows a favourable performance of recycled fines (Rubio *et al*., 2011), or of electric arc furnace iron and steel slag, which develop less sensitivity to water than mixtures containing 100% of limestone aggregate and obtain a higher TSR as the amount of waste added to the mixture increases (Pasetto and Baldo, 2011).

Finally, some researchers, such as Chen *et al*. (2011a), have calculated the TSR index; although by subjecting mixtures containing recycled filler from brick powder to freeze-thaw cycles. Logically, the TSR decreases as the number of freeze-thaw cycles increases but, in general, it increases with the addition of recycled filler (Fig. 15.7).



15.7 TSR values of asphalt mixtures with different filler (Chen *et al.*, 2011a).

15.7 Conclusions

Reusing CDW as aggregates of bituminous mixtures, instead of returning them to the environment through their disposal in garbage dumps while natural quarries are still being worked, implies a major advance in the field of highway engineering within the framework of the fight against climate change. Nevertheless, as described throughout this chapter, the aforementioned practice significantly influences their properties, both detrimentally and beneficially, even improving certain fundamental characteristics. Due to the great diversity and heterogeneity of the different types of RA, it is difficult to draw unique and universally applicable conclusions. However, it is indeed possible to observe certain general trends that affect the performance of bituminous mixtures after the addition of this type of aggregate. These trends are summarized below:

15.7.1 Specific gravity and absorption

In general, the high porosity and lightness of the cement mortar adhered to the recycled concrete aggregate provides bituminous mixtures with slightly lower specific gravities (i.e. 9 and 14%) and a significantly higher absorption (even reaching values up to 7 times higher). Other construction wastes, such as brick, enhance these effects, reaching lower specific gravities and absorptions above 30% (as against customary values of NA, <1.5%). This phenomenon should be taken into serious consideration when working with mixtures that require the addition of precise amounts of water, such as cold bituminous mixtures made with asphalt emulsions.

15.7.2 Rutting

In the light of various research, it still remains unclear whether the addition of aggregates from CDW is beneficial or detrimental. Both within cold and hot asphalt mixtures, beneficial and detrimental results were analysed, being difficult to predict when using different test temperatures and binder content. Particularly good results were obtained from the addition of brick powder to Silestone® decorative stone.

15.7.3 Stiffness

As in the previous case, both within the resilient and the dynamic modulus, it also remains unclear whether the addition of recycled materials has beneficial or detrimental effects on bituminous mixtures, having found very unstable and contradictory results.

15.7.4 Fatigue

Despite the fact that resistance to fatigue is normally related to the stiffness of bituminous mixtures, more positive results than negative results have been found, particularly through the substitution of natural filler for recycled filler. For example, the brick powder used in the mixtures as filler provides them with a higher fatigue life compared with control mixes without recycled filler, being 1.73 and 1.75 times higher for 500 and 600 $\mu\varepsilon$, respectively.

15.7.5 Stripping

It can be concluded that, in general terms, resistance to stripping decreases or is negatively affected when the amount of aggregate from CDW increases. In some cases, the resistance to the action of water was held above 75% (the usually required minimum value), although in other cases this resistance decreased to 50%. These reductions may be explained by the nature of RA containing adhered mortar, which prevents water from coming out from the interior of the mixture, thus displacing the bitumen from the surface of the grains. Nevertheless, it has also been found that other materials, such as Silestone® decorative stone debris, brick powder or granulated blast furnace slags, significantly increase the TSR index, thus improving the performance of bituminous mixtures.

15.7.6 Summary

Few solid conclusions can be drawn, so it is recommended that a particular study be conducted for each case, thus verifying whether the available RA is able to yield positive results. Nevertheless, positive experiences clearly indicate that, under certain requirements, it could be possible to make bituminous mixtures using RA, which are as good as or even better than those mixtures that only contain NA. So all the research reported throughout this chapter is just the starting point of a way that, without doubt, will be developed over the next few years to make road engineering more sustainable and economical.

15.8 Acknowledgements

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16
Recycled asphalt (RA) for pavements

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Abstract: In considering the use of alternative aggregates in pavements, Recycled Asphalt (RA) is the most suitable alternative material. First, because the materials contained in RA are of a high quality, as fresh aggregates, they meet standard requirements. Second, RA can be 100% recycled. Good quality control is critical in determining the use of recycled material in construction projects. This chapter discusses various methods for RA assessment, mix design, mechanical testing procedures, recycling methods and future trends in the recycling of asphalt pavements.

Key words: recycled asphalt (RA), asphalt assessment, asphalt design, hot mix asphalt (HMA) recycling, cold mix asphalt recycling.

16.1 Introduction

The function of an asphalt pavement structure is to distribute wheel loads over a large area of subgrade, which is part of the natural soil. In such a structure, the bituminous mix constituents play an important role. The constituents must have sufficient physical and mechanical strength to withstand high loads. The mechanical performance of a bituminous mix depends on the strength, shape and size of aggregates, along with the properties of the binder. The inclusion of alternative materials, such as RA, further complicates this highly heterogeneous structure. The inclusion of alternative materials in a new pavement mix is often a cause for concern, because of its variability and the possibility that it may contain contaminates. Thus, if an alternative material is to be used in a new pavement mix, it must satisfy the same physical and mechanical criteria as new virgin material.

Reusing asphalt pavement material is one of the best alternatives, since the material is already on site and has good material properties that allowed it to be part of the pavement structure to begin with. Studies have shown that pavement mixtures containing reclaimed asphalt (RA) can perform as well as mixtures designed using only virgin aggregate (Colbert and You, 2012a; Dinis-Almeida *et al*., 2012; Silva *et al*., 2012; Valdes *et al*., 2011; Fallon *et al*, 2010; Brownbridge, 2010; Tabaković *et al*., 2010; Widyatmoko, 2008; Shen *et al*., 2007; Byrne, 2005; Harrington, 2005; Jove *et al*., 2004; Reid and Chandler, 2001; Sherwood, 2001; Sulaiman, 1990; Gerardu and Hendriks, 1985). This chapter describes the use of Recycled Asphalt (RA) as an alternative material source. It also describes testing methods and pavement recycling methods and discusses future trends in recycling of asphalt pavements, including Life-Cycle Analysis (LCA) and Life-Cycle Cost Assessment (LCCA).

16.2 The recycling process for recycled asphalt (RA)

Recycled Asphalt (RA) can be obtained by milling or full depth removal, after which it must be crushed (TRL, 2002). It is important to be selective in the removal of old pavements for use in recycling, to ensure that the material removed is of a sufficiently high grade for future use. Prior to removing the old pavement, material testing can provide substantial information. Core samples should be drilled and extracted to assess the material properties and the amount of material that will be removed (NCHRP, 2001).

16.2.1 Pavement removal and crushing

Milling entails the removal of the pavement surface using a milling machine, which can remove up to 50 mm thickness in a single pass. Full depth removal involves ripping and breaking the pavement using a pneumatic hammer or a rhino horn attached to a bulldozer. Milling is the preferred method, as the use of a cutting or milling tool permits a more selective removal than does a full depth removal. In addition, using the milling method enables continued traffic access during the removal process.

The type of re-use suitable for pavement material is largely determined by the size of the slabs, lumps or blocks created when pavement is removed (Gerardu and Hendriks, 1985). If the material is to be processed through a crusher, the slabs removed from the road should not be larger than $1.0 \times 0.7 \times 0.4$ m.

The Renofalt asphalt regeneration process is a process where the lumps of RA, crushed to a 0 to 40 mm grading, are steam treated prior to the rejuvenation process. In its thermal process, large pieces of RA are passed through five steam heated compartments. Temperature and moisture penetration causes the lumps of RA to disintegrate (Gerardu and Hendriks, 1985). The advantage of the Renofalt process over the crushing method is that material is not shattered and so no dust is formed as a filler. However, if the asphalt regeneration is done using the 'Renofalt process', the lumps should be smaller ($\sim 0.4 \times 0.4 \times 0.4$ m).

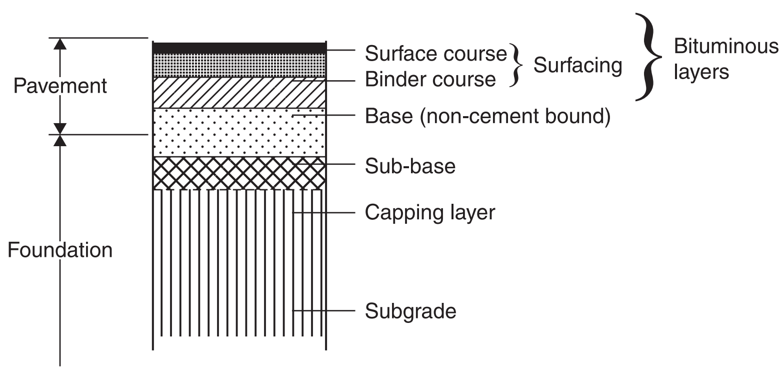
The methods of asphalt removal and crushing are important in the production of well-graded, good-quality RA from brittle, aged and hardened asphalt. New bituminous mixtures containing RA can be modified by the addition of fresh aggregates to suit any pavement layer. However, it is recommended that RA aggregate should not to be used to manufacture bituminous surface course mixtures, unless it possesses the high degree of uniformity required for such a mix (TRL, 2002). The uniformity and quality of the RA coupled with the type of recycling plant will determine the percentage of RA that can be used in new bituminous mixtures. The typical percentages of RA to be included in new mixtures ranges from 10 to 30%, although inclusions as high as 80 to 100% are feasible (EAPA, 2005).

The European standard, EN 13108: 2005, permits use of RA up to 10% in surface mixtures and up to 50% in concrete bases and/or binder course mixtures. The WSDOT in the US permits the inclusion of RAP up to 20% of the total aggregate weight (WSDOT: M 41–10, 2002). However, the M 41–10 standard allows for the inclusion of a higher RA content in Hot Mix Asphalt (HMA) production. A separate mix design is outlined in the M 41–10 standard that specifically accounts for the inclusion of a higher percentage of RA (>20%) in the HMA. This type of mix design often involves binder extraction from the RA, in order to enable grading and the appropriate adjustment of the virgin binder grade.

Once removed, RA should be separated into its constituent layers (Fig. 16.1). The higher grade recycled material from the surface course can be used in the reconstruction of the new surface course and any subsequent lower layers. Due to the lower quality of material used in lower pavement layers, the material from the lower layers cannot be used in the construction of a new top layer.

16.2.2 Stockpiling of RA

Where asphalt material is removed from the road and not used immediately, stockpiling of RA is a significant component of the recycling process. The full benefits of comprehensive material testing can be lost if great care is not taken when stockpiling the RA material. Depending upon the variability found during testing, it might be necessary to build separate stockpiles of materials taken from different sections of a road (TRL, 2002; Gerardu and Hendriks, 1985). It is important to record the detail of each load as it arrives into storage, taking note of



16.1 Schematic cross-section of a flexible pavement.

the type and amount of material, place of origin and any undesirable contaminants. If any batch is contaminated, it must be kept separate while identification tests are undertaken.

Gerardu and Hendriks (1985) emphasised the importance of ensuring that stockpiled RAP should not be piled higher than 3 m, in order to prevent caking, which can occur under the influence of generated heat. They also state that asphalt destined for regeneration should either be covered with a layer of sand or kept under a roof to prevent water penetration. However, the National Asphalt Pavement Association (NAPA, 1996) have stated that large piles of RA do not agglomerate, although a 250 to 300 mm crust may form at the surface of the stockpile and this should be removed prior to recycling. Nevertheless, the NAPA report concurs that stockpiles should be covered with a waterproof sheet or stored under a roof in an open-sided building to protect the stockpile from rainwater. RA can hold up to 7 to 8% of moisture, which would seriously reduce the amount of material that could be hot mixed and would also raise fuel costs and limit productivity (TRL, 2002). It is recommended that storage areas or collection points should not be accessible to the public, so as to prevent any unauthorised dumping of other types of waste.

16.2.3 Asphalt pavement recycling methods

There are two main methods of asphalt recycling: hot and cold, which can be further sub-divided into in-plant or on-site recycling (Dinis-Almeida *et al.*, 2012; Byrne, 2005; EAPA, 2005a). Recycling the old asphalt (RA) generally involves heating the material with or without the new mineral aggregates and the mix is laid hot (TRL, 2002; Hendriks and Janssen, 2001; Sherwood, 2001). For hot processing, materials are either mixed on site (on-site mixing) or in a mixing plant (in-plant mixing). Alternatively, cold regeneration of the old asphalt (RA) is generally processed in a crusher and can be used for the new binder course or sub-base. Virgin aggregate and bitumen can also be added to a new mix (Byrne, 2005; EAPA, 2005a; Gerardu and Hendriks, 1985). Over the last ten years, an alternative recycling method has become popular, Warm Mix recycling (Dinis-Almeida *et al.*, 2012). Warm Mix recycling is conducted in plant, the process being very similar to the hot processing method, but conducted at temperatures between 20 and 55 °C, lower than typical hot mix recycling (EAPA, 2010; Vaitkus *et al.*, 2009; D'Angelo *et al.*, 2008). The choice of the recycling process will depend on several factors, such as (EAPA, 2005a):

- the proximity of a suitable recycling plant;
- the nature, quantity and quality of the RA;
- the amount and type of possible contaminants within the RA;
- the programmed duration of construction;
- the availability of space for interim storage of RA prior to recycling;
- the engineering performance required from the new pavement.

Hot in-plant regeneration

In hot in-plant regeneration, the material removed from the surface of the existing road is sent to a hot mix plant, where it may be stockpiled for future use or processed immediately. In hot regeneration, RA is directly preheated and this method requires an extra dryer. The process begins with the RA being heated and dried in a second drum and transferred to a mixer. The hot gases from the recycling drum are either directed to the first drum (as secondary air near the burner) or to the baghouse. The virgin aggregate heated in the first drum is conveyed to a pug mill mixer. The batch hot in-plant recycling method can typically achieve 30 to 80% of recycling, with the upper limit being determined by the quality requirements of the mix specification in relation to the properties of the old asphalt (EAPA, 2005a).

Another variation of the batch hot in-plant recycling method consists of feeding the RA into the dryer via a recycling ring. In this process, the virgin aggregate and RA are introduced in separate sections of the same drum. The heating of the RA occurs behind the drum, ensuring that it does not overheat. This method allows up to 35% recycling (EAPA, 2005a).

Strengthened environmental requirements led to the development of three improved asphalt plants (EAPA, 2005a; Brock, 1997):

1. the parallel flow drum mixer;
2. the counter flow drum mixer;
3. the double barrel drum mixer.

The parallel flow drum mixing plant uses a parallel flow mixing drum, using both direct flame heating and super heated aggregate principles. In the split feed drum mixers, the processed RA is introduced into the mid-point of the drum, where both the superheated virgin aggregate and the hot burner gases heat the bituminous material.

The counter flow drum mixing plant differs from the parallel flow drum mixing plant in that the flow of hot burner gases and aggregates occur in the opposite direction. Technically, the counter-flow principle enables a reduction of the exit gas temperatures and an improved environmental performance, through reduced heating of RA. Problems can occur through 'blue smoke', the hydrocarbon emissions that are the result of the formation of very small particles generated when the binder on the reclaimed material is either boiled or burned due to the very high temperatures in the vicinity of the burner flame.

The counter-flow principle successfully overcomes the problem of blue smoke, as it retains the mix in the heating drum for longer and as a result the smoke emitted by the plant is minimised (*Shell Bitumen Handbook*, 1991). The virgin aggregates are introduced at one end of the drum (opposite the burner) and RA is introduced into the middle of the drum. The burner nozzle is extended into the drum, so that the preheating of RA takes place behind the flame before entering the mixing zone. As a result, the bitumen and the RA are never in direct contact with the flame or the heated gases. Under optimal conditions, this process allows up to 50% recycling (EAPA, 2005a).

The double barrel drum mixing plant consists of an ordinary revolving counter-flow drum surrounded by a fixed outer drum. The RA is introduced in the outer shell outside the hot gas stream. The virgin material is dried and super heated in the inner drum. It then enters the outer drum by falling through openings in the inner drum. Virgin aggregates then travel in the opposite direction to be mixed with injected bitumen and RA. Thus the mixing takes place in the space between the two drums through blending flights mounted on the exterior shell of the inner drum.

Another novel process is the Microwave Asphalt Recycling System (MARS) process (Hendriks and Janssen, 2001; Shoenberger *et al.*, 1995), which involves pavement removal and subsequent crushing of the asphalt rubble to a 0/14 size. This is followed by heating the mix to 135 °C in a counter-current dryer drum. Further heating then takes place by microwave in a second drum. Finally, a small quantity of rejuvenating oil is added. This method results in a high level of recycling (up to 100%) and produces very low air emissions. Heating the RA with electromagnetic energy (microwaves) avoids the problem of RA superheating encountered in conventional asphalt heating plants (Shoenberger *et al.*, 1995).

A new method, of preparing asphalt mixes using bitumen emulsion or cement as a binder, was introduced in the Netherlands in 1995. Known as Finfalt (Hendriks and Janssen, 2001), this process is based on the method of mobile plant material treatment first developed in Finland. The bitumen emulsion is developed on site immediately prior to dosage. The temperature during the manufacturing process is 90 °C and the heating process is achieved via the use of steam. This type of plant produces 100% RA aggregate in mixes. Thus, the asphalt aggregate containing tar can be converted into a high grade product. The advantage of processing this type of RA aggregate containing tar is that the aggregate is heated by steam, which fully condenses in the aggregate, and minimises emissions.

In comparison with the production of new asphalt, the hot regeneration of asphalt offers significant energy savings. This energy saving (in the form of materials, preparation, laying and transport) amounts to 20 to 70% when compared with conventional reconstruction methods (Gerardu and Hendriks, 1985). However, these figures depend on local conditions, the amount of new minerals and bitumen incorporated into the new mix and the process used.

Hot on-site regeneration

The hot on-site regeneration process is also known as 'surface regeneration', whereby the asphalt pavement is heated to a depth of a few centimetres, processed and then immediately re-laid. In this process, the heating temperature must be carefully controlled, because it must be sufficiently high to ensure that a strong bond will form between the old and the new materials. However, the temperature must not rise above 140 °C, to avoid bitumen ageing (Gerardu and Hendriks, 1985). Generally, the total depth of the treatment is about 50 mm, with cost savings of between 15 and 20% achieved when compared with more traditional recycling methods (Sherwood, 2001).

If the top layer processing is limited to heating, turning up, spreading and compacting without adding new material, it is called 'reshaping'. The heated and up-turned layer may be blended with asphalt chips (possibly pre-coated) to improve the uniformity and roughness of the surface layer, a process known as 'regrip'. When an additional layer of new material is applied to the heated and up-turned layers, the process is called 'repave'. The final type of surface regeneration entails gathering the heated and up-turned layer into a machine, mixing it with binder or a regenerator and processing it. This is known as the 'remix' process. The successful application of any one of these methods must satisfy two basic conditions (Hendriks and Janssen, 2001):

1. the layer to be treated must not exceed 40 mm;
2. the entire construction below the layer under treatment must be in a suitable condition for continuing use.

Warm recycling

Warm recycling is a technology that allows asphalt mixes to be produced and placed on-site at low temperatures. Warm recycling operates at temperatures of between 20 and 55 °C, lower than typical HMA (Dinis-Almeida *et al.*, 2012; EAPA, 2010; Vaitkus *et al.*, 2009; D'Angelo *et al.*, 2008). The warm recycling is simple and does not require any major modification to the hot-mix plant system currently used (Goh *et al.*, 2007). However, the warm recycling processes use different bituminous additives in the process. Various bituminous modifiers used in the mix preparation are patented products and are commercially available, such as WAM – Foam, Aspha-Min, Sasobit, Evotherm, etc. (EAPA, 2010; Vaitkus *et al.*, 2009). Some of the WAM processes can be described as follows (EAPA, 2005b):

- In one type of manufacturing process, fine powder (Zeolites – hydrated aluminium silicates) is added to the asphalt mix, which releases (upon heating, at temperatures between 130 and 140 °C) its hydration-bound water, which generates a foaming effect that triggers a volume increase of the asphalt binder. The fine steam bubbles form micro-pores that increase the compaction properties of the asphalt.
- Another suggested WMA method involves a two-stage process. In the first stage, a specially manufactured soft asphalt binder is used, which covers the aggregate surface at temperatures between 100 and 120 °C. In the second stage, a harder grade of asphalt binder is added in powder, foam or emulsion form to these pre-coated aggregates. The final mixture can be compacted at temperatures as low as 80 to 90 °C.

- In another process, organic modifiers are mixed with an asphalt binder, where the modifiers liquefy at about 100 °C, chemically changing the viscosity-temperature behaviour of the asphalt binder. The mix remains workable at temperatures as low as 90 °C.

When compared with hot recycling technology, warm recycling technology presents an improvement in working conditions for the construction crew. Reductions in pavement mix temperature reduce the fumes that workers are exposed to and provide a cooler, safer working environment. Reduced mixing and the compaction temperature of warm recycling offers the potential for the extension of the paving season, and can also be beneficial during cold-weather paving or when mixtures must be hauled long distances before placement (Vaitkus *et al*., 2009; Brosseaud and Saint Jacques, 2008; Manolis *et al*., 2008; Tušar *et al*., 2008; Kristjansdttir, 2006). The smaller differential between the mix temperature and ambient temperature results in a slower cooling rate. Since warm recycled mix can be compacted at lower temperatures, more time is available for compaction. The mechanisms that allow warm recycling to improve workability at lower temperatures also aid compaction. Improved compaction or in-place density tends to reduce permeability and binder hardening due to ageing, which tends to improve performance in terms of cracking resistance and moisture susceptibility (Hurley and Prowell, 2006).

Despite economic and environmental benefits of the WMA technologies, there are still doubts about their long-term performance. Prowell *et al*. (2007) recommended that further research is needed to validate the expected field performance of WMA mix, particularity in relation to mix compactibility, rate of gain of structural strength after construction (i.e. curing), rutting, fatigue, moisture sensitivity and the effect of different binder modifiers on the pavement design life.

The potential environmental and social benefits promised by WMA technology will undoubtedly stimulate interest for its wider use. It may be appropriate to place some emphasis on 'green' technologies during the procurement process, in order to encourage their use, as were used in the Greenroad project rating system (Soderlund, 2007). However, if the long-term performance of WMA is inferior to HMA, this negates any long-term financial or environmental benefits. The European Asphalt Pavement Association (EAPA), in its position paper on use of WMA (EAPA, 2010), stated that WMA procurement should be subjected to LCCA, to ensure that WMA technologies provide equivalent performance and value to HMA technologies and that the appropriate maintenance scenarios are fully assessed.

Cold recycling

The cold recycling process includes all methods where the old asphalt is processed and possibly mixed with new material, either on-site or in-plant, without heating. Cold mix recycling can occur at partial/shallow or full depth in an asphalt pavement, with the mixing carried out on-site or in-plant. The process preserves both aggregate and bitumen, minimises air quality problems and has low energy requirements. The existing pavement layers are processed with the addition of virgin aggregate, if required. The following aggregates could also be added: sand, bitumen emulsion, rejuvenating agent, lime, cement or a combination of these materials. The asphalt mix produced using cold mix recycling is suitable for binder course layers. In-place mixing produces material with a lower quality compared with the in-plant process, mainly because of the poorer mixture resulting from the mixing process (Gerardu and Hendriks, 1985).

Cold batch in-plant recycling

In cold mix in-plant recycling, the old pavement material is excavated, transported off site to a centralised facility, crushed and graded, mixed with bitumen and stored, awaiting site delivery. In this process, RA is added at the discharge of the dryer into the hot elevator. The RA material is heated by mixing with the hot virgin aggregate before entering the pug mill. Here, the designed amount of new bitumen is added to the mixture. This recycling method could result in recycling percentages of 10 to 40%, depending on the RA moisture content, the plant's vapour extraction system, the RA quality in relation to the required specification and the technical processes regarding maximum permitted temperatures.

Cold recycling in a stationary plant

Cold mix technology in an off-site central plant is a recent development that has been successfully used in a number of EU states for several years (EAPA, 2005a). RA removed from the site is brought to an off-site plant and put through a control process of crushing and screening. Two types of binder, foamed bitumen and bitumen emulsion, are used and combined with RA in a pug mill. Both recycling methods accommodate the production of over 90% of RA mixtures at a low energy cost (EAPA, 2005a). The smaller number of components and less complex nature of cold mix plants has resulted in their successful adoption for transportation to remote locations for short-term reconstruction projects (EAPA, 1995).

Cold on-site recycling

Cold mix on-site recycling to a shallow depth or 'Retread' (Sherwood, 2001) recycles roads to a depth of 75 mm, a process that has been in service in the UK for over 50 years. The procedure involves milling the existing road surface and adding virgin aggregate to re-profile the road surface, if necessary. When the desired profile has been achieved, a bitumen emulsion is applied. This is followed by compaction and a first sealing dressing of 14 mm chippings to fill any surface voids. Finally, a surface dressing is applied, typically using 6 mm chippings to give adequate texture depth to the surface. This process has been found to be cost effective, and is claimed to provide savings of approximately 25 to 35% when compared with traditional recycling methods (Sherwood, 2001).

For seriously damaged pavements, deep cold mix on-site recycling should be considered. This process typically treats the road to a depth of approximately 300 mm. This process offers two options: a full width process and a haunch repair process (Sherwood, 2001). The full width process operates at depths from 125 to 300 mm and the haunch process operates at depths from 150 to 250 mm. This method has proved an effective solution to potholes and deteriorating edges typical of older rural roads. During the process, the road is stabilised with cement to a depth of 200 mm and 1 m width. The full width of the road is then given a treatment to a depth of 80 mm.

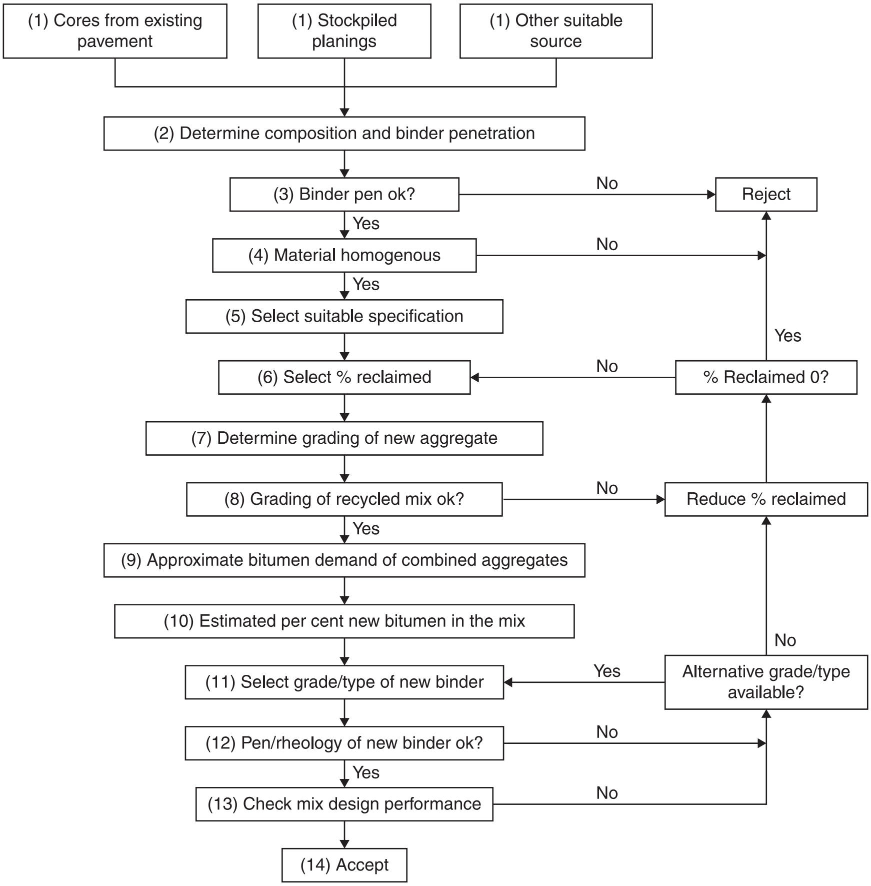
In 1996, Somerset County Council and W S Atkins embarked on a 4-year project in the United Kingdom to undertake road reconstruction works (Reid and Chandler, 2001). The project was named 'the linear quarry project' and sought to produce a structural design method for the cold on-site recycling process (CIRP) of bituminous pavements (Sherwood, 2001). The study illustrated that:

- CIRP was 15% less expensive than conventional reconstruction methods.
- Both the bitumen and cement-bound recycled treatments appeared to be equal in performance to virgin materials.
- More technical control was required in the CIRP.
- The main material variables were water content, quality and morphology of fines, binder content and the uniformity of mixing.

This work resulted in the production of TRL Report 386, *Design Guide and Specification for the Structural Maintenance of Highways by Cold* In-Situ *Recycling* (Milton and Earland, 1999). The report contains detailed information and advice relating to environmental consideration and site evaluation, which are required for the design of cold on-site recycled materials and their use for the structural maintenance of highway pavements.

16.3 Assessment of the properties of RA

The primary concern in using RA within a bituminous mix design is that RA contains a variety of materials such as crushed rock aggregates and binder and may also contain some contaminants. The base and surface courses from old roads could be mixed together in a single stockpile, along with RA from several projects. If the RA varies widely in properties such as gradation or binder content, the resulting binder course may also be variable. Therefore it is advised that material properties of each RA stock be investigated and proper stockpiling procedures be followed for storage. RA European Standard, EN 13108: Part 8: 2005, specifies requirements for the classification and description of reclaimed asphalt as a constituent material for an asphalt mixture. However, the standard fails to produce



16.2 Mixture design and assessment of suitability and proportioning of RA (Widyatmko, 2008).

an assessment procedure for RA use in an asphalt mix. A flow chart, proposed by Widyatmoko (2008), for assessing the suitability of reclaimed materials and the proportion to be recycled, is shown in Fig. 16.2.

16.3.1 Binder content

European standard, EN 12697: Part 39: 2004, provides a procedure for determining RA binder content (Bernier *et al*., 2012; Tabaković *et al*., 2010, Fallon *et al*. 2010). Four representative samples of RA were used and weighed before being heat treated. The samples were then placed in an oven at 530 °C for 30 min. Once the samples cooled, they were weighed again and the percentage of binder content in the mix was calculated. Tabaković *et al*. (2010) reported the binder content in the RA sample to be 1.83%, a very low binder amount in an RA sample. The explanation given by Tabaković *et al*. (2010) was that the RA was from a 'bad batch' that had been inappropriately stored and managed. This example illustrates the importance of correct RA storage management procedures. Typical RA binder content would be expected to be between 4.0 and 5.3% (Colbert and You, 2012a; Dinis-Almeida *et al*., 2012; Fallon *et al*, 2010)

16.3.2 RA binder viscosity

During a pavement's service life, the volatile components of bitumen evaporate and oxidation and polymerisation may occur (Gerardu and Hendriks, 1985) and bitumen may lose part of its visco-elastic properties, that is, it ages. Asphalt binder is a combination of asphaltenes and maltenes (resins and oils). Asphaltenes are more viscous than either resins or oils and play a major role in determining asphalt viscosity (Airey, 2003; cited in Wu, 2007). Oxidation of aged asphalt binder during construction and service causes the oils to convert to resins and the resins to convert to asphaltenes, resulting in age hardening and a higher viscosity, than in a fresh binder (Kandhal *et al*., 1995: cited in Wu, 2007). Although this process is irreversible, the visco-elastic state of the asphalt mix can be recovered through the addition of either bitumen with a higher penetration value or a rejuvenating agent such as cationic emulsions (Silva *et al*., 2012; Brownbridge, 2010; Tabaković *et al*., 2010; Widyatmoko, 2008; Jove *et al*., 2004; Sherwood, 2001; NCHRP, 2001; Gerardu and Hendriks, 1985). In order to optimise mechanical properties, including deformation resistance, it is recommend to use a virgin binder of not more than one grade softer to rejuvenate the reclaimed binder (Widyatmoko, 2008).

A rejuvenator is an engineered cationic emulsion containing maltenes and saturates. The primary purpose of a rejuvenator is to soften the stiffness of the oxidized asphalt binder and to flux the binder to extend the pavement life by adjusting the properties of the asphalt mix (Brownbridge, 2010). Some commercially available rejuvenating agents are Reclamite, Paxole 1009, Cyclepave and ACF Iterlene 1000.

An asphalt binder consists of two main fractions, asphaltenes and maltenes. The maltenes consist of sub-fractions, which are oily or resinous and chemically reactive. The principal obstacle to understanding the chemistry of asphalt aging was the lack of a reliable method for subdividing and defining the resinous and oily fractions of the maltenes (Brownbridge, 2010). The Rolster Analysis provides a reliable method of subdividing and defining the resinous and oily fractions of the maltenes (Brownbridge, 2010).

The Rostler Analysis provides a subdivision by determining four principal fractions of maltenes:

1. PC = Polar Compounds
2. A1 = First Acidss
3. A2 = Second Acidss
4. S = Saturated Hydrocarbons.

The influence of maltenes on the durability of asphalts as cementing agents has been shown to depend on the ratio of these four fractions. The maltenes distribution ratio is (ASTM Test D-2006-70):

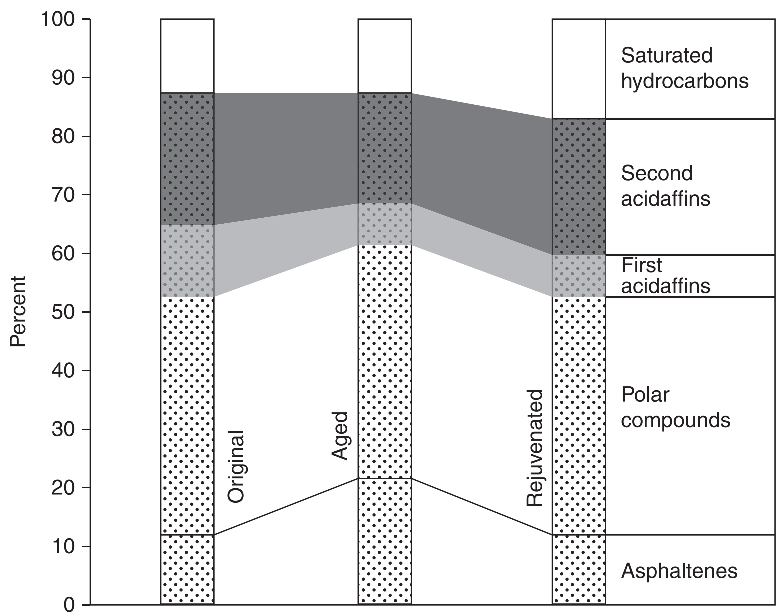
$$\frac{PC + A_1}{S + A_2} \tag{16.1}$$

The ratio sets the stage for a properly formulated rejuvenator. Figure 16.3 shows typical changes in the chemical composition of bitumen with pavement aging and addition of rejuvenators. After applying a rejuvenator, the aged binder re-acquires, almost entirely, its initial characteristics.

A simpler Colloadical Index, I_c, based on the ratio of the four main chemical components in an asphalt binder (asphaltenes, resins, aromatics, saturates) (Celauro *et al.*, 2010) is:

$$I_c = \frac{(asphaltenes + saturates)}{(aromatics + resin\ s)} \tag{16.2}$$

Rejuvenators need to be a fine-particle size cationic, oil-in-water emulsion of a selected blend of maltene components, tailored to facilitate and assure the desired mode of incorporation of the added maltene fractions into an asphalt pavement



16.3 Typical changes in chemical composition of bitumen after ageing and rejuvenation (Brownbridge, 2010).

(Brownbridge, 2010). In order for a rejuvenator to fulfil its functions, two key factors must be considered (Brownbridge, 2010):

1. **Proper base is essential**: naphthenic or wax free base is ideal – the molecular make-up offers more solvency, or absorption and fluxing ability with the binder.
2. **Rejuvenator must be manufactured as an emulsion:** typically 60 to 65% residual, it must have the ability to coat/wet the existing binder in the asphalt mix.

In order to determine the effectiveness of the rejuvenating agent in an asphalt mix, it is necessary to measure the reduction in the mix binder viscosity (Brownbridge, 2010). The tests required to achieve this are: i) tests for extraction and recovery of an asphalt binder, and ii) viscosity measurement tests.

The European Standard, EN 12697-Part 3:2007, outlines the procedure for binder recovery from an RA sample, using a centrifugal extractor (Dinis-Almeida *et al*., 2012; Fallon *et al*., 2010). North American (USA) procedures for asphalt extraction and recovery for asphalt binders include (Colbert and You, 2012a): AASHTO T164 quantitative extraction of asphalt binder from HMA, *ASTM D2172 Standard Test Methods for Quantative Extraction of Bitumen from Bituminous Paving Mixtures, AASHTO T170 Recovery of Asphalt from Solution by Abson Method*, and *ASTM D1856 Standard Test Method for Recovery of Asphalt from Solution by Abson Method*.

The EN 13302: 2010, ASTM 4402 or AASHTO T3T316 standards explain the procedures required to determine the rotational viscosity of the recovered binder or binder blends at 135 °C. Research results showed that RA recovered binder could be approximately 160% more viscous than fresh binder (Colbert and You, 2012a).

The needle penetration test can be conducted on the extracted RA binder sample, according to EN 1429: 2007. The penetration value of the mix binder can be calculated using the relationship (Fallon *et al*., 2010):

$$\log pen_m = \frac{b_1}{100} \times \log pen_1 + \frac{b_2}{100} \times \log pen_2 \qquad [16.3]$$

where:

b_1 = proportion of binder 1 in the mix
b_2 = proportion of binder 2 in the mix
pen_1 = penetration value of binder 1 (dmm)
pen_2 = penetration value of binder 2 (dmm)
pen_m = penetration value of binder mix (dmm)

If binder rejuvenator is not to be used in a mix containing RA, due to the ageing of the binder in the RA, the penetration value of the binder in the mix will decrease with increasing quantities of RA. Fallon *et al*. (2010) reported decreasing binder penetration values up to 36%, with the inclusion of 40% of RA in the mix, compared to the control mix.

New pavement mixtures, such as Stonemastic asphalt (SMA) and Porous asphalt, would contain binder containing polymer modifiers (Fallon *et al*., 2010). Therefore, the ATR FT-IR test (Bernier *et al*. 2012) should be carried out on extracted RA binders. Bernier *et al*. (2012) tested RA binder sourced from eight different RA samples, and found that all eight contained polymer modifiers.

16.3.3 Asphalt binder ageing procedures

Standards EN 12607-1: 2007 and EN 14769: 2005 describe short- and long-term asphalt binder ageing. The Rollong Thin Film Oven (RTFO) simulates short-term ageing, where binder specimens are aged at 163 °C for 85 min. This test is used to simulate HMA plant ageing. In order to simulate long-term binder ageing, the Pressure Ageing Vessel (PAV) is used, whereby test specimens are aged at 100 °C for 20 h. After short- and/or long-term ageing, test samples should be subjected to binder viscosity performance tests, in order to determine the influence of ageing on asphalt binder performance (Colbert and You, 2012a).

16.3.4 Particle size distribution

Using the aggregate sample from the burn off procedure described in Section 16.2.1, following an appropriate particle size distribution procedure, such as EN 12697-Part 2: 2002 (Fallon *et al*., 2010) or the British standard BS 812: Part 103.1: 1985 (withdrawn, replaced by EN 933: Part 1: 1997) (Tabaković *et al*., 2010), the particle size distribution of the RA sample can be determined. Table 16.1 shows that RA is a continuously graded material and so can be used to replace a proportion of each constituent in a 20 mm binder course mix.

Table 16.1 20 mm binder course mix constituent (including RA) percentage by weight passing

Sieve size (mm)	20 mm	14 mm	10 mm	CRF	Sand	RA
28	100	100	100	100	100	100
20	92.46	100	100	100	100	95.80
14	23.10	94.65	100	100	100	85.85
10	2.80	32.01	98.73	100	100	75.41
6.3	0.87	2.23	33.41	99.37	100	58.38
3.35	0.75	1.00	1.65	90.83	100	43.73
0.300	0.74	0.70	0.85	23.46	58.32	25.64
0.075	0.66	0.62	0.77	13.62	6.53	17.34

Source: Tabaković, 2007.

16.3.5 Relative density (RD)

The Relative Density (RD) is directly related to aggregate volume and is important in designing asphalt mixtures. Standard BS 812: Part 2: 1995 (replaced by EN 1097-Part 3: 1998), describes a procedure for determination of the aggregate relative density. The standard outlines three different methods of expressing the relative density of an aggregate: oven dried RD, saturated surface dried RD and apparent RD. Several methods may be employed, using either a wire basket, glass jar or pycnometer to hold the sample. The grading of the test sample should influence the choice of method.

16.3.6 Mineral content test

Due to the high possibility of RA aggregate variance, it is advised to investigate its mineral content. The X-ray diffraction (XRD) and X-ray flourescence (XRF) tests are used to verify the mineral content of the aggregate. Bernier *et al*. (2012) describe the XRD analysis and XRF procedures used in their study. For XRD analysis, each RA aggregate sample was pulverized to a fine powder, using a mortar and pestle mounted on a plastic holder. The testing was conducted by X-ray diffraction with a Scintag diffractometer with Cu source ($k = 1.5418$ Å). The X-ray tube was operated at 40 kV and 40 mA using a diffracted beam graphite-monochromator.

The data was collected between two theta values of 5 to 75 degrees with a step size of 0.02 degrees and an average counting time of 0.6 s per step. Qualitative analysis of the XRD data was performed using Jade software, version 8.5, with reference to the patterns of the International Centre for Diffraction Data database (powdered diffraction files, PDF-2. International Center for Diffraction Database, 2002, http://www.icdd.com). For RA aggregate XRF analysis, 500 g samples were prepared for each of the four burned-off RA aggregates split on the No. 4 sieve. After the split, particles were pulverized to a homogeneous stage, and placed into the XRF device (Innov-X Alpha™). The results of the XRF were collected and analysed. Bernier *et al*. (2012) used the XRF test results to support XRD findings.

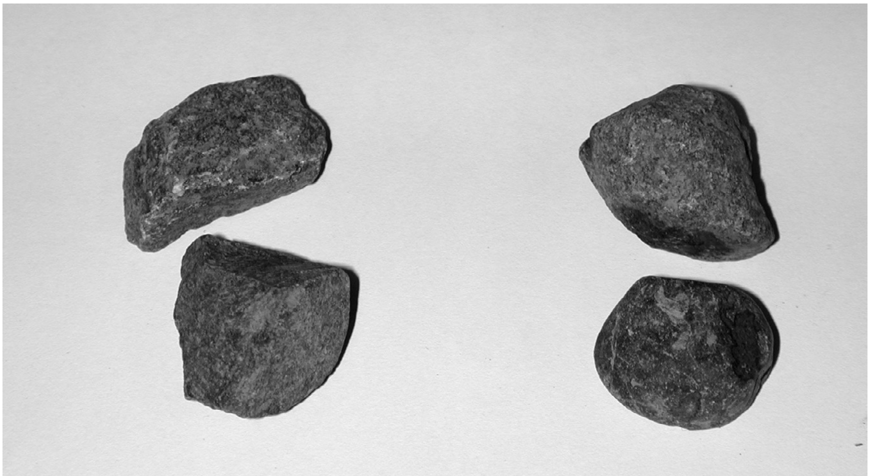
16.3.7 Aggregate particle shape

The main purpose of the investigation into aggregate particle shape was to ascertain the size and angularity of aggregates and to determine their potential to influence mix behaviour. In both natural and crushed rock aggregates, the particles within a particular size fraction contain a range of shapes. The shapes reflect the intrinsic petrographic characteristics of the material. Neville (1981) classifies aggregates into six groups: rounded, irregular, flaky, angular, elongated and flaky elongated. The shape of the aggregates in a mix has an influence on the performance of the mix. For example, increasing the proportion of flaky and elongated particles will cause the overall strength of the aggregate to decrease. Also flaky-elongated particles will tend to break beneath the roller and provide a poor embedment and mineral texture depth. However, a high flakiness index will improve the permeability of the mix (Woodside and Woodward, 1998).

Tabaković (2007) investigated the surface features of the RA mix constituents and reported that most of the RA aggregates had angular, flaky and flaky-elongated shape. The inspections did not reveal any significant differences in the shape and size of the RA and virgin aggregates. However, small differences were found to exist between large RA and virgin aggregates. The edges of the RA aggregates were not as sharp as those of the virgin aggregates (Fig. 16.4), which is believed to be caused by in-service effects such as temperature, water drainage and traffic, to which the RA aggregates were exposed during their service life and as a result of RA processing methods such as milling and crushing. Such physical properties of aggregates may influence the mechanical behaviour of the mix, for instance perhaps leading to premature pavement rutting.

Bernier *et al*. (2012) used Fine Aggregate Angularity (FAA) test (AASHTO T304) to measure the non-compacted air voids of an aggregate sample prepared from specific fractions of aggregate from number 8 to number 50 sieves. The sample was prepared by allowing fine aggregate particles to free fall into a known volume from a particular height. Knowing the specific gravity, the percentage of air voids was then calculated. In combination with the FAA test, the Flat and Elongated (FE) test (ASTM D4791) was then used to quantify the percentage of a coarse aggregate that had a greater than 5:1 length to width ratio (Bernier *et al*., 2012). It is known that there is a relationship between non-compacted air voids, course to fine aggregate ratio and rutting susceptibility (Bernier *et al*., 2012).

Bernier *et al*.'s (2012) RA aggregate testing did not reveal any firm evidence to prove the rutting susceptibility of mixtures. Basalt RA aggregate illustrated the



16.4 Samples of 20 mm virgin (left) and RA (right) aggregate indicating angular shape of the aggregates (Tabaković, 2007).

lowest FAA value and a finer gradation, which normally suggests increased rutting susceptibility for mixes with this RA source. However, these effects were offset by the low FE value as well as the higher binder content in the basalt-based RA (assuming RA binder will have a stiffening effect on the rutting performance test). However, limestone-based mixes could be identified as potentially prone to rutting due to high FE and low binder content, but these were also balanced with fairly high FAA values.

16.3.8 Mix surface area factor

The surface area factors (SAF) of the mixture were used to establish the change in surface area resulting from the inclusion of different percentages of RA into the mix and to identify whether adjustment was required to the amount of added binder content in the resulting mix. The SAF calculation was used to identify potential anomalies that could cause the result to deviate from those expected. The SAF of a mix aggregate was calculated by the following equation (Shell Bitumen, 1991):

$$SAF = \frac{SSAF}{MPSD} \qquad [16.4]$$

where:

SAF = surface area factor (m^2/kg);
$SSAF$ = sieve size surface area factor (m^2/kg);
$MPSD$ = mix particle size distribution (%).

Table 16.2 summaries the SAF calculations. From the results we can observe that the SAF for each constituent decreased gradually with increased RA content in the mix. The sole exception to this trend was for constituents with a sieve size of

Table 16.2 20 mm binder course mix surface area factor

Sieve size (mm)	Surface Area Factor (*SAF*) (m^2/kg)			
	0% RAP	10% RAP	20% RAP	30% RA
28	0.410	0.410	0.410	0.410
20	0.400	0.396	0.393	0.389
14	0.308	0.302	0.305	0.306
10	0.245	0.241	0.243	0.247
6.3	0.184	0.181	0.183	0.182
2.36	0.271	0.263	0.257	0.243
1.18	0.409	0.397	0.379	0.345
0.6	0.564	0.571	0.554	0.512
0.3	0.927	0.944	0.913	0.839
0.15	1.082	1.077	1.053	0.992
0.075	1.515	1.420	1.411	1.384
Total SAF:	**6.315**	**6.202**	**6.101**	**5.849**

Source: Tabaković, 2007.

0.6 mm and 0.3 mm for the mix containing 10% RA. The results show that the SAF of the mix containing 10% RA (for 0.6 and 0.3 mm sieve sizes) is higher than the SAF of the mix containing 0% RA for the same sieve size grading. Nevertheless, the total SAF of the binder course mix decreases with increased RA content, implying a reduction in the required added binder content.

16.3.9 Environmental properties of RA

In the recycling process, the environmental properties of recovered materials must be considered, in order to ensure that there are no potentially harmful constituents present in the mix (no leaching or toxic constituents, dust particles which might cause air emission concerns, etc.). Concerns over potential pollutant leaching from RA come from various sources (Brantley and Townsend, 1999). Bitumen is a derivative of petroleum containing different types of hydrocarbons. During its service life, it comes into contact with many chemicals generated from traffic, such as vehicle exhaust, gasoline, lubricating oils and metals from tyre brake lining wear.

The major chemicals typically contained in asphalt pavement are heavy metals (Cd, Cr, Cu, Ni, Pb and ZN) and Polycyclic Aromatic Hydrocarbons (PAHs) (Legret *et al*., 2005; Brantley and Townsend, 1999). Traffic related sources of PAHs include vehicle exhaust, lubricating oils, gasoline and tyre particles (Sadler *et al*., 1999; Takada *et al*., 1990). Legret *et al*. (2005) studied the leaching effects of heavy metals and PAHs from RA. They reported that pollutant leaching is weak and that the concentration of leaching remained below EU limit values for drinking water. However, the study also showed that in comparison with new conventional asphalt, the concentrations of total hydrocarbons and some PAHs were higher in leachate from RA. This shows that environmental tests, such as batch and column leaching tests (EN 12457:2002; DD EN/Ts 14405; 2005 (E)), should be conducted on recovered materials due for re-use in new road construction projects. The tests may also be used for quality control purposes, such as to ensure that materials used in road construction comply with regulating criteria (Reid and Chandler, 2001).

16.4 Designing a pavement mix containing RA

16.4.1 RA mix design

The primary concern in using RA in the mix is that it contains a variety of materials such as crushed rock aggregates and binder, as well as possibly containing some contaminants. Base and surface courses from old roads could be mixed together in a single stockpile, along with RA from several projects. If RA varies widely in properties such as gradation or binder content, the resulting binder course may also be variable. Good stockpile management should be followed to control material variability. A set of grading results for the RA and virgin aggregates are presented in Table 16.3, showing that RA is a continuously graded material and as such can be used to replace a proportion of each constituent in the mix (Taboković *et al*., 2010).

Following BS 4987: Part 1: 1993 (withdrawn, replaced by EN 13108: Part 1: 2006), 4 mixes containing 0, 10, 20 and 30% RAP were designed (Tabaković *et al.*, 2010). The grading results are presented in Fig. 16.5, illustrating how the mix designs fit within the standard grading envelope region. The standard design control envelope to region allowed us to create a best particle distribution for the mix designs, and consequently to design the best mix (Table 16.4).

Table 16.3 Percentage by weight passing

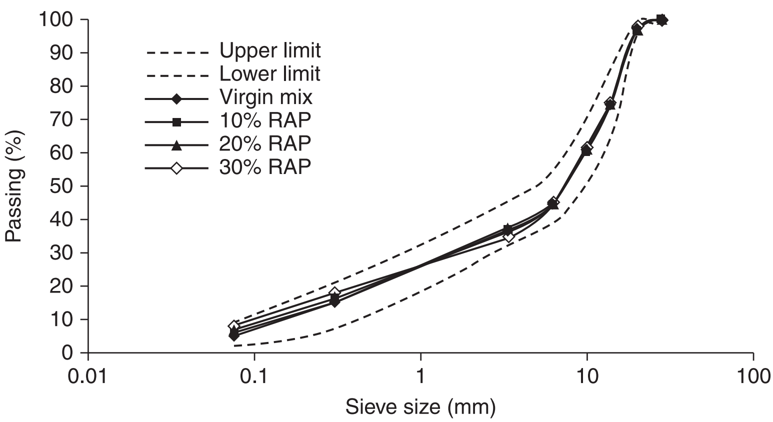
Sieve size (mm)	20 mm	14 mm	10 mm	CRF[1]	Sand	RAP
28	100	100	100	100	100	100
20	92.46	100	100	100	100	95.80
14	23.10	94.65	100	100	100	85.85
10	2.80	32.01	98.73	100	100	75.41
6.3	0.87	2.23	33.41	99.37	100	58.38
3.35	0.75	1.00	1.65	90.83	100	43.73
0.300	0.74	0.70	0.85	23.46	58.32	25.64
0.075	0.66	0.62	0.77	13.62	6.53	17.34

Source: Tabaković *et al.*, 2010.

Table 16.4 Mix designs

Percentage content of constituent in the mix						
Mix No.	RAP	20 mm	14 mm	10 mm	CRF	Sand
1	0	31	12	18	27	12
2	10	31	10	16	20	13
3	20	28	10	14	16	12
4	30	26	8	14	12	10

Source: Tabaković *et al.*, 2010.



16.5 Particle size distribution percentage passing (Tabaković *et al.*, 2010).

The mix binder content can be calculated following the binder extraction from the RA and determining the amount of binder contained in the RA. The extraction procedure is explained in the Section 16.2.1. Approximate binder content can be calculated, based on combination of aggregates in RA, by employing the following equation (Dinis-Almeida *et al*., 2012; Widyatmoko, 2008):

$$P_b = 0.035a + 0.045b + Kc + F \tag{16.5}$$

where:

- P_b = the approximate total bitumen demand of recycled mixture, percentage by weight of mixture;
- a = the percentage of mineral aggregate retained on a 2.36 mm sieve;
- b = the percentage of mineral aggregate passing 2.36 mm sieve and retained on a 75 micro sieve;
- c = the percentage of mineral aggregate passing 75 micro sieve;
- K = the 0.15 for 11–15% passing 75 micro sieve, 0.18 for 6–10% passing 75 micro sieve, 0.20 for 5% or less passing 75 micro sieve; and
- F = the absorption factor of aggregates (0–2%).

Knowing the approximate amount of binder in the RA, the quantity of new bituminous binder (P_{nb}) to be added to the mix expressed as percentage by weight of aggregate is calculated by (Widyatmoko, 2008):

$$p_{nb} = \frac{(100 - rP_{sb})P_b}{100(100 - P_{sb})} - \frac{(100 - r)P_{sb}}{100 - P_{sb}} \tag{16.6}$$

where:

- P_{nb} = the percentage of new bitumen (rejuvenating binder, plus recycling agent, if used) in recycled mixture;
- r = the new aggregate expressed as a percentage of the total aggregate in the recycled mixture;
- P_b = the percentage, bitumen content of total RA mixture or asphalt demand, determined by empirical formula above;
- P_{sb} = the percentage bitumen content of reclaimed asphalt pavement.

Dinis-Almeida *et al*. (2012) and Tabaković *et al*. (2010), in an attempt to determine the optimum binder content for the mix, employed the Marshall test (BS 598: Part 107: 1990). The specimens were compacted in the gyratory compactor in accordance with the EN 12697-31:1998 test standard. Tabaković *et al*. (2010) compacted the test specimens at dimensions of 100 mm in diameter and 63.5 mm in height with a void content of 6%. First, the specimens were placed in a water bath at 60 °C for 45 min. Once the specimens reached the test temperature, they were placed into the test frame and a load was applied at a constant rate of deformation of 50 mm/min. During the tests, the maximum applied load and the vertical deformation of the specimen were recorded. Table 16.5 presents the test

Table 16.5 Optimum binder content for binder course mixes

RAP content (%)	Optimum binder content in the mix (%)
0	4.70
10	4.20
20	4.16
30	4.00

Source: Tabaković *et al*., 2010.

results, which illustrate that the optimum binder content for the mix decreases with an increase in the percentage of RAP in the mix.

However, Dinis-Ameida *et al*. (2012) reported that the Marshall test results for estimating the optimum binder content remained inconclusive for all the recycled mixtures, when compared to the national (Portuguese) specification limits of the virgin hot mix.

16.5 Testing the mechanical properties of designed mixtures

In order to be considered viable for re-use, the RA as an alternative aggregate must have good chemical and physical properties. In addition, the pavement mix containing RA must satisfy the same mechanical requirements as a standard mix containing only virgin aggregate. This is determined by conducting standard laboratory tests on the mix containing RA (Dinis-Almeida *et al*, 2012; Tabaković *et al*., 2010; Fallon *et al*., 2010), such as stiffness, fatigue, moisture damage and wheel tracking tests.

16.5.1 Laboratory mixing

The specimens used to establish the mechanical properties of a bituminous mixture must be homogenous in density and compaction (Bonnot, 1997). This means that for laboratory based tests it is necessary to simulate the mixing process, conditioning and the compaction process in order to ensure that the test specimens produced are compatible with the material found on site. Visser (1996; cited in Hartman, 2000) reported difficulties in the reproducibility of plant mixing in the laboratory. This indicates that the mixing process could have some influence on the material characteristics determined from laboratory mixed specimens. A problem arises with bitumen, which is distributed more uniformly throughout the mix in plant, in comparison with laboratory prepared mixtures (Hartman, 2000).

The amount of bitumen required to smear the mix drum in a plant is small compared to a laboratory mixing bowl, which causes the amount of bitumen in a given mixture to vary depending on batch size. The degree of ageing during plant mixing depends heavily on wet mixing time. Binder penetration decreases with longer mixing times and prolonged exposure to air during mixing (Brock, 1997). However, Pereira *et al*. (1997) revealed that dense binder course mix (DBM) test specimens have identical fatigue properties, regardless of whether they are mixed in the plant or in the laboratory.

16.5.2 Laboratory compaction methods

Compaction is one of the most important factors to be considered when constructing laboratory test samples, as it influences the sample density, the air voids and the shape, size, distribution and orientation of the aggregate particles (Fordyce, 1997). Methods of laboratory compaction include:

- static
- impact (Marshall hammer)
- vibratory
- gyratory
- roller compaction.

The influence of compaction on the properties of bituminous mixtures has been the focus of numerous studies (Byrne, 2005; Hunter *et al*., 2004; Gibney, 2002; Harman *et al*., 2002; Pereira *et al*., 1997; Brock, 1997; Fordyce, 1997; Huschek, 1985; Nunn, 1978; Epps *et al*., 1969). Some of the material properties influenced by compaction include aggregate orientation, void content, fatigue, stiffness and permanent deformation. However, Button *et al*. (1994) concluded that asphalt mixtures with stiff binders are relatively unaffected by compaction.

The Marshall compactor lacks a kneading action to re-orientate the aggregates and the impact forces degrade aggregates (Hartman *et al*., 2001). During the static compression technique, very high pressures must be applied to achieve the required material density, resulting in the crushing of aggregates and squeezing of the binder film and so the microstructure of the compacted mixture is changed compared to *in-situ* material (Bonnot, 1997).

Gyratory compaction produces specimens that are closer to field material compared to impact and static compression but specimens are not always homogeneous (Hartman *et al*., 2002). The segregation of larger aggregates to the sides of the cylindrical compaction mould occur, which results in higher void contents towards the exterior of the specimen (Voskuilen, 1996; Harvey *et al*., 1994).

Rolling wheel compaction appears to be the most appealing due to its similarities with field compaction (i.e. compaction action and air void distribution) (Hartman *et al*., 2002). However, rolling wheel compaction proved to be a somewhat impractical method as the sole means of laboratory compaction, because the equipment involved was large and required very large batches of mixture (Tabaković, 2007).

16.5.3 Test specimen storage

EN Standards, such as EN 12697 (2004), specifies a short- and long-term specimen storage procedure. If a testing programme is designed to occur 24 h after mix compaction (Tabaković *et al*., 2010), the specimens should be kept in the laboratory at room temperature (20 ± 2 °C). However, if the laboratory test programme takes place over several weeks, it would therefore be necessary to store specimens in suitable conditions until testing, to avoid mixture ageing effects. When the specimens were not to be used within 96 h (four days) of compaction, the specimens should be stored at 5 °C, as required by the EN Standards (EN 12697). The specimens should be removed from the fridge 24 h prior to testing and stored in a temperature conditioning chamber. The specimens then should be conditioned at a test temperature of (20 °C) for at least 6 h prior to testing (Tabaković *et al*., 2010).

16.5.4 Laboratory testing

A pavement mix containing RA must satisfy the same mechanical requirements as a standard mix containing only virgin aggregate, in order to be considered as a viable asphalt mix. This is determined by conducting standard laboratory tests on the mix containing RA (Silva *et al*., 2012; Colbert and You, 2012b; Tabaković *et al*., 2010; Celauro *et al*., 2010; Fallon *et al*. 2010; Shu *et al*., 2008). A list of standard mechanical laboratory tests is given in Table 16.6. Research findings show that asphalt mixtures containing RA have as good and in some cases even better mechanical properties comparing to the control asphalt mix. For example, Tabaković *et al*. (2010) reported that a 20 mm binder course mix containing up to 30% of RA can outperform a control mix containing no RA.

However, Tabaković *et al*. (2010) also warn that inclusion of higher RA content (> 50%) could reduce the durability of the mix. Other studies (Silva *et al*., 2012; Colbert and You, 2012b; Widyatmoko, 2008; Shu *et al*., 2008), report that mixtures containing a high concentration of RA in the mix (up to 50%) are not susceptible to water damage. Silva *et al*. (2012) analysed completely (100%) recycled HMA and reported good water sensitivity results, although the mixture without additive

Table 16.6 List of standard asphalt pavement mix tests

Test	Standard
Indirect Tensile Stiffness Modulus Test (ITSM)	EN 126–26
Water Sensitivity Test	EN 12697–12
Indirect Tensile Strength Test (ITS)	EN 126–23
Creep Test	EN 14771
Fatigue Tests	EN 126–24
Particle Loss Test (Contabro Test)	EN 12697–17
Accelerated Wheel Tracking Test	EN 12697–22 & BS 598–110

(RA) was slightly more sensitive to the presence of water (lower ITSR due to higher voids content).

The remaining mixtures containing rejuvenators had a better performance (durability). Research reports (Colbert and You, 2012b; Shu *et al*., 2008; Sulaiman 1990) also reported higher stiffness values of the mixtures containing the RA, due to the ageing of the RA binder, which can increase mix brittleness and reduce mix fatigue life. However, with addition of a rejuvenator, this improved and stiffness values were reduced (Silva *et al*., 2012; Widyatmoko, 2008; Terrel and Fritcher, 1978). Widyatmoko (2008) reported that the fatigue performance of containing higher RA content improved with increasing proportions of rejuvenating binder in the mixture.

Widyatmoko (2008) also reported lower resistance to permanent deformation of RA mixtures compared to control mix. However, the results of the resistance to permanent deformation of the RA mixtures would be considered to be at an acceptable level and meet the UK wheel-tracking requirements for very heavily stressed sites requiring high rut resistance. These findings illustrate that RA is an ideal asphalt mix aggregate; however, a careful RA analysis and mix design procedure must be followed in order to achieve an ideal RA mix.

16.6 Future trends

International research has demonstrated that RA is an ideal alternative material for road construction, provided there is good material quality control. This is a good indicator that use of RA in pavement construction and maintenance is becoming a popular option. It is safe to say that future trend will focus on:

1. improving the standards and procedures for an evaluation of RA material; and
2. further development of construction processes, such as warm mix asphalt (WMA) technology.

The use of recycled materials in road construction is important to divert loads that would otherwise be disposed of in landfills. However, simply diverting the waste from landfills to aggregates supply is being questioned for its energy usage and CO_2 footprint. Recycling or re-use of recycled materials needs up-to-date studies on the associated environmental impacts, including energy use, emissions, leaching, etc. A LCA approach is gaining popularity in meeting the needs of sustainable asphalt pavement construction (Carlson, 2011; Santero *et al*., 2011a–c; Huang *et al*., 2009, Chiu *et al*., 2008). The LCA is used for establishing environmental footprints, comparing alternative systems, validating and marketing 'green' claims and identifying opportunities for improvement within the life cycle, with the strongest application focused on reducing environmental impact (Santero *et al*., 2011c). For the past two decades, specific LCA models have been designed specifically for pavements, and there is general acceptance of LCA as a tool in pavement design (Carlson, 2011; Santero *et al*., 2011a–c). Santero *et al.*

(2011a,b) evaluated the overall utility of existing research and development of pavement LCAs and provided recommendations for filling identified gaps. The study found that that there are still notable differences between existing studies, including functional units, system boundaries, goals, scopes and data.

Santero *et al*. (2011a,b) identified that most studies are project specific and that the conclusions are not necessarily universal. The study concluded that there is a need for a standardised pavement LCA framework, which will provide designers, researchers and other stakeholders with the ability to accurately and consistently characterise the impacts of pavement construction and maintenance. Most importantly, the development of a standardised framework would reduce inconsistencies across the methodologies (Santero *et al*., 2011a,b). It would ensure that system boundaries are based on a given goal and scope. The continual advancement of LCA will fill the knowledge gaps, such as mechanistic pavement-vehicle interaction models, it will improve the accuracy of the pavement LCA, and allow the assessment process to provide a robust solution to a wider set of problems.

It is foreseen that future trends in pavement recycling processes will be the development of a general LCA tool that will account for all parameters in pavement design, environmental burden caused by material production, transport, pavement-vehicle interaction, traffic congestion and even economic aspects of pavement maintenance and construction. The LCA will generate comprehensive and scientifically defensible strategies for lowering emissions, reducing waste and minimising consumption of energy and natural resources in pavement construction and maintenance processes. Adopting a LCA is the key to establishing an effective path toward reaching environmental goals in pavement construction and maintenance.

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17

The suitability of concrete using recycled aggregates (RAs) for high-performance concrete (HPC)

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Abstract: Most studies related to concrete made with recycled aggregates (RA) use uncontaminated aggregates produced in the laboratory, revealing the potential to re-use as much as 100%. However, industrially produced RA contain a certain level of impurities that can be deleterious for Portland cement concrete, thus making it difficult for the concrete industry to use such investigations unless uncontaminated RA are used. This chapter reviews current knowledge on concrete made with RA, with a focus on the crucial importance of the presence of impurities, and how those aggregates are not suitable for the production of high-performance concrete (HPC). The potential of geopolymers to produce HPC based on high volume RA is also discussed.

Key words: concrete and demolition waste (C&DW), recycled aggregates (RA), impurities, high-performance concrete (HPC), geopolymers.

17.1 Introduction

The high volumes of concrete and demolition waste (C&DW) that are currently generated constitute a serious problem in terms of landfill space. Pacheco-Torgal and Jalali (2011) estimate that non-recycled C&DW, generated in the 15 leading EU member states (MS), represents a volume occupying 10 m in height across 13 km^2 of landfill each year. Such estimates are only approximate, because these kinds of wastes are illegally dumped in most countries. C&DW recycling rates differ from country to country. While the European average is only 47% (Solis-Guzman *et al.*, 2009), some countries may reach 80%, as is the case in Denmark or in the Netherlands (Chini, 2005). Eurostat (2010) mention a total of 970 million tons/year of C&DW, representing almost 2 tons per capita. However, currently the average recycling rate of C&DW for the 27 EU MS is just 47% (Table 17.1).

According to the Revised Waste Framework Directive 2008/98/EC, the minimum recycling percentage of C&DW by the year 2020 should be at least 70% by weight. This target and also the initiative, ‘A resource efficient

Table 17.1 Recycling rates of C&DW in Europe

Countries	Recycling rates%
Belgium (Flanders)	>90
Denmark, Estonia, Germany, Ireland, Netherlands	>70
Austria, Belgium, France, Lithuania, UK	60–70
Luxemburg, Letónia, Eslovenia	40–60
Average recycling rate for EU–27	47
Cyprus, Czech Republic, Finland, Greece, Hungary, Poland, Portugal, Spain	<40
Bulgaria, Italy, Malta, Romania, Slovakia, Sweden	No data available

Source: Sonigo *et al.*, 2010.

Europe' (COM, 2011), shows the determination of the EU to promote the importance of recycling. Consumption of aggregates worldwide amounts to about 20 000 million tons/year and an annual growth rate of 4.7% is expected (Bleischwitz and Bahn-Walkowiak, 2011). More than one-third of this consumption is related to concrete production. Concrete is the most used construction material on Earth, with approximately 10 km^3 produced every year (Gartner and Macphee, 2011).

The environmental impacts of primary aggregates include consumption of non-renewable raw materials, energy consumption and the reduction of biodiversity at extraction sites. Since the cost of aggregates is highly dependent upon transport distances, extraction operations have to be near construction sites, which in turn multiply the number of quarries and their impact upon biodiversity. However, the benefits of proper C&DW management are not solely environmental.

A good example of the economic benefits associated with recycling C&DW is illustrated by the Environment Agency of the United States, which looks at the impact of job creation from C&DW. Incineration of 10 000 tons of waste can lead to the creation of 1 job, landfilling can create 6 jobs, but if the same amount of waste is recycled, it can create 36 jobs (Pacheco-Torgal and Jalali, 2011). The recent report, 'Strategic Analysis of the European Recycled Materials and Chemicals Market in Construction Industry,' states that the market for recycled construction materials generated revenues of €744.1 m in 2010, and is estimated to reach €1.3 bn by 2016. However, this is a low estimate, because it does not account for the 100% C&DW re-use scenario.

Although the use of recycled aggregates (RA) in concrete has been studied for almost 50 years (Pacheco-Torgal and Jalali, 2011), modern concrete structures are still made with primary aggregates. The reasons for this stem from their low cost, low deposition taxes for C&DW, and the lack of positive discrimination toward the use of RA. Furthermore, the use of RA concrete in high-grade applications is rarely reported, because of its poorer compressive strength and high variability in mechanical behaviour (Tam *et al.*, 2005). This low performance means less durable concrete structures, which require frequent maintenance and conservation operations, or even entire replacement (with the associated consumption of more raw materials and energy).

The importance of durability in the context of eco-efficiency of construction and building materials has been ascertained by Mora (2007), who state that increasing concrete durability from 50 to 500 years would mean a reduction of its environmental impact by a factor of 10. According to Hegger *et al.* (1997), an increase in the amount of compressive strength in concrete would mean a reduction in reinforced steel by as much as 50%. These are crucial issues when considering the efficiency of materials (Pacheco-Torgal and Jalali, 2011; Allwood *et al.*, 2011), highlighting the need for studies that allow for high mechanical strength and high durability concretes capable of reusing a high volume of RA.

17.2 High performance concrete (HPC) with recycled aggregates (RAs): an overview

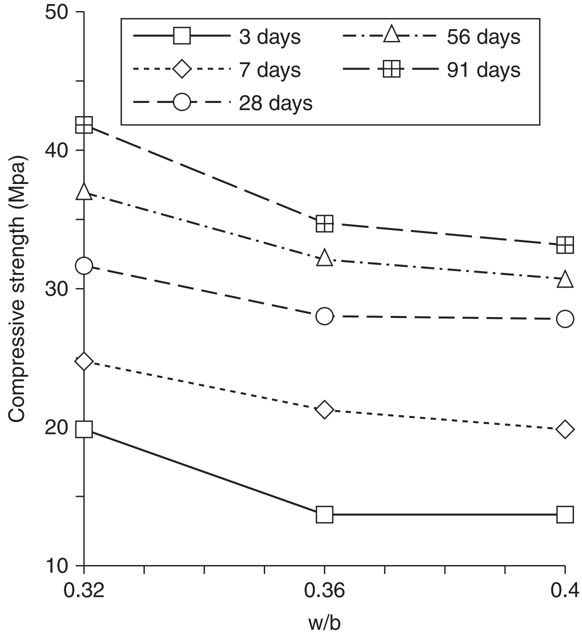
17.2.1 HPC trials

Very few studies related to the re-use of RA were able to achieve a high performance in terms of both mechanical properties and a greater resistance to chemical attack typical of HPC (Aitcin, 2003). Ajdukiewicz and Kliszczewiczv (2002) obtained 80 MPa compressive strength concretes, but used RA from an original concrete of about 60 MPa. Since the possibility of RA produced in recycled plants coming from 60 MPa used concretes is almost zero, it is unrealistic to expect that RA HPC could be produced in this way. Other authors (Tu *et al.*, 2006) also attempted (and failed) to produce such material, reporting a 28 days' compressive strength of around 30 MPa (Fig. 17.1) and concluding that, 'It is suggested not to utilize RA for high, concrete strength applications, due to long-term durability problems'.

17.2.2 Other relevant mechanical strength and durability studies

Recent studies using RA produced under laboratory conditions show that the use of fine RA must not exceed 30%, otherwise the concrete performance could be at risk. For instance, the CO_2 penetration depth increased by about 110% for concrete made solely with fine RA (Evangelista and Brito, 2007). Etxeberria *et al.* (2007) studied the performance of concrete with natural fine aggregates and different replacement percentages of coarse RA, concluding that a replacement percentage of 25% is associated to a compressive strength of almost 40 MPa.

These authors used a type I 52,5R cement, which is not cost-efficient and has a high amount of clinker. Therefore it is not obvious that the environmental advantages associated with the use of RA outweigh those of using cement with high CO_2 emissions. Berndt (2009) also studied concretes with RA, fly ash and



17.1 The compressive strength of recycled aggregates HPC versus the age and the w/b ratio (Tu *et al.*, 2006)

blast furnace slag (W/C = 0.4), obtaining a compressive strength loss of around 40 MPa. Corinaldesi and Moriconi (2009) demonstrated that it is possible to use 100% industrially produced RA (70% old concrete, 27% bricks and tiles and 3% miscellaneous (asphalt, glass, wood, paper, and other similar construction debris)) with a compressive strength of almost 45 MPa, as long as silica fume are used with a W/C of 0.4.

Those authors further mention that no organic or alkali-silica reactive (ASR) materials were detected, and the amount of chlorides and sulphates were below 0.04 and 0.15% (by weight), respectively. Xia *et al.* (2012) reviewed research concerning the mechanical property, durability and the structural performance of RA concrete that has been carried out in the past 15 years (1996–2011) in China, concluding that mechanical performance, as well as durability, is lower when compared to conventional concrete.

The poor performance of RA concrete is associated with cracks and fissures, which are formed in RA during processing, thereby rendering the aggregate weaker and more susceptible to permeation, diffusion and absorption of fluids. These drawbacks limit the utilisation of the RA with higher percentages (>30%) in structural concrete (Kou and Poon, 2012). The use of SCMs can compensate for the drawbacks associated with the use of RA; however, permeability and sorptivity of the matrix increases and the ingress of atmospheric CO_2 is facilitated, leading to an increase in concrete carbonation (Pacheco-Torgal *et al.*, 2012).

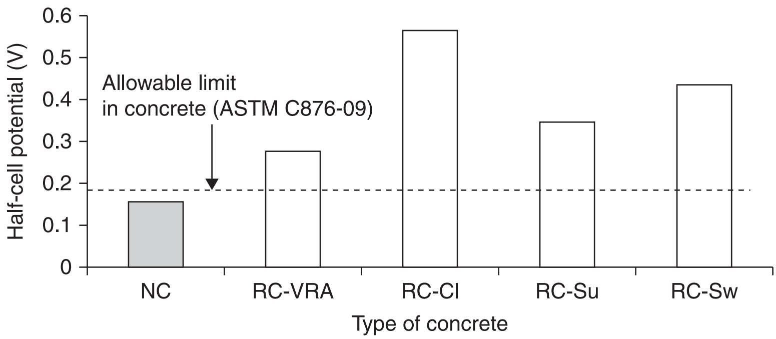
17.2.3 The problem of impurities in RAs

The majority of studies related to the re-use of C&DW in concrete use non-contaminated aggregates produced in the laboratory, making the results difficult to extrapolate when contaminated RA from recycled plants are used. Even those RA obtained from real wastes containing almost zero contamination have been previously submitted to specific treatments, which are costly and increase the environmental impact of RA. Current RA have particles of impurities such as soil, plastics, waste paper, wood, metals and organic matter. Organic matter can delay Portland cement hydration, thus leading to lower mechanical performance and lower concrete durability.

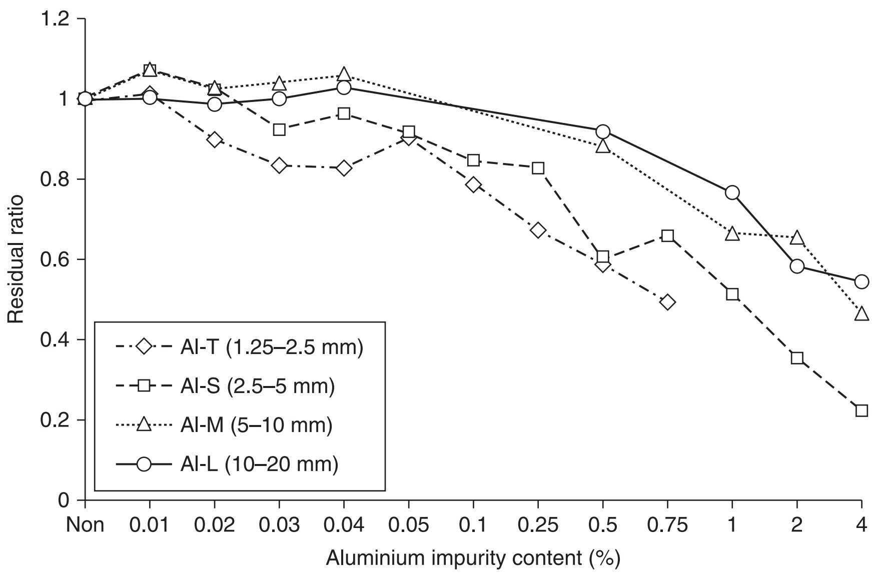
The use of RA contaminated with gypsum particles is a risk factor for concrete durability. Concrete deterioration is caused by the chemical reaction of sulphate ions with the alumina of the aggregates or with the tricalcium aluminate (C_3A) of the hardened cement paste in the presence of water; both expansion products that can lead to the cracking of concrete, which is why the regulations on C&DW limit the presence of SO_3 to less than 1%. Aggregates that come from concrete structures affected by ASR or with a high chloride or sulphate content can also be considered to be containing some kind of impurities. Algarvio (2009) studied a 50 to 80 ton/h C&DW recycling plant, noticing that contaminant percentage is very low, with wood and metals identified as the most frequent (0.048 and 0.047%, respectively).

Agrela *et al.* (2011) analysed the physical and chemical characteristics of 35 mixed RA obtained from 11 different C&DW treatment plants in Spain, noticing that 25.7% of aggregates showed 2% of gypsum. Other authors (Martin-Morales *et al.*, 2011) also detected a high sulphate content of 1.52%, which clearly exceeds the Spanish Structural Concrete Code EHE–08 upper limit of 0.80%. Debieb *et al.* (2012) studied the performance of concrete with RA contaminated by chlorides rather than sulphates, stating that the contamination did not seem to influence mechanical performance. However, they mentioned that concrete with contaminated RA is much more prone to corrosion (Fig. 17.2).

Those authors mention that precautions and specific measurements need to be taken, especially with aggregates from hazardous or critical origins such as sewage water plants, road infrastructures or buildings under marine environments. Park and Noguchi (2012) studied concrete containing metal impurities of various sizes and contents, finding that aluminum contained in RA, caused performance degradation in both mechanical properties and durability of RA concrete, even at quantities of less than 0.1% (Fig. 17.3). The chemical reaction between aluminum impurity and alkaline concrete can generate hydrogen gas which, in turn, is responsible for gas layer, foam, crack and rock pockets in hardened concrete. This leads to the significant degradation of mechanical properties of concrete, pointing



17.2 Half-cell potential of recycled reinforced concrete beams (Debieb *et al.*, 2012).



17.3 Relationship between aluminum impurity content and residual ratio of compressive strength (Park and Noguchi, 2012).

to more efficient screening methods that may increase the cost of RA, reducing its environmental advantage.

17.2.4 RA concrete standards

Different countries have different standards related to the production of RA concrete. In Portugal, the standard LNEC E 471 (2006) puts into practice the content of Directive EN 12620:2002. According to LNEC E 471, C40/50 is the

Table 17.2 Impurity content in the Japanese, British and Korean standards

Standard	Contents of impurities		Limit by mass fraction (%)
JIS A 5021 (Japan)	A	Tile, brick, ceramics, asphalt concrete lump	2.0
	B	Glass piece	0.5
	C	Gypsum, plasterboard piece	0.1
	D	Inorganic board	0.5
	E	Plastic piece	0.5
	F	Wood, wastepaper, asphalt lump	0.1
	Limit of total amount		3.0
BS 8500–2 (UK)	Maximum masonry content		5.0
	Maximum fines		5.0
	Maximum lightweight material		0.5
	Maximum asphalt		0.5
	Maximum other foreign material, e.g. glass, plastics, metals		1.0
KS F 2576 (Korea)	Wood, wastepaper, plastic piece, etc. (volume fraction (%))		1.0
	Tile, brick, ceramics, asphalt concrete lump, etc.		1.0

Source: Park and Noguchi, 2012.

maximum compressive strength class allowed for structural concrete made with RA. This standard limits the volume of RA to 25% for the C40/50 strength class, and requires that concrete and stone aggregates should be at least 90%, as well as setting upper limits for the impurity content. Glass and other undesirable particles cannot exceed 0.2%.

Other standards also set limits on the impurity content (Table 17.2). DIN 4226–100 (2002) limits the maximum content of impurities to less than or equal to 1% in terms of aggregates weight. The existing standards in this field also make the evaluation of ASR contamination mandatory, as well as leaching tests. However, these prerequisites represent a cost that will reduce the attractiveness of concrete made with RA.

17.3 Applications of HPC using RAs

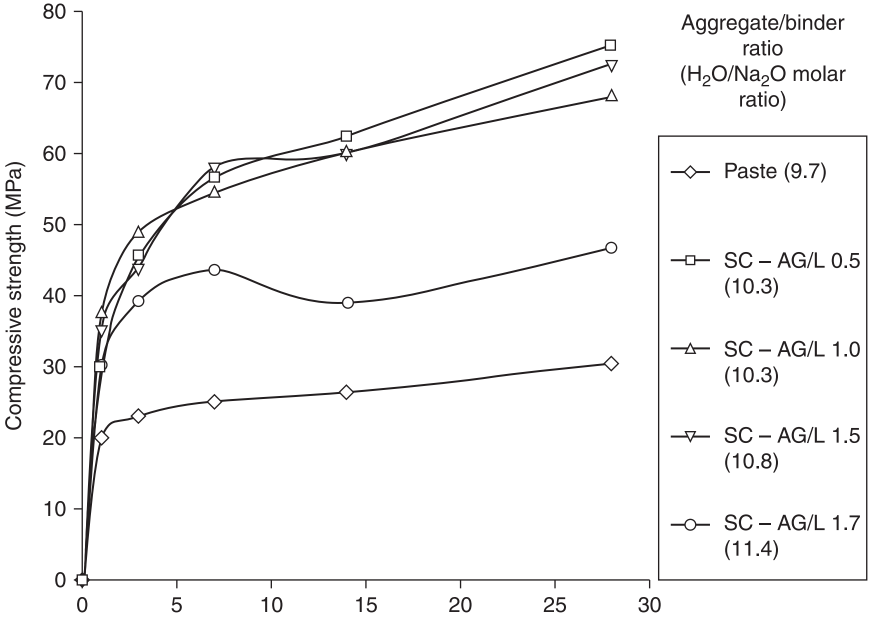
To date, studies related to the geopolymerisation of C&DW are scarce (Lampris *et al.*, 2009; Allahverdi and Kani, 2009). Nevertheless, it seems that this binder has potential features for the re-use of RA in the production of HPC. For the same water/binder ratio, several authors report that geopolymers present a higher mechanical strength than Portland cement. Wang (1991) states a case of a geopolymeric concrete with 125 MPa compressive strength. Other authors (Davidovits, 1994) declare having obtained a 20 MPa strength after just 4 h, increasing to 70 to 100 MPa after 28 days' curing time. Fernandez-Jimenez *et al.* (1999) studied mortars (w/b = 0.51) activated with NaOH and waterglass, reporting 100 MPa for compressive strength. Fernandez-Jimenez and Palomo (2005) used slag/fly ash mixtures activated with NaOH and waterglass (w/b = 0.35), announcing a 90 MPa compressive strength after just 20 h.

Bakharev (2005) studied fly ash pastes activated with NaOH and waterglass (w/b = 0.3), stating a 60 MPa compressive strength after 2 days. Other authors (Pacheco-Torgal *et al.*, 2007, 2008) report a compressive strength higher than 30 MPa after only 1 day, reaching almost 70 MPa after 28 days' curing and 90 MPa at 90 days' curing (Fig. 17.4). In conventional concrete, the aggregates form a rigid skeleton of granular elements, which are responsible for compressive strength. In geopolymers, most of the compressive strength is related to the matrix characteristics, so this material does not rely on well-proportioned aggregate mixtures.

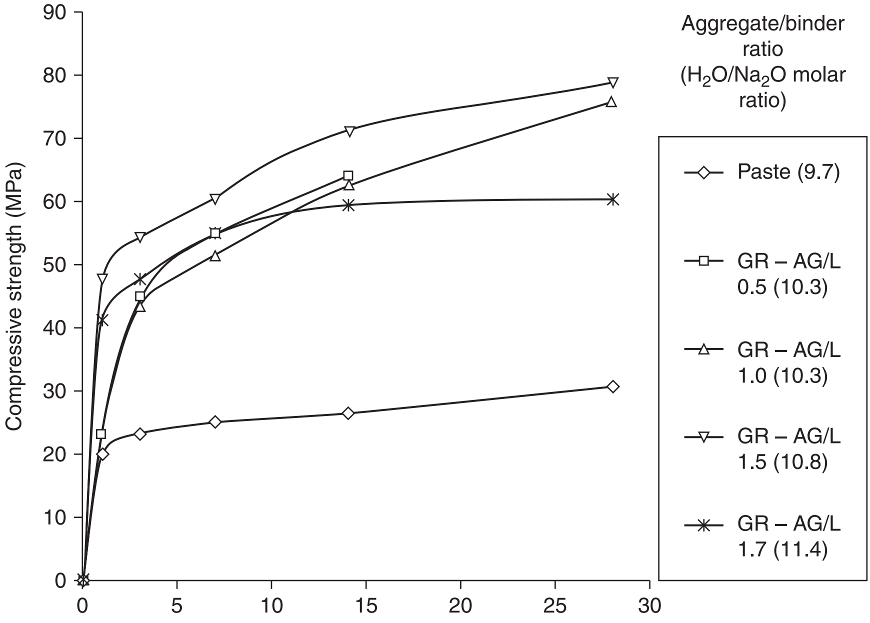
This makes geopolymeric concrete more suitable to RA. Concerning the resistance to acid attack, geopolymer performance is far better than that of Portland cement concretes because it does not contain $Ca(OH)_2$, a soluble hydration product that constitutes the 'Achilles' heel' of Portland cement concrete. Davidovits *et al.* (1990) reported mass losses of 6 and 7% for geopolymeric binders immersed in 5% concentration hydrochloric and sulphuric acids over 4 weeks. For the same conditions he also observed that Portland cement-based concretes suffered mass losses of between 78 and 95%.

Other authors (Gourley and Johnson, 2005) mentioned that a Portland cement concrete with a service life of 50 years lost 25% of its mass after 80 immersion cycles in a sulphuric acid solution (pH = 1), while a geopolymeric concrete required 1400 immersion cycles to lose the same mass, thus meaning a service life of 900 years. More recently, Pacheco-Torgal and Jalali (2010a) mentioned an average mass loss of just 2.6% after being submitted to the attack of (sulphuric, hydrochloric and nitric) acids during 28 days, while the mass loss for Portland cement concretes is more than twice that value. Geopolymers are also less susceptible to generate expansion by ASR than OPC (García-Lodeiro *et al.*, 2007), and show excellent freeze–thaw resistance (Fu *et al.,* 2011).

These materials have another advantage over Portland cement concrete that is particularly interesting in the case of re-using contaminated RA; namely a high immobilisation capacity. According to Hermann *et al.* (1999), the use of alkali-activated binders is a good way to immobilise a wide range of harmful constituents such as toxic metals, hydrocarbonates and even nuclear wastes in a final product with high durability and costing much less than the current vitrification process. Vinsova *et al.* (2007) state that alkali-activated binders show a good performance

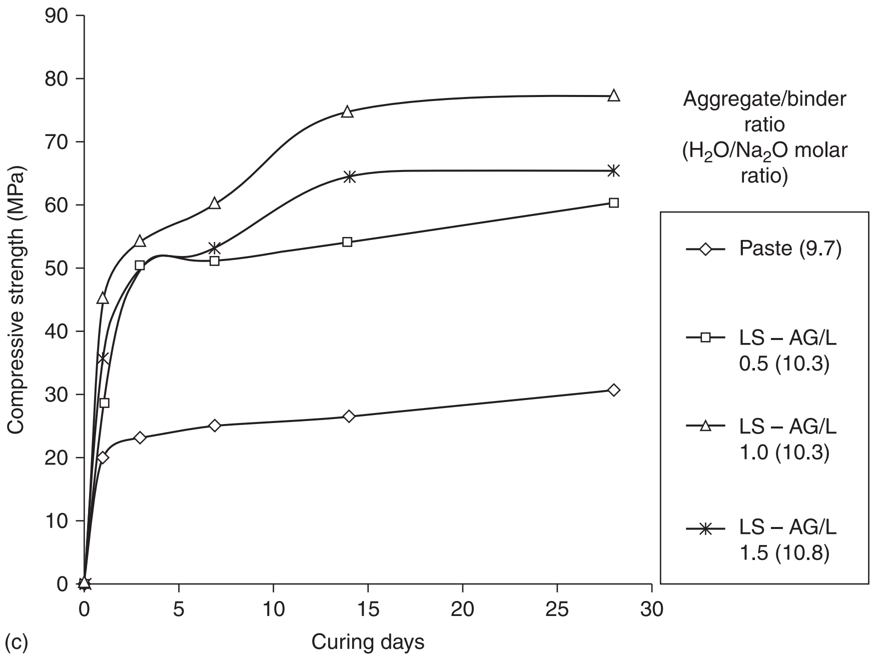


(a)



(b)

17.4 Compressive strength according to aggregate/binder mass ratio and $H2O/Na_2O$ molar ratio in AAMWM mortars made with different aggregates: (a) schist fine aggregates; (b) limestone coarse aggregates.

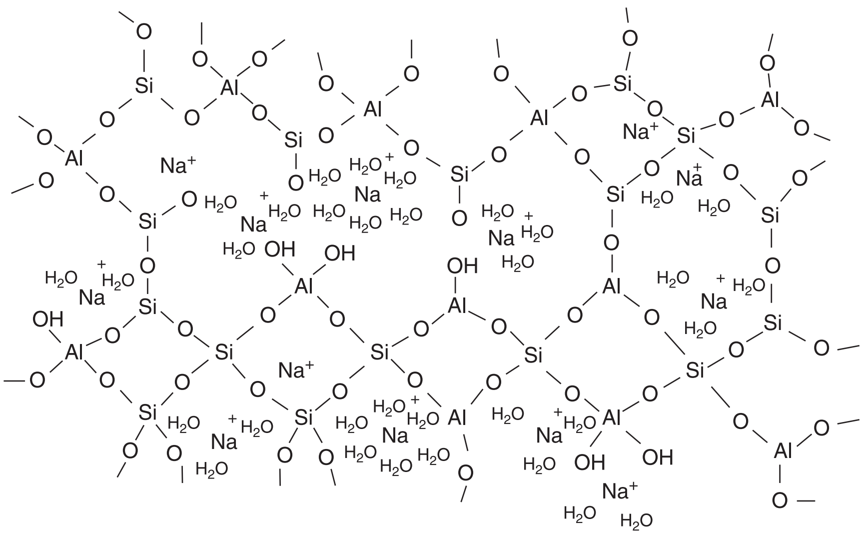


17.4 (Continued) (c) granitic coarse aggregates (Pacheco-Torgal *et al.*, 2007).

in the immobilisation of lead, cadmium and chromium, but being less effective for immobilisation of arsenic.

Lancellotti *et al.* (2010) showed that metakaolin-based geopolymer binders are able to immobilise toxic metals present in fly ash due to the incineration of municipal solid wastes. Immobilisation of a municipal solid waste incineration residue using geopolymers was recently reported (Galiano *et al.,* 2011). Other authors (Pacheco-Torgal *et al.*, 2010b; Zhang *et al.*, 2011) showed that geopolymeric binders can be used for the re-use of mine wastes. In addition, geopolymeric concretes are associated with lower CO_2 emissions than Portland cement concretes (Duxson *et al.*, 2007; Weil *et al.*, 2009; Habert *et al.*, 2011).

This is a crucial advantage, as Portland cement represents almost 80% of the total CO_2 emissions of concrete which, in turn, constitute about 6 to 7% of the planet's total CO_2 emissions (Shi *et al.*, 2011; Pacheco-Torgal *et al.*, 2012a). Nevertheless, geopolymers suffer from severe efflorescence (Pacheco-Torgal *et al.*, 2010b), because the bond between the sodium ions (Na^+) and the aluminosilicate structure is weak, thus explaining the leaching behaviour (Skvara *et al.*, 2008, 2009). According to those authors, it is the presence of water that weakens the bond of sodium in the aluminosilicate polymers, a



17.5 Geopolymer structure model (Rowles *et al.*, 2007).

behaviour confirmed by the geopolymer structure model proposed by Rowles *et al.* (2007) (Fig. 17.5). However, other authors (Temuujin *et al.*, 2009) mention that efflorescence does not occur when geopolymers are cured at elevated temperatures. This means the leachate sodium could be a sign of insufficient geopolymerisation, indicating that further investigations are needed to solve this issue.

Recently, Kani *et al.* (2011) showed that efflorescence can be reduced either by the addition of alumina-rich admixtures or by hydrothermal curing at temperatures of 65 °C or higher. These authors found that the use of 8% of calcium aluminate cement greatly reduces the mobility of alkalis, leading to minimum efflorescence.

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18
Use of construction and demolition waste (CDW) for alkali-activated or geopolymer cements

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Abstract: Alkali-activation or geopolymerisation of construction and demolition waste (CDW) provides a useful way of recycling that can form an important step towards waste management. This chapter evaluates the suitability of waste brick and waste concrete as source materials for the synthesis of alkali-activated or geopolymer cements after a comprehensive review of alkali-activation or geopolymerisation technology. The mechanical properties of the prepared geopolymer samples are suitable for use on an industrial scale. Efflorescence, however, is an issue that should be considered and controlled.

Key words: alkali-activation, geopolymerisation, construction and demolition waste (CDW), efflorescence.

18.1 Introduction

After almost 175 years of industrial production, ordinary Portland cement (PC) has become the dominant inorganic binder used in large-scale concrete structures. The PC industry is expected to expand significantly due to rapid infrastructural developments in developing countries and the annual worldwide production of PC is expected to exceed 3.5 Gt in the near future (USGS, 2011). However, such a globally accepted inorganic binder suffers from a number of disadvantages. The production process of PC is energy intensive, taking up approximately 2 to 3% of global primary energy use (Damtoft *et al.*, 2008). In addition, the PC industry is under pressure to reduce CO_2 emissions, as the production process involves approximately 0.87 tonnes of CO_2 emissions for every tonne of cement produced (Damtoft *et al.*, 2008). Other disadvantages of PC include durability problems in certain aggressive environments. All these shortcomings have prompted interest in developing low-cost, environmentally-friendly inorganic binder alternatives to PC that provide improved concrete durability and performance.

Among alternative binders, alkali-activated or geopolymer cements are attracting increasing attention in both research and practice, although they have not yet been commercialised widely. The only report on commercialisation of an alkali-activated cement in civil infrastructure projects and other areas is from

Australia under the trade name of E-crete™ (van Deventer *et al.*, 2010). Until now, a variety of industrial by-products and wastes, as well as a number of aluminosilicate raw materials, have been used in the development and production of alkali-activated or geopolymer cements. However, little effort has been devoted to the use of construction and demolition waste (CDW) as an alternative source in the development and production of alkali-activated or geopolymer cements. This chapter first presents a short but comprehensive review of alkali-activation or geopolymerisation technology and then focuses on the suitability of waste brick and concrete (two of the main constituents of CDW) as preliminary aluminosilicate materials for synthesis of alkali-activated or geopolymer cements.

18.2 The development of alkali-activated or geopolymer cements

The history of alkali-activated cements dates back to the 1940s, with the work of Purdon (1940) on the activation of BF-slag with sodium hydroxide. He also proposed a two-step mechanism for the process, involving the liberation of silica, aluminium and calcium hydroxides in the first step and the formation of silica and alumina hydrates as well as the regeneration of the alkali solution in the second step. A significant contribution to his research was then made by Glukhovsky (1965, 1959, 1978, 1980, 1981) during the 1960s until the 1980s. He identified a new class of zeolite-like hydrates (calcium and sodium aluminosilicate hydrates) as solidification products in the process of alkali treatment of rocks and clay minerals. He introduced the name 'soil cement' to the binder that he developed from ground aluminosilicate material mixed with alkali-rich industrial wastes. He also applied the name 'alkaline cement' to the binder that he developed from alkali activation of BF-slag.

The most comprehensive contribution in this field was made by Davidovits (1979), who invented a new liquid binder based on metakaolin (MK) and soluble alkali silicate in 1975, as the first mineral resin or mineral polymer ever manufactured. Then, in 1978, he applied the term 'geopolymer' to represent a broad range of materials characterised by chains or networks of inorganic molecules (Davidovits, 2008). In 1984, he developed an early high-strength geopolymer binder based on both geopolymer and PC (Davidovits, 1984). Since Davidovits' work, research in this field has increased exponentially.

The lack of a clear system of nomenclature for describing alkali-activated or geopolymer cements is a significant complication in this area. Until now, many different names have been applied to these very similar materials. In addition to soil cement, alkaline cement, mineral polymer and geopolymer, a variety of different nomenclature, including geocements (Krivenko, 1994), low-temperature aluminosilicate glass (Rahier *et al.*, 1996), alkali-activated cement (Palomo, 2003), alkali-bonded ceramic (Mallicoat *et al.*, 2005), hydrocement (Bao *et al.*, 2005) and inorganic polymer concrete (Sofi *et al.*, 2007), have been used by different researchers.

The suitability of these nomenclatures is the subject of much discussion in the literature. Some researchers do not agree with the names ‘alkaline cements’ or ‘alkali-activated cements’, because of potential confusion with PC, which is an alkaline material. Davidovits (1994a,b, 2005) also believes that these designations are not suitable and may result in confusion with alkali-silica reaction and what he introduced as real geopolymer. He also suggested the name ‘Polysialate’ for the chemical designation of geopolymers. The polysialate network is composed of silica and alumina tetrahedral anions linked and alternatively shared by oxygen atoms. Such a molecular structure can be designated by the following empirical formula:

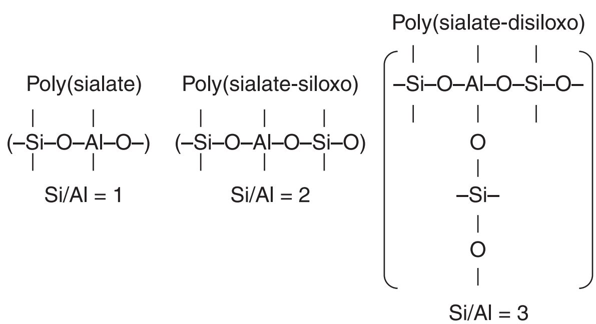
$$M_n\{-(SiO_2)_z - AlO_2\}_n, wH_2O \qquad [18.1]$$

where *n* is the degree of polymerisation, *z* is 1, 2 or 3 and *M* is an alkali cation.

The molecular structure of geopolymers, therefore, comprises the molecular units or chemical groups represented in Fig. 18.1.

According to the literature, alkali-activated cements, inorganic polymers and geopolymers can be classified and defined as follows:

- Alkali-activated cements as the broadest class of materials produced by the reaction of an alkali-activator (a solid or dissolved alkaline salt) with a solid calcium-silicate (e.g. conventional clinker) or aluminosilicate precursor (e.g. slag, pozzolan, fly ash) (Buchwald *et al.*, 2003; Shi *et al.*, 2006).
- Inorganic polymers as a subset of alkali-activated cements possessing a disordered silicate network of a higher degree of connectivity (more prevalent Q^3 and Q^4) as the primary binding phase (Duxson *et al.*, 2007a). Precursors of a higher ratio $(SiO_2 + Al_2O_3)/(CaO + Na_2O)$ than is observed in classical OPC chemistry are required for synthesising inorganic polymers.
- Geopolymers as a further subset of inorganic polymers with an almost exclusively aluminosilicate binding phase of highly coordinated, overwhelmingly Q^4 (Davidovits, 1991). The most prevalent precursors suitable for geopolymer synthesis are calcined clays and low-calcium fly ashes.



18.1 Molecular units of the molecular structure of geopolymers (Davidovits, 2005).

18.3 Mechanisms of alkali activation and properties of alkali-activated cements

18.3.1 Mechanisms for alkali activation

The general mechanism for alkali-activation was first proposed by Glukhovsky (1980), which is composed of the conjoined reactions of destruction–coagulation–condensation. Over the past two decades, the chemistry of alkali-activation has been the subject of much discussion.

The activation process starts with the breakdown of the covalent bonds Si–O–Si and Al–O–Si, forming a colloid phase. The accumulation of the destroyed products and the interactions among them result in the formation of a coagulated structure, leading to the generation of a condensed structure.

Provis *et al.* (2005a) presented a simplified model for the reaction processes involved in the transformation of a solid aluminosilicate precursor (e.g. MK) into a synthetic amorphous or nano-crystalline alkali aluminosilicate. Their model starts with the dissolution of solid aluminosilicate source in alkali-activator and oligomerisation of silicate and aluminate monomers. Depending on the processing conditions, the aluminosilicate oligomers produced could form either amorphous aluminosilicate gel upon polymerisation and gelation or nano-crystalline zeolite phase upon nucleation and crystallisation.

It has also been shown that the reaction mechanism and binder type depends on the nature of both the precursor and the alkali-activator, or more precisely on the level of calcium available for the reaction. Lower levels of calcium lead to the formation of N–A–S–(H) gel-type binder that is widely referred to as a geopolymer, that is, a highly cross-linked aluminosilicate lacking in long-range crystalline order (Davidovits, 1991; Richardson *et al.*, 1994; Wang and Scrivener, 1995; Duxson *et al.*, 2007; Bell *et al.*, 2008; Yip *et al.*, 2008). The amount of calcium also determines the type of gel being formed (Richardson *et al.*, 1994; Wang and Scrivener, 1995; Yip *et al.*, 2008). However, higher levels of calcium result in the formation of C–(A)–S–H gel-type binder, lacking in long-range order as that usually derived from the alkali-activation of BF-slag (Richardson *et al.*, 1994; Wang and Scrivener, 1995).

Considerable effort has also been devoted to the molecular structure of alkali-activated or geopolymer cements (Davidovits, 1999, 2005; Barbosa *et al.*, 2000; Lecomte *et al.*, 2003). Lecomte *et al.* (2003) compared the molecular structure of the hardened paste of PC with those of alkali-activated slag and geopolymer. Based on their studies, the C–S–H gel in PC is composed of silicate groups of mainly SiQ^1 and SiQ^2 species organised in linear finite chains. In alkali-activated slag, the C–S–H gel is formed from longer silicate chains with mid-member units of predominantly SiQ^2 and SiQ^2(1Al) types. In geopolymer materials, for example alkali-activated MK, the molecular structure is composed of a three-dimensional (3D) silicate network consisting of SiQ^4(2Al) and SiQ^4(3Al) cross-linked units.

18.3.2 Properties of alkali-activated cements

Since their invention, the set and strength behaviour and durability of alkali-activated and geopolymer cements have received increasing interest in the scientific literature. As a general conclusion, we can say that these properties depend on the composition of the starting materials as well as the processing method. A low atomic ratio of Si:Al of 1, 2 or 3 initiates a 3D-network that is very rigid, while a Si:Al ratio higher than 15 provides a polymeric character to the geopolymer material (Davidovits, 1999). There also exists an optimum level for alkali concentration in the activator, excessive amounts of which inhibit the polycondensation reactions (Komnitsas *et al.*, 2009).

Use of KOH instead of NaOH is also more beneficial for achieving higher compressive strength. This is due to the larger size of K^+ cation that favours the formation of longer silicate oligomers with which $Al(OH)^-_4$ prefers to bind (Komnitsas *et al.*, 2009). The presence of an optimum amount of dissolved silicate in the activator has also been found to be beneficial (Komnitsas *et al.*, 2009). Kaps and Buchwald (2002) showed that the addition of sodium silicate solution in the activator increases the 14-day compressive strength of a kaolinite-based geopolymer from 13 to 38 MPa. The compressive strength of geopolymer cements can also be improved by the addition of calcium compounds (specifically calcium hydroxide) to the preliminary materials (Yip *et al.*, 2008; Temuujin *et al.*, 2009).

A number of different formulations providing relatively high compressive strengths have been reported in the literature, such as geopolymer paste, mortar and concrete developed in the Czech Republic, using combinations of brown coal fly ash and BF-slag that exhibit 28-day compressive strengths up to 138, 44 and 55 MPa, respectively (Škvara, 2007).

Many researchers have concentrated on the durability of alkali-activated or geopolymer cements in aggressive environments as the key consideration regarding their application as construction materials. The resistance of alkali-activated or geopolymer cements to acid attack has been investigated in many studies (Palomo *et al.*, 1999a; Shi and Stegmann, 2000; Allahverdi and Škvara, 2001a,b, 2005; Bakharev *et al.*, 2003; Pacheco-Torgal *et al.*, 2010). Based on the results of mass and strength losses, all the researchers confirmed superior acid resistance for alkali-activated or geopolymer cements compared to PCs.

Another durability-determining property is resistance to high temperatures and to fire. PC concretes show a weak performance, including spalling when subjected to high temperatures. However, according to the published experimental results (Pawlasova and Škvara, 2007; Krivenko and Guziy, 2007; Perna *et al.*, 2007; Kong *et al.*, 2008; Temuujin *et al.*, 2011; Zhao and Sanjayan, 2011), alkali-activated or geopolymer cements demonstrate high stability when exposed to high temperatures or to fire.

Resistance to cycles of freeze-thaw is another important property determining the durability of inorganic binders. All the researchers (Yunsheng and Wei, 2006; Dolezal *et al.*, 2007; Bortnovsky *et al.*, 2007; Slavik *et al.*, 2008; Brooks *et al.*, 2010) investigating the performance of alkali-activated or geopolymer cements against cycles of freeze-thaw reported a much better resistance compared to PC.

In spite of much discussion regarding the durability properties of alkali-activated or geopolymer cements in the literature, their resistance to sulphate attack, alkali-silica reaction, carbonation and corrosion of steel reinforcement are still unproven issues requiring more investigation. A significant durability-determining property that has received little attention is efflorescence. It should be noted that alkali-activated or geopolymer cements can be prone to damage caused by efflorescence.

18.4 Applications of alkali-activated or geopolymer cements

Alkali-activated or geopolymer cements have not yet been commercialised widely. One of the major barriers to the introduction of these binders as a new and viable construction material is their relatively high cost compared to PC (Habert *et al.*, 2011). Since the most expensive component of alkali-activated concretes is the activator, the key factor in commercialising these concretes is to find a cheaper composition and dosage. Environmental impacts must also be taken into account, because many of the activators are significant sources of greenhouse gas emissions in the production of alkali-activated concretes.

The results obtained from a systematic analysis (Weil *et al.*, 2006) of a wide range of aluminosilicate raw materials used for the production of geopolymers show that MK with a cost-to-benefit factor amounting to 7 to 8 is the worst choice, considering the consumption of energy, environmental impacts and cost. Slags are less demanding and fly ashes with a cost-to-benefit factor of 0.5 can be considered as the best choice. The cost-to-benefit factor of PC determined in the same way is reported to be 1.2.

Other major barriers to the introduction of alkali-activated or geopolymer cements as new viable construction materials have been highlighted by van Deventer *et al.* (2010) as follows:

- The need for standards in each governmental jurisdiction, where the development and introduction of such documents is at best a gradual process.
- The unanswerable questions relating to the durability of concrete, given the requirement for structural concretes to last for at least several decades, where data on such timescales cannot possibly be available for a newly developed material.

In addition to the above, Davidovits (2011) highlighted the use of corrosive alkaline chemicals in the formulation of these cements as another barrier. As he reported, a comparison of patents shows that most researchers and scientists do not consider the safety of end users.

In spite of all these significant barriers, alkali-activated or geopolymer cements are targeting a small market share. According to Pacheco-Torgal *et al.* (2008a–c, 2009) alkali-activated mortars can be seven times cheaper compared to current commercial repair mortars, which provides a viable application for alkali-activated or geopolymer cements. In addition, experimental results (van Jaarsveld *et al.*, 1997; Bankowski *et al.*, 2004; Yunsheng *et al.*, 2007; Zhang *et al.*, 2008) obtained on the ability of alkali-activated or geopolymer matrices to immobilise toxic materials appear to be acceptable. A pilot-scale experimental study performed in the water treatment facility of the Wismut mine at Schlema-Alberoda, Germany (Hermann *et al.*, 1999) showed that geopolymer matrixes could be considered as a cost-efficient solution for the storage of radioactive residues under critical environmental conditions.

In spite of alkali-activated or geopolymer cements' relatively higher costs, lack of standards and unanswered questions on durability (e.g. efflorescence), there are still strong incentives to continue the research work. Global concerns over greenhouse gas emissions and an increasing interest in climate change issues are strong driving forces for developing environmentally friendly alternatives to PC.

Formulations based on industrial by-products or waste materials exhibiting latent hydraulic and/or pozzolanic properties have attracted increasing interest by those developing alternatives to PC. Re-use of these materials, including fly ash from coal combustion and granulated BF-slag from pig iron production, is not only appropriate considering global concerns over greenhouse gas emissions, but is also environmentally friendly and beneficial to waste management. This is of major importance for the further development of such cements, especially considering the growth in global annual output of fly ash from power plants, and other industrial by-products or waste materials. In the United States, about 131 million tonnes of fly ash is produced annually by 460 coal-fired power plants. An industry survey in 2008 estimated that only 43% of this ash is re-used and therefore ash piles (and their associated environmental threat) are poised to increase (Johnson, 2009). Over and above all these environmental advantages, formulations based on industrial by-products or waste materials provide more opportunities for development of low energy cost formulations compared to PC.

As previously discussed, most studies devoted to the durability of alkali-activated or geopolymer cements in different aggressive environments showed superior performance compared to PC. Considering the durability limitations of PC in certain aggressive environments, these studies have resulted in increased interest in developing more durable alternatives.

Although specific applications based on technical performance, such as repair mortars and storage of toxic wastes, are expected to become more commercialised, applications as viable construction materials still require further research to overcome the obstacles in the commercialisation of this group of inorganic binders. Based on the published literature (Roy, 1999; Shi *et al.*, 2006; van Deventer *et al.*, 2010; Habert *et al.*, 2011; Davidovits 2011), it is now possible to outline a list of key areas for future activities as research and technical prerequisites.

18.4.1 Future research trends

Future research trends include the following:

- **Improved characterisation of both appropriate source materials and hydration products**: The response of aluminosilicate materials of different sources in alkali-activation cannot be precisely predicted from the results of X-ray fluorescence on oxide contents. The behaviour of the phases present in the preliminary materials during alkali-activation, therefore, should be considered and investigated in more detail. In addition, a more detailed characterisation of hydration products necessitates application of suitable laboratory techniques.
- **Processing of raw materials**: Additional studies are needed to clarify the effects of the physical properties of the source materials, especially particle size distribution on kinetics of alkali-activation and therefore on setting time, mechanical properties and microstructure of the alkali-activated or geopolymer cements.
- **Optimised curing conditions**: Further research is needed to optimise the effects of processing parameters during hydration, such as temperature, curing ambient, and other factors.
- **Reinforcement performance**: The behaviour of alkali-activated or geopolymer cements with different reinforcements is not fully understood.
- **Soundness**: The potential and probable deterioration caused by efflorescence and relatively large drying shrinkage must be controlled.
- **Durability performance**: The lack of detailed knowledge regarding long-term performance in some aggressive environments is an important barrier to further uptake.
- **Relatively high cost**: Further research work is necessary to recognise formulations based on activators of economically suitable composition and dosage.
- **Greenhouse gas emissions**: Environmental impacts must also be taken into account by developing environmentally-friendly activators that are not significant sources of greenhouse gas emissions.
- **Safety of end users**: The safety of end users requires development of formulations based on user-friendly activators.

18.4.2 Future technical trends

Future technical trends include the following:

- **Standards**: Considerable efforts must be devoted to the evolution of performance-based standards.
- **Quality control and assurance**: The properties of alkali-activated or geopolymer cements are sensitive to variations in the chemical and physical characteristics of the source materials and these characteristics may vary from time to time or from source to source.
- **Development of databases**: Greater confidence will be gained in the manufacture and use of alkali-activated or geopolymer cements if a more extensive database is available to enhance the predictability of performance.
- **Chemical admixtures**: Current chemical admixtures on the market are suitable only for PC concrete and do not work with alkali-activated or geopolymer concretes.

18.5 Precursors for alkali-activated or geopolymer cements

The majority of the work on alkali-activated or geopolymer cements has been focused on the activation of MK, granulated BF-slag and fly ashes. Since these cements form by the co-polymerisation of dissolved silica and alumina in alkali-activator, any source of silica and alumina that is easily dissolved in high pH alkaline medium can be considered as a suitable precursor for synthesis of alkali-activated or geopolymer cements. This means that alkali-activated or geopolymer cements can be synthesised from a wide range of artificial or natural aluminosilicate materials. However, a review of the literature shows that alkali-activated or geopolymer cements synthesised from calcined (dehydroxylated) source materials exhibit higher compressive strengths than those synthesised from raw materials (Xu and van Deventer, 2000b; Valeria *et al.*, 2000; van Jaarsveld *et al.*, 2000).

18.5.1 Industrial slags

Attempts for alkali-activation of industrial slags began in the 1940s (Purdon, 1940) by activating BF-slag. Systematic studies were then continued in the 1970s and 1980s by Glukhovsky (1978, 1981). Recently, studies on hydration, mechanical properties and durability of alkali-activated BF-slag cements have been increased extensively due to their potential environmental, energetic and technical performance benefits (Brough and Atkinson, 2002; Al-Otabi, 2008; Bougara *et al.*, 2010; Hubler *et al.*, 2011). According to recent comprehensive review studies (Shi and Qian, 2000; Shi *et al.*, 2011), mortars and concretes of alkali-activated BF-slag cements with mechanical properties and durability performance comparable to or better than those of PC can be produced through the proper selection of activator and curing conditions. Shi and Qian (2000) and Shi *et al.* (2011) summarised the major features of alkali-activated BFS cements as follows:

- The main hydration product formed in all cases is C–S–H with a low C/S ratio. Few minor products, depending on the nature of the activator, may also be present, but there are no free $Ca(OH)_2$.
- The performance of the cement is governed primarily by the nature of the slag and the nature and dosage of the activator used.
- In moist conditions, alkali-activated BFS cement is less permeable to water and chlorides and therefore more resistant to corrosive media such as acids, sulphates or chlorides than conventional PCs.

The most commonly used slag in the development and production of alkali-activated slag cements is BF-slag, which is high in calcium oxide with moderate quantities of silica and alumina. However, studies performed on other industrial slags (Shi and Li, 1989; Wang *et al.*, 1994; Maragkos *et al.*, 2009) confirm that some different slags, including electrothermal phosphorous and Ferronickel slags, can also be used in the development of alkali-activated slag cements.

18.5.2 Metakaolin (MK)

MK is a thermally activated CaO-free aluminosilicate material, exhibiting high pozzolanic activity. It is produced by calcination of kaolinitic clay at temperatures ranging between 650 and 800 °C, depending on the purity and crystallinity of the precursor clays (Pera, 2001). As mentioned previously, alkali-activation of MK and the term 'geopolymer' were first introduced by Davidovits (1979, 2008) in 1978. Alkali-activation of MK is a way of producing a binding material of excellent cementing properties in terms of mechanical strength (Palomo *et al.*, 1999a). The hydration products of alkali-activated MK are different to those reported for alkali-activated BF-slag. It is generally accepted that the main binding phase in alkali-activated BF-slag is C–S–H, whereas the binding properties of alkali-activated MK are assumed to be the result of the formation of an amorphous N–A–S–H gel (Granizo *et al.*, 1997, 2002; Davidovits, 1999).

The chemical composition of the geopolymer mixture and the curing temperature are the most important factors influencing the properties of the geopolymer cements. Duxson *et al.* (2005) studied the relationship between composition, microstructure and mechanical properties in samples of MK activated with sodium silicate. They reported that the Si/Al atomic ratio of the binder plays a key role with respect to the microstructure and compressive strength. According to their study, ratios below 1.40 yield microstructures with clustered dense particulates and large interconnected pores, whereas ratios above 1.65 develop homogenous dense microstructures exhibiting significantly higher compressive strengths.

Although the effect of curing temperature on geopolymer properties has not yet been completely studied, it is known that curing above ambient temperatures favours the development of high early strengths (Barbosa and MacKenzie, 2003a,b).

18.5.3 Fly ash

Fly ash refers to the fine ash particles that arise with flue gases during coal combustion. Since coal is a major global energy source, large quantities of fly ash (up to several hundred million tonnes) are produced worldwide. Traditionally, fly ash has been used as a supplementary cementing material to enhance concrete properties. However, only a proportion of fly ash is currently reused for this purpose, for a variety of different applications, and most is still dumped as waste in ponds and landfills (Kim and Prezzi, 2008). The useful deployment of this industrial waste, therefore, has been the subject of numerous studies.

According to ASTM C618, fly ash is classified into two classes; class F and class C. Class F is produced from burning anthracite and bituminous coals. It contains less than 20% CaO and exhibits pozzolanic properties. However, Class C fly ash is produced from lignite and sub-bituminous coals and contains more than 20% CaO. In addition to pozzolanic properties, Class C fly ash has some self-cementing properties (Taylor, 1997). Since both types contain reasonable amounts of silica and alumina, they have been considered as important source materials for developing alkali-activated or geopolymer cements, and many papers have been published on this issue (van Jaarsveld and van Deventer, 1999; Palomo *et al.*, 1999b; Xu and van Deventer, 2000a; Brough *et al.*, 2001; Swanepoel and Strydom, 2002; Krivenko and Kovalchuk, 2002a,b; Fernandez-Jimenez and Palomo, 2003; Fernandez-Jimenez *et al.*, 2005; Palomo *et al.*, 2004; Bakharev, 2005; Škvara *et al.*, 2005a; Provis *et al.*, 2005b; Panias *et al.*, 2007; Rattanasak and Chindaprasirt, 2009).

Fly ash particles are generally spherical and comprise glassy as well as crystalline (often mullite and quartz) phases. According to Fernandez-Jimenez *et al.* (2005), the dissolution process of Si and Al starts as soon as fly ashes are submitted to the alkaline solution. The molecules of the reaction product condense and the alkali attack opens the particles, exposing small particles on the inside. The bi-directional alkaline attack, therefore, continues both from the outside in and from the inside out, and consequently the reaction product is generated both inside and outside the shell of the sphere, until the ash is completely or partially consumed.

Depending on the mix composition and curing regime, the main hydration products are gel-like sodium aluminosilicate hydrates and zeolite-like hydroxysodalite (Palomo *et al.*, 1999b; Brough *et al.*, 2001; Swanepoel and Strydom, 2002; Bakharev, 2005; Škvara *et al.*, 2005a; Provis *et al.*, 2005). According to Fernandez-Jimenez and Palomo (2003), original crystalline phases are unchanged by the activation reactions. They found zeolitic phases such as hydroxysodalite ($Na_4Al_3Si_3O_{12}OH$) and herschelite ($NaAlSi_2O_6 \cdot 3H_2O$) in the XRD patterns of alkali-activated fly ashes. Krivenko and Kovalchuk (2002a) also reported the formation of zeolite-like analcime and hydroxysodalite for fly ashes activated with NaOH and waterglass. The microstructure of the hardened paste of alkali-activated or geopolymer cements based on fly ash, therefore, consists of aluminosilicate gel, unreacted or partially reacted fly ash particles, and zeolite-like crystalline phases.

The type and concentration of the activator play important roles in the geopolymerisation process. The most commonly used alkali-activators are NaOH and KOH. Leaching of alumina and silica from fly ash particles is generally high with sodium hydroxide solution compared to potassium hydroxide solution (Panias *et al.*, 2007; Rattanasak and Chindaprasirt, 2009). The concentration of sodium hydroxide in the activator significantly affects the dissolution of fly ash particles, as well as their bonding in the final microstructure (Panias *et al.*, 2007, Rattanasak and Chindaprasirt, 2009). The addition of soluble silicate (sodium silicate solution) to the sodium hydroxide solution also enhances the reaction between the source material and the alkali-activator (Palomo *et al.*, 1999b; Xu and van Deventer, 2000a).

Since alkali-activated or geopolymer cements based on fly ashes harden slowly at ambient temperature, they are usually subjected to hydrothermal curing at elevated temperatures between 40 and 95 °C (van Jaarsveld and van Deventer, 1999; Palomo *et al.*, 1999b, 2004; Krivenko and Kovalchuk, 2002a,b; Fernandez-Jimenez and Palomo, 2003).

18.5.4 Miscellaneous materials

A number of other aluminosilicate source materials, including uncalcined clays (Zhang *et al.*, 2004), tungsten mine waste (Pacheco-Torgal *et al.*, 2008a,b,c, 2009, 2010), natural pozzolans (Verdolotti *et al.*, 2008; Najafi Kani and Allahverdi, 2009a; Miller *et al.*, 2010; Allahverdi *et al.*, 2011; Najafi Kani *et al.*, 2012b) and natural zeolite (Villa *et al.*, 2010), have been studied and evaluated as alternative precursors. Some researchers have also investigated the suitability of composite source materials including fly ash and granulated BF-slag (Puertas *et al.*, 2003a; Puertas and Fernandez-Jimenez, 2003b), PC and granulated BF-slag (Sajedi and Abdul Razak, 2010; Guerrieri *et al.*, 2010), granulated BF-slag and MK (Nocun-Wczelik, 2006; Bernal *et al.*, 2011) and granulated BF-slag and red mud (Pan *et al.*, 2002, 2003) as precursors. There still exist different aluminosilicate materials, especially those among industrial wastes, for example CDW, that are worth being evaluated as suitable precursors for production of alkali-activated or geopolymer cements.

The management of CDW has become an important worldwide issue, due to the large volumes generated annually in both advanced and developing countries. Up until recently, the common tendency in the industry was to discard CDW in landfills and even in open dumps. However, this practice is not environmentally sound, due to impacts including soil and water contamination, air pollution as a result of fires, destruction of the landscape and open spaces, and reduced land value (El-Haggar Salah, 2007). CDW management, therefore, needs greater emphasis to be placed upon prevention and reduction of waste generation, followed by recycling, reusing and/or energy recovery (Mercader *et al.*, 2010). Consequently, the development of alkali-activated or geopolymer cements based on CDW is a useful way of recycling and reusing materials, forming an important step towards successful waste management and control.

Waste generated by construction and demolition activities typically includes a broad range of materials, from concrete, stones, bricks, blocks, gypsum wallboard, plaster and glass, to steel, lumber, shingles, plumbing, asphalt roofing, heating and electrical parts (Gavilan and Bernold, 1994; Seo and Hwang, 1999). Despite factors that will affect the composition of CDW in the future, it is likely that inorganic non-metallic materials will continue to form the largest component. A high proportion of this component, and particularly the fraction derived from bricks, tiles, glass and concrete, are well suited to being processed and utilised as new alternative precursors for production of alkali-activated or geopolymer cements.

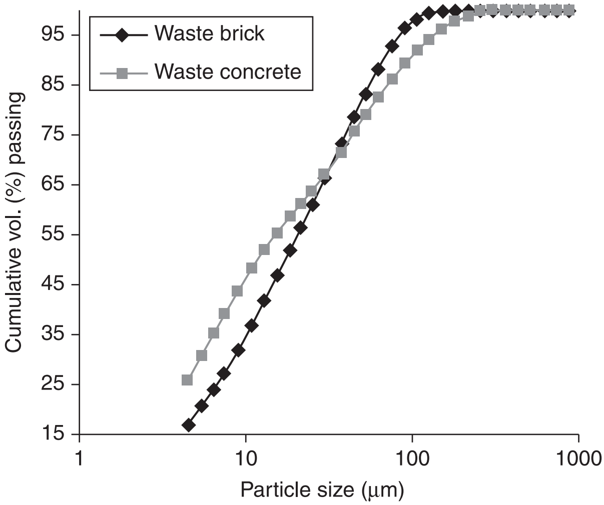
The next section of this chapter is devoted to an experimental study on the suitability of waste brick and concrete as preliminary aluminosilicate materials for alkali-activation or geopolymerisation. The study presented here is a preliminary work on this issue and more detailed investigations are necessary to evaluate CDW constituents as alternative precursors for alkali-activated or geopolymer cements.

18.6 The development of alkali-activated or geopolymer cements based on construction and demolition waste

This section first covers some studies on the potential of utilising CDW consisting of waste brick and concrete as raw materials in the production of alkali-activated or geopolymer cements, and second on the engineering properties of the produced binders. For this purpose, the most effective parameters on the synthesis of geopolymers will be investigated.

18.6.1 Materials characterisation

Waste brick and concrete were used as raw materials during an investigation into the possibility of utilising CDW as precursors in the production of an inorganic polymeric binder. The samples were crushed and ground using laboratory crushers and ball mills. The prepared materials were then characterised by determining their chemical composition and particle size distribution. The results of wet chemical analysis are presented in Table 18.1. The particle size distribution of the waste brick and concrete powders was determined by a laser particle size analyser (Malvern Mastersizer, 2000) and is presented in Fig. 18.2. The mean particle sizes of the waste brick and concrete were 26.74 μm and 24.17 μm, respectively.

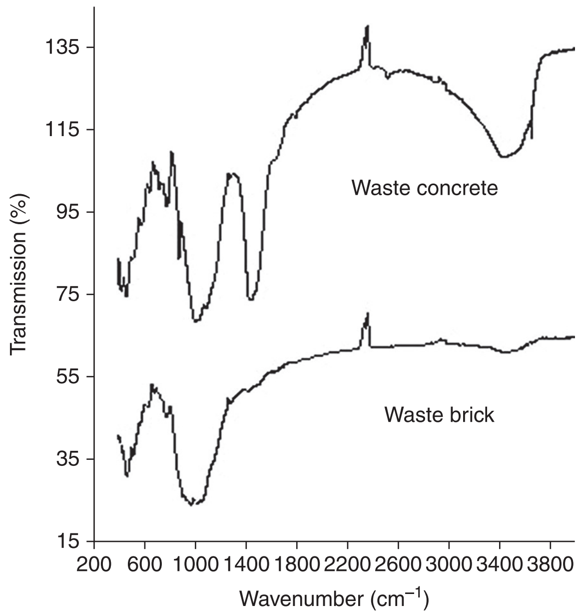


18.2 Particle size distributions of ground waste brick and concrete.

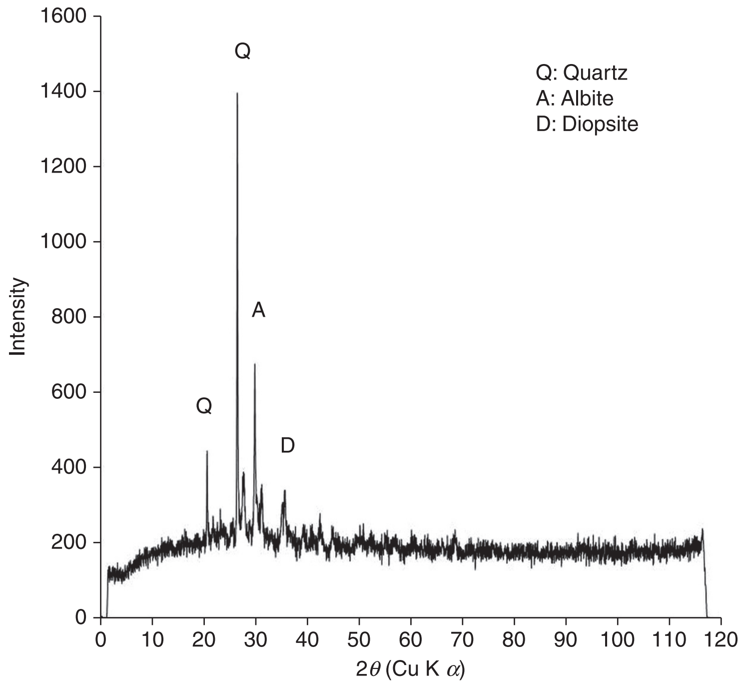
Table 18.1 Chemical composition of the waste brick and concrete (wt%)

Oxide	SiO_2	Al_2O_3	Fe_2O_3	CaO	MgO	K_2O	Na_2O	LOI
Waste brick	58.62	9.13	7.75	19.81	1.36	2.10	1.55	0.68
Waste concrete	40.05	8.60	10.36	25.14	1.85	2.27	1.80	9.85

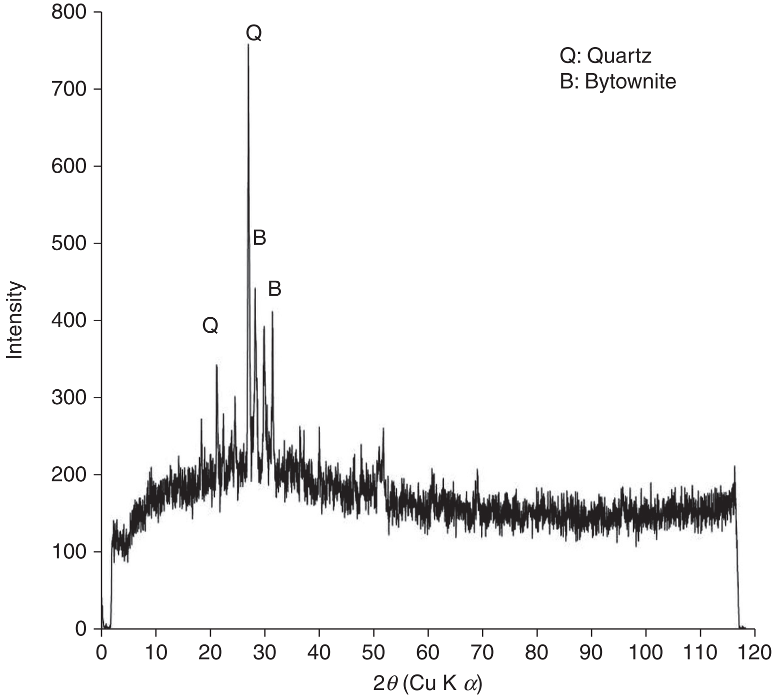
The Fourier Transform Infrared (FTIR) spectra of the used brick and concrete are shown in Fig. 18.3. FTIR spectra were collected using a Nicolet 740 FTIR spectrometer in transmittance mode from 400 to 4000 cm^{-1} using the standard KBr technique (0.5 mg sample with 250 mg KBr). All spectra were obtained with a sensitivity of 4 cm^{-1} and 64 scans per spectrum taken. As seen, the FTIR spectra of the waste brick and concrete show two strong peaks, one at wavenumber about 460 cm^{-1} and the other a broad peak at wavenumbers about 1000 cm^{-1}, which are both attributed to asymmetric stretching of Al–O and Si–O bonds of aluminosilicate structure. Figures 18.4 and 18.5 show the XRD pattern (Philips X'pert diffractometer, CuKα radiation, 2 degree/min, divergence and anti-scatter slits 1 degree each, receiving slit 0.01 mm) of the used brick and concrete. The crystalline mineral phases present in waste brick include the minerals quartz, albite and diopsite, and the crystalline mineral phases present in concrete waste include the minerals quartz and bytownite. Industrial sodium silicate solution (mass ratio $SiO_2/Na_2O = 0.86$ and SiO_2 content of 34.32 wt%) and industrial-grade NaOH (99% purity) were used throughout all experiments.



18.3 FTIR spectra of waste brick and concrete.



18.4 X-ray diffraction pattern of waste brick.



18.5 X-ray diffraction pattern of waste concrete.

18.6.2 Experimental procedure

Mixtures of waste brick and concrete as dry binders, categorised into the seven groups given in Table 18.2, were homogenised using a domestic grinder (type SANA SCG-3001). Enough sodium hydroxide was added to sodium silicate for preparing the alkali-activator, having different silica modulus (mass ratio SiO_2/Na_2O) of 0.6, 1.0 and 1.4. The sodium oxide contents of the designed activated mixes were adjusted at three different levels of 3, 5 and 7% (by weight of dry binder). The water-to-dry binder ratios (W/DB-ratios) were adjusted in the range between 0.28 and 0.34 for an approximately constant consistency of about 170 mm spread diameter measured by the flow-table test in accordance with ASTM C230. The designed formulations for activators are presented in Table 18.3.

After adding the alkali-activators to the dry binders and enough mixing, the prepared pastes were cast into 5 cm cubic moulds, and cured at around 95% relative humidity (RH) to minimise drying, giving enough time (24 h) before demoulding for the pastes to set and harden at ambient temperature (25 °C). After demoulding, the specimens were cured in an atmosphere of 95% relative humidity at 25 °C until testing time for compressive strength measurement. From each mix,

Table 18.2 Mixture proportions of waste brick and concrete

Mix name	Composition	
	Brick	Concrete
M1	100	0
M2	90	10
M3	80	20
M4	70	30
M5	60	40
M6	50	50
M7	40	60

Table 18.3 The used activators

Activator no.	Activator formulation	
	Silica modulus (Ms)	Na_2O%
1	0.6	3
2	0.6	5
3	0.6	7
4	1	3
5	1	5
6	1	7
7	1.4	3
8	1.4	5
9	1.4	7

three specimens were used to determine 28-day compressive strength, using a Toni Technic instrument (Toni Technic, Germany).

To qualitatively investigate the severity of efflorescence, one 28-day hardened 2 cm cube paste specimen from each mix was immersed in 40 mL of distilled water, and kept in an open-air atmosphere at ambient temperature (25 °C) until the water was dried completely. The results of the efflorescence test were obtained qualitatively by visual comparison of the specimens.

The mixes were characterised by measuring their final setting time and 28-day compressive strength. Final setting time of all the mixes was measured using a Vicat needle in accordance with ASTM standard C191. Samples of waste brick and concrete and hardened paste of the mix exhibiting the highest 28-day compressive strengths were characterised by laboratory techniques of Fourier transform infrared spectroscopy (FTIR, Nicolet 740), XRD pattern (Philips X'pert diffractometer) and scanning electron microscopy (SEM, CamScan MV 2300) for studying molecular, crystalline phases and microstructure of the materials. For

SEM studies, a number of 28-day hardened specimens were cut into halves to expose internal regions. Suitable halves were then coated with gold before SEM studies. The following sections discuss some of the key results.

18.6.3 Setting time

Table 18.4 shows the effects of Na_2O content, W/DB-ratio and silica modulus on final setting time of the mixes. As can be seen, the final setting time of mixes varies in the range between 67 and 550 min. Such final setting times could be practically acceptable compared to normal values reported for ordinary PC. In this

Table 18.4 Effects of Na_2O content, W/DB-ratio, and silica modulus on final setting time

Mix name	Na_2O (wt%)	Ms	W/DB	Final setting time (min)	Mix name	Na_2O (wt%)	Ms	W/DB	Final setting time (min)
M1-1	3	0.6	0.34	481	M4-6	7	1	0.31	230
M1-2	5	0.6	0.34	460	M4-7	3	1.4	0.30	174
M1-3	7	0.6	0.34	352	M4-8	5	1.4	0.30	183
M1-4	3	1	0.33	301	M4-9	7	1.4	0.30	213
M1-5	5	1	0.33	332	M5-1	3	0.6	0.32	152
M1-6	7	1	0.33	492	M5-2	5	0.6	0.32	140
M1-7	3	1.4	0.32	400	M5-3	7	0.6	0.32	103
M1-8	5	1.4	0.32	511	M5-4	3	1	0.31	110
M1-9	7	1.4	0.32	550	M5-5	5	1	0.31	122
M2-1	3	0.6	0.33	392	M5-6	7	1	0.31	151
M2-2	5	0.6	0.33	378	M5-7	3	1.4	0.29	67
M2-3	7	0.6	0.33	297	M5-8	5	1.4	0.29	109
M2-4	3	1	0.32	292	M5-9	7	1.4	0.29	145
M2-5	5	1	0.32	307	M6-1	3	0.6	0.32	171
M2-6	7	1	0.32	3395	M6-2	5	0.6	0.32	139
M2-7	3	1.4	0.32	271	M6-3	7	0.6	0.32	121
M2-8	5	1.4	0.32	309	M6-4	3	1	0.31	117
M2-9	7	1.4	0.32	372	M6-5	5	1	0.31	130
M3-1	3	0.6	0.33	303	M6-6	7	1	0.31	163
M3-2	5	0.6	0.33	270	M6-7	3	1.4	0.29	92
M3-3	7	0.6	0.33	192	M6-8	5	1.4	0.29	99
M3-4	3	1	0.32	220	M6-9	7	1.4	0.29	135
M3-5	5	1	0.32	252	M7-1	3	0.6	0.31	168
M3-6	7	1	0.32	317	M7-2	5	0.6	0.31	153
M3-7	3	1.4	0.32	171	M7-3	7	0.6	0.31	138
M3-8	5	1.4	0.32	202	M7-4	3	1	0.30	130
M3-9	7	1.4	0.32	241	M7-5	5	1	0.30	142
M4-1	3	0.6	0.32	191	M7-6	7	1	0.30	178
M4-2	5	0.6	0.32	182	M7-7	3	1.4	0.30	111
M4-3	7	0.6	0.32	154	M7-8	5	1.4	0.28	127
M4-4	3	1	0.31	181	M7-9	7	1.4	0.28	155
M4-5	5	1	0.31	190					

case, the setting time and almost all of the other mechanical properties of the products can be influenced by two groups of parameters – one being the proportion of waste brick and concrete, and the second the chemical composition of the used alkali-activator. It is seen that at constant silica modulus, the final setting time reduces when the proportion of concrete is increased. Also, at the lower value of silica modulus, that is Ms = 0.6, the final setting time increases when the Na_2O concentration is increased. Although the chemical composition of the dry mixes is different here, the trend in setting time when Ms = 0.6 is in good compatibility with our previous work on construction waste (Allahverdi and Najafi Kani, 2009).

At the higher values of silica modulus, that is Ms = 1 and 1.4, the final setting time reduces when the Na_2O concentration is increased. When the silica modulus is low, any increase in the Na_2O concentration could accelerate activation reactions, which in turn results in a significant decrease in final setting time. However, when the silica modulus is high, the increase in the Na_2O concentration has less effect on acceleration of activation reactions. In this case, the dry mix proportion is more relevant in affecting the setting time. A comparison of the results clearly shows the presence of harmonic variations between final setting time and the other variables including W/DB-ratio and mix proportion. It is reasonable to expect that any increase in W/DB ratio results in an increase in the final setting time. The time of setting could be considered as an indication of the kinetics of the very first reactions of geopolymerisation, which leads to polycondensation and hardening of the produced gel. The lowest setting time here is related to the mix M5-7 with 60% of waste concrete and 40% of waste brick as dry binder with Na_2O content of 3% and Ms of 1.4. The highest setting time related to the mix M1-9 with 100% of waste brick as dry binder, with Na_2O content of 7% and Ms of 1.4.

18.6.4 Compressive strength

Before measuring 28-day compressive strength, the specimens were observed visually for possible cracking. They were sound and no cracks were observed. Table 18.5 shows the 28-day compressive strength of different studied mixes. As can be seen, in all mixes the strength increases by increasing Na_2O concentration. However, mixes with relatively higher proportions of waste concrete, that is M5, 6 and 7, with higher amounts of silica modulus, that is Ms = 1.0 and 1.4, show much stronger strength development with any increase in the Na_2O concentration.

The maximum achievable 28-day compressive strength is 50 MPa for mix M5-8 comprising of 60% waste concrete and 40% waste brick and containing 8% Na_2O by weight of dry binder with silica modulus of 1.4. In the studied dry binder, not only is the composition of the dry binder, that is combination of waste brick and concrete, not constant, but also all of the changes made in sodium hydroxide

Table 18.5 28-day Compressive strength of the prepared mixes

Mix name	Compressive strength (MPa)	Mix name	Compressive strength (MPa)	Mix name	Compressive strength (MPa)	Mix name	Compressive strength (MPa)
M1-1	3.2	M2-8	25.0	M4-6	30.1	M6-4	37.2
M1-2	5.0	M2-9	23.2	M4-7	35.2	M6-5	40.0
M1-3	5.0	M3-1	8.0	M4-8	40.0	M6-6	35.5
M1-4	15.0	M3-2	10.0	M4-9	39.5	M6-7	46.8
M1-5	17.5	M3-3	10.0	M5-1	12.3	M6-8	47.5
M1-6	16.5	M3-4	26.3	M5-2	15.0	M6-9	48.0
M1-7	15.3	M3-5	30.0	M5-3	15.5	M7-1	10.2
M1-8	16.0	M3-6	28.0	M5-4	30.1	M7-2	12.5
M1-9	16.1	M3-7	30.7	M5-5	33.8	M7-3	9.6
M2-1	11.2	M3-8	33.8	M5-6	29.7	M7-4	33.7
M2-2	12.5	M3-9	33.0	M5-7	45.2	M7-5	37.5
M2-3	13.0	M4-1	11.1	M5-8	50.0	M7-6	31.5
M2-4	13.1	M4-2	12.5	M5-9	46.7	M7-7	39.2
M2-5	15.0	M4-3	10.0	M6-1	15.2	M7-8	47.5
M2-6	14.3	M4-4	29.3	M6-2	18.5	M7-9	47.0
M2-7	21.7	M4-5	32.5	M6-3	19.0		

and sodium silicate contents of the activator cause effective changes in the SiO_2/Na_2O, SiO_2/Al_2O_3, and Na_2O/Al_2O_3 molar ratios of the produced materials.

Both sodium hydroxide and sodium silicate play significant roles in geopolymerisation reactions. Sodium hydroxide provides both hydroxide anion (OH^-), which is important for dissolution of the aluminosilicates in the first stage, and sodium cation (Na^+), which is important for charge balance of the aluminosilicate network formed in the last stage. Solubility of aluminosilicate increases with increasing OH^- concentration. However, using too much sodium hydroxide in preparation of the alkali-activator is not beneficial. Excess OH^- could cause the polycondensation reactions to occur not only faster, but also too soon, therefore hindering dissolution of the aluminosilicates, which need enough time to proceed. Such a condition leads to formation of a material with a short setting time due to rapid polycondensation reactions, and relatively lower compressive strength due to a not quite matured molecular structure caused by incomplete dissolution of aluminosilicates (Allahverdi *et al.*, 2011; Najafi Kani and Allahverdi, 2009b; Khale *et al.*, 2007; Komnitsas *et al.*, 2007). Excess Na^+ remaining unreacted in the geopolymer matrix could be simply leached out or result in the formation of efflorescence due to accumulation of carbonated sodium oxide on the surface of the material.

Sodium silicate also provides good inter-particle bonding and therefore mechanical strength of the material by synthesising aluminosilicate gel. In addition, it hinders water evaporation and provides suitable conditions for structure formation (Khale and Chaudhary, 2007). The properties of the geopolymer material therefore strongly depend on its chemical composition, and to develop a sound geopolymer cement exhibiting acceptable set and strength behaviour, it is necessary to determine the optimum chemical composition.

Hydrothermal curing could show a positive effect on compressive strength of the studied inorganic polymeric binders. Any increase in both time and temperature of curing causes an increase in compressive strength. Long precuring at an ambient temperature of more than 95% relative humidity at room temperature, before hydrothermal treatment, is beneficial for higher strength development in the studied inorganic polymeric binders. Hydrothermal curing could effectively improve the microstructure of the specimens by preventing the formation of micro-cracks and hence increasing the compressive strength (Najafi and Allahverdi, 2009a).

18.6.5 Efflorescence

From each mix, a 28-day hardened paste specimen was tested to investigate the severity of the efflorescence. The results of the efflorescence test were obtained qualitatively by simply comparing the specimens visually. Table 18.6 presents the results obtained. The severity of the efflorescence has been differentiated by letters A, B and C. Mixes exhibiting no efflorescence are shown by the letter A. Those showing slight and severe efflorescence are distinguished by letters B and C, respectively. As seen in Table 18.6, most of the mixes containing 3 wt% Na_2O do not show any efflorescence. Some mixes having higher silica modulus

Table 18.6 Severity of efflorescence in the prepared mixes

Mix name	Efflorescence severity	Mix name	Efflorescence severity	Mix name	Efflorescence severity	Mix name	Efflorescence severity
M1-1	A	M2-8	B	M4-6	B	M6-4	A
M1-2	B	M2-9	C	M4-7	A	M6-5	B
M1-3	B	M3-1	A	M4-8	A	M6-6	C
M1-4	B	M3-2	B	M4-9	A	M6-7	A
M1-5	C	M3-3	B	M5-1	A	M6-8	A
M1-6	C	M3-4	A	M5-2	B	M6-9	A
M1-7	B	M3-5	A	M5-3	C	M7-1	A
M1-8	B	M3-6	B	M5-4	A	M7-2	B
M1-9	B	M3-7	A	M5-5	A	M7-3	B
M2-1	B	M3-8	A	M5-6	C	M7-4	A
M2-2	C	M3-9	A	M5-7	A	M7-5	B
M2-3	C	M4-1	B	M5-8	A	M7-6	B
M2-4	B	M4-2	B	M5-9	A	M7-7	A
M2-5	B	M4-3	B	M6-1	A	M7-8	A
M2-6	C	M4-4	A	M6-2	A	M7-9	B
M2-7	B	M4-5	A	M6-3	B		

(Ms = 1.4) exhibit moderate or no efflorescence. Any increase in the proportion of waste concrete in dry mix results in a decrease of efflorescence severity.

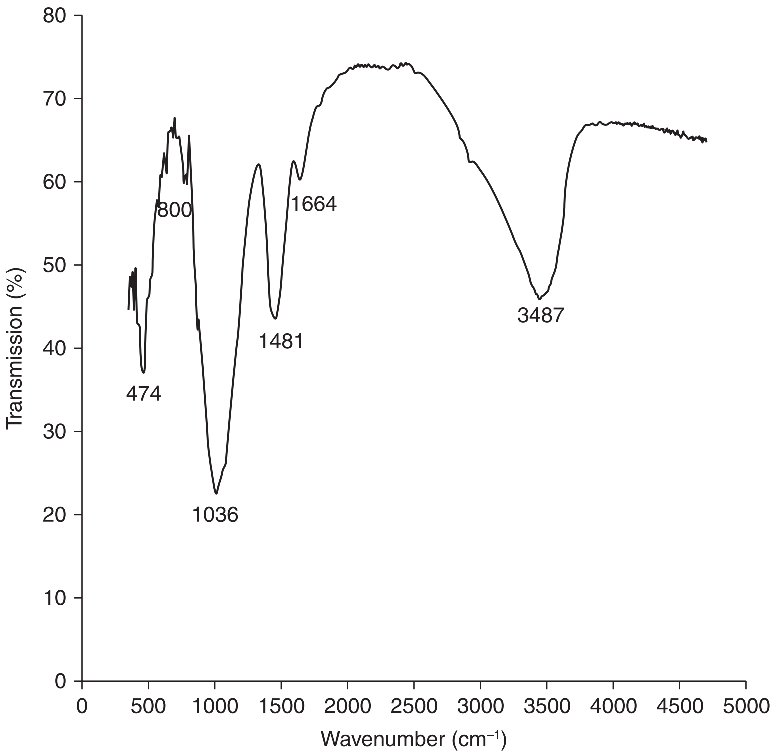
A comparison of the results of 28-day compressive strength and efflorescence reveal that a relatively high compressive strength attainable at higher silica modulus in moderate Na_2O contents could have suitable soundness and durability. However, mixes showing no efflorescence exhibit good 28-day compressive strength. Previous investigations have proved that the appearance of efflorescence is due to leaching of non-reacted sodium hydroxide, which in a secondary reaction reacts with atmospheric carbon dioxide, producing sodium carbonate (Allahverdi *et al.*, 2009; Najafi *et al.*, 2012). In the case of construction waste as a precursor for geopolymer production, an optimum Na_2O content along with hydrothermal treatment of paste specimens could potentially result in the development of a suitable geopolymer cement system.

Efflorescence is sometimes an issue in alkali-activated binders derived from CDW precursors, but according to the authors' experiences (Najafi *et al.*, 2012), it can be effectively reduced either by the addition of alumina-rich admixtures or by hydrothermal curing. Hydrothermal curing at temperatures higher than 45 °C provides a significant reduction in efflorescence, and also slight strength improvements. The addition of high-alumina cements is particularly beneficial; an optimum dosage of calcium aluminate cement, especially Secar 71, as an admixture is more effective than hydrothermal curing in reducing the extent of efflorescence in the construction waste-based geopolymer binders studied. The additional alumina supplied by the high alumina cement admixtures leads to an increased extent of cross-linking in the geopolymer binder, reduces the mobility of alkalis (which is the key cause of efflorescence in these materials) and also generates a hardened geopolymer binder product with markedly improved mechanical properties compared to the systems with no admixtures (Najafi *et al.*, 2012).

18.6.6 Fourier Transform Infrared (FTIR) analysis

The mix exhibiting the highest compressive strength with 60% of waste brick and 40% of waste concrete as dry binder, with Na_2O content of 5% and Ms of 1.0, that is M5-8, was selected and a sample of its 28-day hardened paste was analysed by FTIR spectroscopy. Figure 18.6 displays the corresponding infrared spectrum. As can be seen, the infrared spectrum of the sample is presenting analogous absorption bands. It shows bands at 3440 and 1650 cm^{-1}, respectively, related to O–H stretching and bending modes of molecular water and also near 1000 and 450 cm^{-1} due to asymmetric Si–O–Al stretching vibrations and to in-plane Si–O bending vibrations in SiO_4 tetrahedral, respectively (Clayden *et al.*, 1991; Ortego and Barroeta, 1991; Allahverdi *et al.*, 2011).

The Si–O stretching modes for the SiQ^n units show infrared absorption bands localised around 1100, 1000, 950, 900 and 850 cm^{-1} for n = 4, 3, 2, 1 and 0,



18.6 FTIR spectrum of 28-day hardened paste of mix M5-8.

respectively (Clayden *et al.*, 1991). These values shift to lower wavenumbers when the degree of silicon substitution by aluminium in the second coordination sphere increases, as a consequence of the weaker Al–O bonds. A comparison of the wavenumbers shows that the Si–O stretching band shifts progressively towards higher wavenumbers. For the tested sample, the Si–O stretching band appears at the wavenumber of 1036 cm^{-1}. This indicates a distribution of the polymerised Q^n units centred on Q^3 and Q^4 units. The tested sample contains carbonate species highlighted by the presence of the large absorption band near 1450 cm^{-1}, related to asymmetric stretching and out-of-plane bending modes of CO_3^{2-} ions (Phair and van Deventer, 2002).

The obtained spectrum shows some differences when compared to the spectra of the preliminary waste brick and concrete (Fig. 18.3). The broad peak at around 1000 cm^{-1} in the preliminary materials has been concentrated in the cured hardened paste, confirming that geopolymerisation reactions have resulted in a more orientated molecular structure. The geopolymerisation reactions occurring will probably result in the formation of more aluminosilicate gel, which in turn produces a more orientated molecular structure upon polycondensation.

18.6.7 Scanning electron microscopy (SEM)

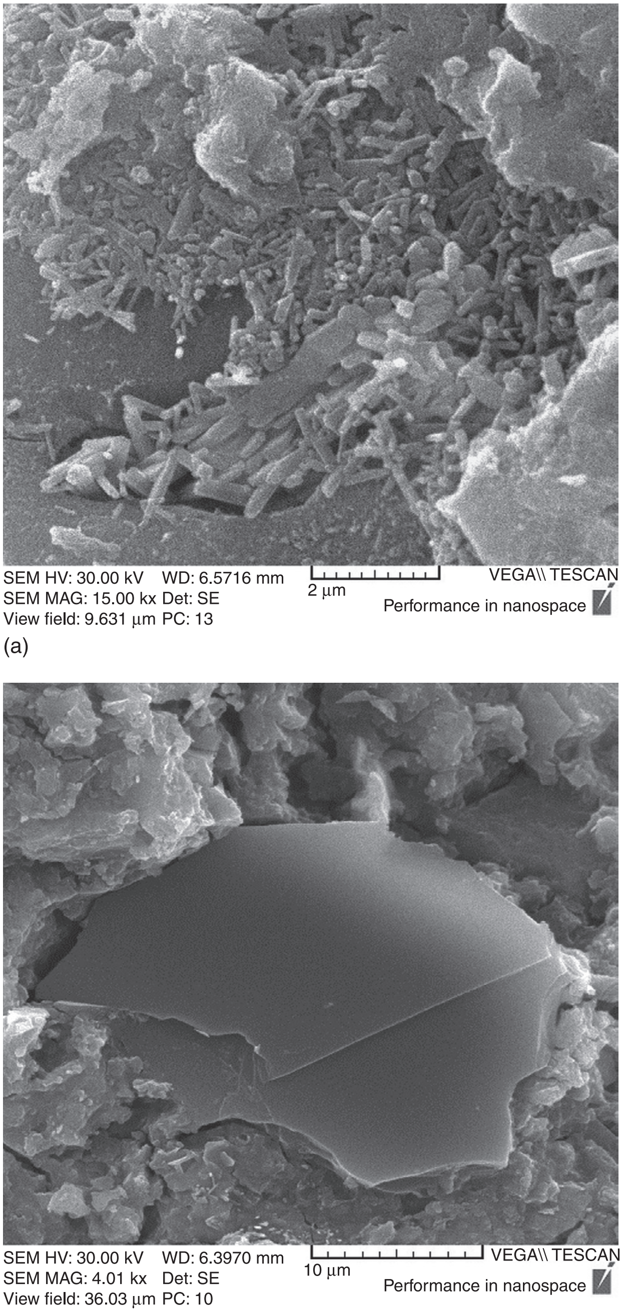
Investigations were carried out by scanning electron microscopy (SEM) on 28-day hardened pastes comprising 60% of waste brick and 40% of waste concrete as dry binder, with Na_2O content of 5% and Ms of 1.0, that is mix M5-8. Figure 18.7 shows the microstructure of the hardened paste at six different magnifications. The observed particles are those of the raw materials which were bound together by dissolution of their surface in alkali-activator and formation of geopolymer compounds. The authors (Najafi *et al.*, 2009a) hypothesise that hydrothermal curing could be useful to dissolve a higher proportion of waste brick and concrete and to increase the extent of geopolymerisation reactions.

In the aforementioned cases, the microstructures with higher magnifications consist of a matrix in which small particles of different sizes and shapes are embedded, and a few fibre-like crystals that are difficult to identify. Alkali-activated materials are known to contain amounts of unreacted solid aluminosilicate source material. This is confirmed by the embedded particles observed in the SEM micrographs. However, there is no definitive and accurate method for quantitatively determining the amount of unreacted material in a particular specimen. For any qualitative or semi-quantitative description of the effect of unreacted material on the strength of geopolymers, a measure of the amount of unreacted material is required.

As seen in the SEM images taken at a lower magnification, the microstructure shows a relative coarsening along with a high number of micro-cracks extending in a non-uniform way. In our experience (Najafi *et al.*, 2009a), the microstructure of the specimen cured hydrothermally could be quite sound with no micro-crack. In addition, hydrothermal curing could effectively improve the microstructure of the specimens by preventing the formation of micro-cracks and hence increase the compressive strength.

18.6.8 X-ray diffraction (XRD) analysis

X-ray diffractometry was used to investigate the 28-day hardened paste of mix M5-8. Figure 18.8 shows the XRD pattern of the tested sample. The obtained pattern shows a mostly amorphous material with few crystalline phases including quartz, albite and diopsite, in which quartz was originally present in the both preliminary raw materials. This indicates that the changes responsible for the differences in compressive strength originate and take place within the amorphous part of the structure. Albite belongs to the group of plagioclase with a chemical formula of $(Na,Ca)(Si,Al)_4O_8$.

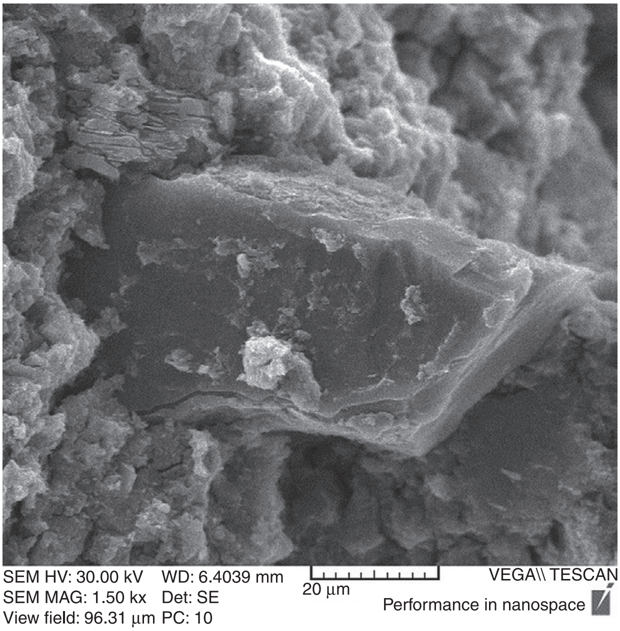


(a)

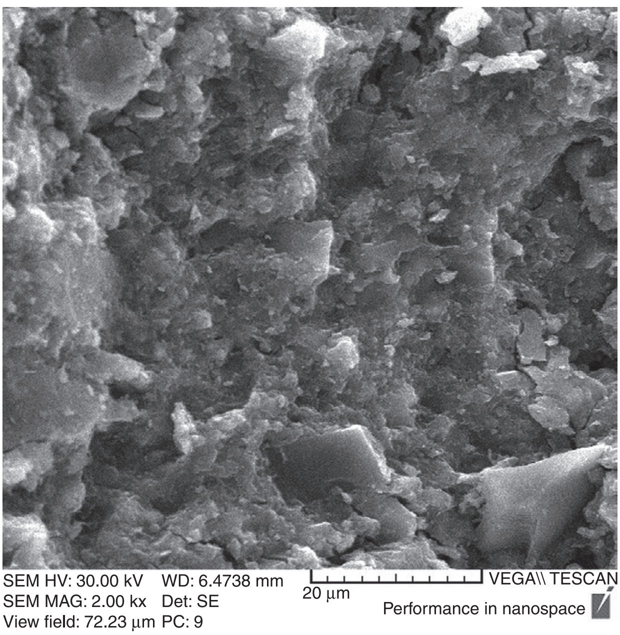
(b)

18.7 (a–h)SEM images from microstructure of 28-day hardened paste of the mix M5-8 at 8 different magnifications.

(*Continued*)

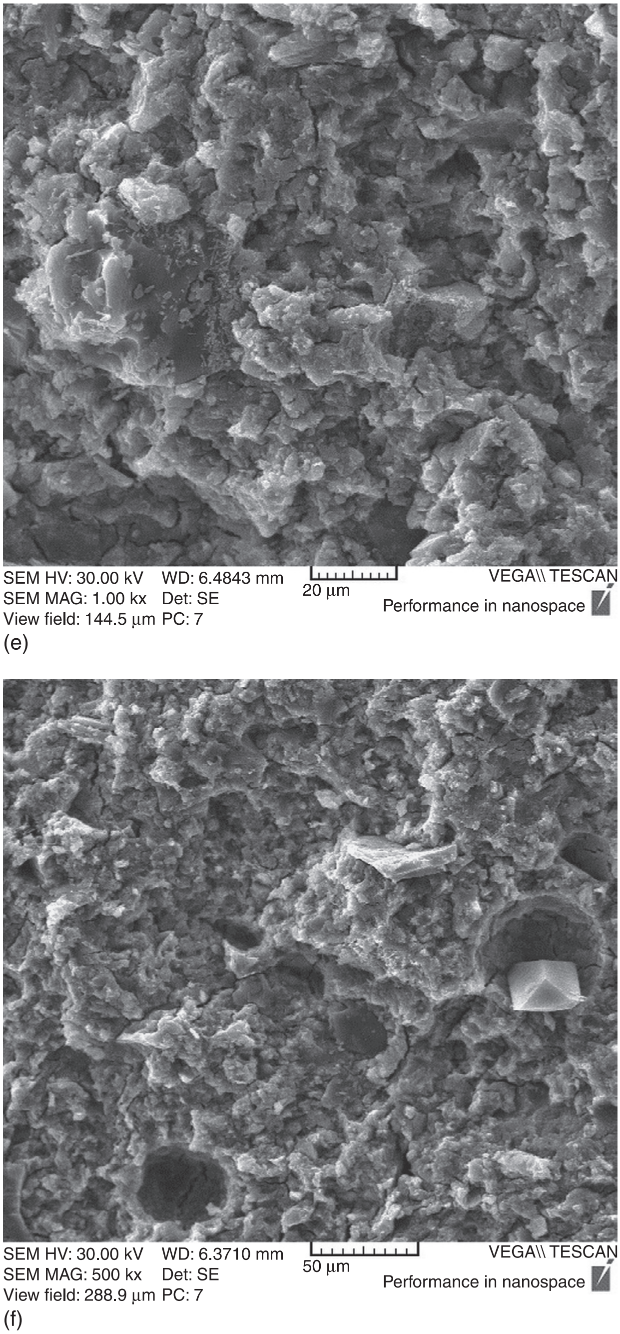


(c)

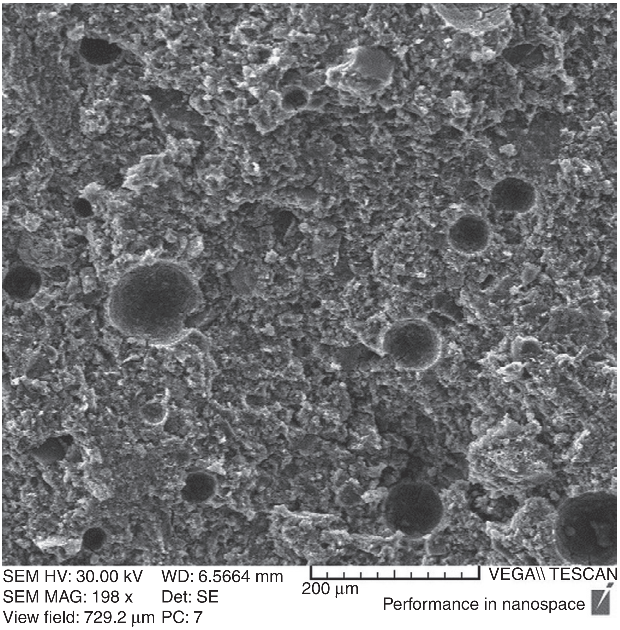


(d)

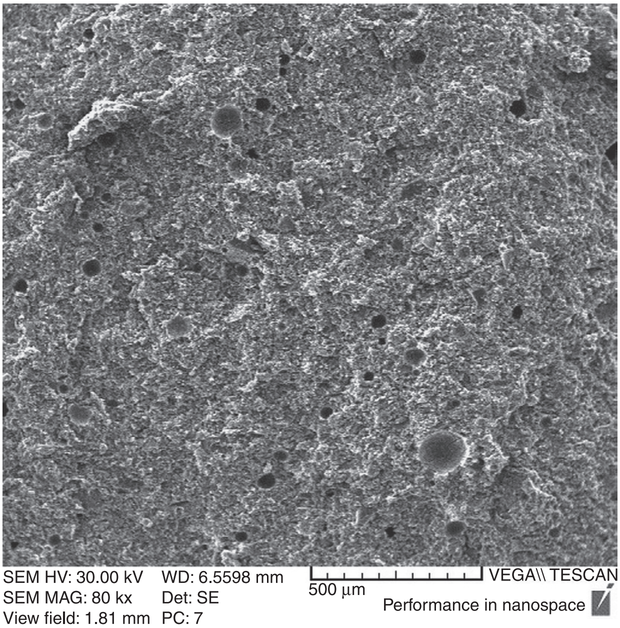
18.7 (Continued).



18.7 (Continued).

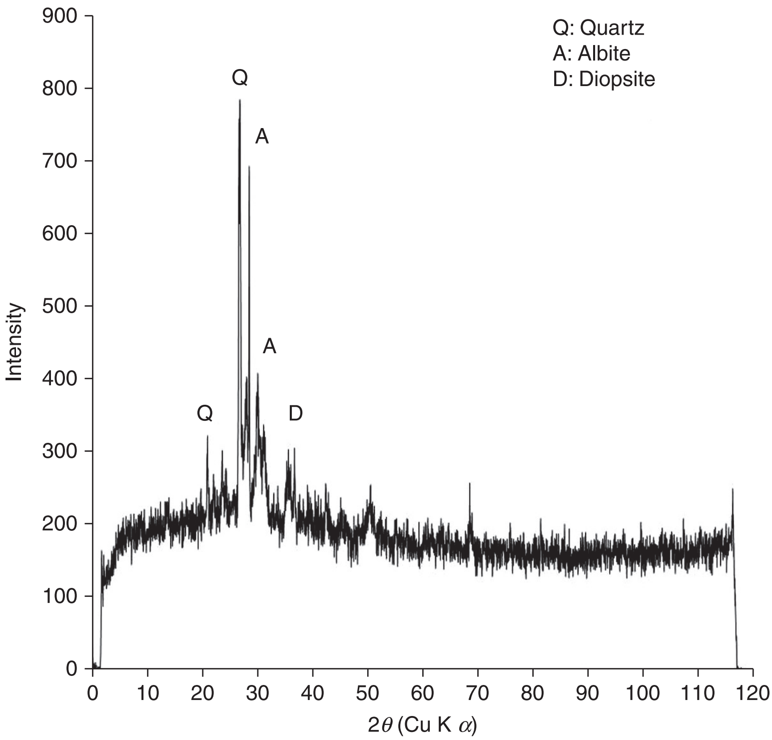


(g)



(h)

18.7 (Continued).



18.8 X-ray diffraction pattern of the 28-day hardened paste of the mixture M5-8.

18.7 Conclusions

It can be summarised that the alkali-activation of CDW (mixtures of waste brick and concrete) with a proportioned alkali-activator can prove the possibility of producing geopolymer binders from these materials as precursors. The mechanical properties of the prepared geopolymer samples based on CDW with different proportions of waste brick and concrete and activators show suitable properties to be acceptable on an industrial scale, but it needs some optimisation in both dry mixture proportions and also the composition of the used activator. Efflorescence is another issue that needs to be considered when CDW is going to be used as geopolymer binder precursors. It must be controlled by optimisation in the chemical composition of the activator and also by utilising the efflorescence control approaches discussed before.

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19
Removing gypsum from construction and demolition waste (C&DW)

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Abstract: In this chapter, gypsum and its use as a construction material is described. Then the pathways for contamination of construction and demolition waste (C&DW) by gypsum during crushing and recovery processes at an intermediate treatment facility and the problems arising from this contamination are disclosed. Finally, the possibility of removing gypsum from C&DW by some current methods and the desired contamination level for various uses of the recovered aggregate are presented.

Key words: construction and demolition waste (C&DW), waste gypsum board, hydrogen sulfide gas, sieving.

19.1 Introduction

The management of construction and demolition waste (C&DW) is an important problem in many countries because of its massive quantity (Poon, 1997; EC, 2000; US EPA, 2001; Huang *et al.*, 2002; Kartam *et al.*, 2004; Nunes *et al.*, 2007). In Japan, landfill waste mainly consists of sludge (39%), stones, bricks and blocks (12%) (Japan ME, 2008). C&DW, which is generated at construction and/or demolition sites and contains the above-mentioned two landfill waste components, is the major landfill waste, accounting for 18% of the total amount of industrial solid waste deposited in landfill sites in 2005 (Japan ML, 2006).

One-third of C&DW deposited in landfill sites is a mixture of different types of waste called 'mixed construction and demolition waste (mixed C&DW)'. The amount of waste processed at an intermediate treatment facility was 46% of the total amount of generated mixed C&DW in 2005 in Japan, that is less than half, whereas the amount of waste deposited in landfill sites was 72% of the total amount of generated mixed C&DW. Clearly, a large amount of untreated, mixed C&DW is deposited in landfill sites without reduction or recovery of resources. From the point of view of resources saving, resources in C&DW should be recovered.

Gypsum board, a common construction material, is found in processed C&DW (Jang and Townsend, 2001; Musson *et al.*, 2008). Waste gypsum board elutes sulfate (Jang and Townsend, 2001; Barbudo *et al.*, 2012) into the landfill layer. Under anaerobic conditions, sulfate is reduced to hydrogen sulfide (H_2S) gas, which is poisonous and emits an obnoxious smell (Lee *et al.*, 2006). This is caused by the decomposition of organic matter, such as wood and paper. However, this phenomenon is not limited to the landfilling of mixed C&DW. The same condition, such as the existence of sulfate and anaerobic zone, is created when recovered aggregate containing gypsum is used as backfill material under ground. Therefore, gypsum board (and organic matter in the form of wood and paper) should be removed from the recovered aggregate by an intermediate treatment step before the recovered aggregate is reused as a construction material.

In this chapter, gypsum and its use as a construction material are described first. Then the pathways for contamination of C&DW by gypsum during the crushing and recovery processes at an intermediate treatment facility and the problems arising from the contamination are disclosed. Finally, the possibility of removing gypsum from C&DW by some current methods and the desired contamination level for various uses of the recovered aggregate are presented.

19.2 Definition and utilization of gypsum

Gypsum is found in nature in large quantities. Commercial quantities of gypsum are found in Brazil, Pakistan, Jamaica, Iran, Thailand, Spain, Germany, Italy, England, Ireland, Canada and the United States. In addition, synthetic gypsum is recovered via flue-gas desulfurization in some coal-fired electric power plants and used interchangeably with natural gypsum in some applications.

19.2.1 Definition of gypsum

Gypsum is a sulfate mineral composed of calcium sulfate dihydrate and has the chemical formula $CaSO_4{\cdot}2H_2O$. It exists as massive materials (the alabaster variety), clear crystals (the selenite variety) and parallel fibers (the satin spar variety). When the crystal lattice is heated, water from crystallization is lost and the crystal becomes bassanite ($CaSO_4{\cdot}0.5H_2O$) at approximately 373 K and anhydrite ($CaSO_4$) at approximately 433 K (Troev *et al.*, 1994).

19.2.2 Utilization of gypsum as construction material

Gypsum has a wide variety of applications. Gypsum board is primarily used as a finish for walls and ceilings, and is known in construction as drywall or plasterboard. Gypsum is also an ingredient of surgical splints, casting molds and modeling, utilizing the property of coagulation. Furthermore, gypsum is used for soil improvement in the form of fertilizer and soil conditioner. Gypsum is often used as part of a strategy to correct compacted soil or soil with large amounts of clay. It is also used to counteract excessive saline levels in soil and has the added benefit of not affecting the pH of soil.

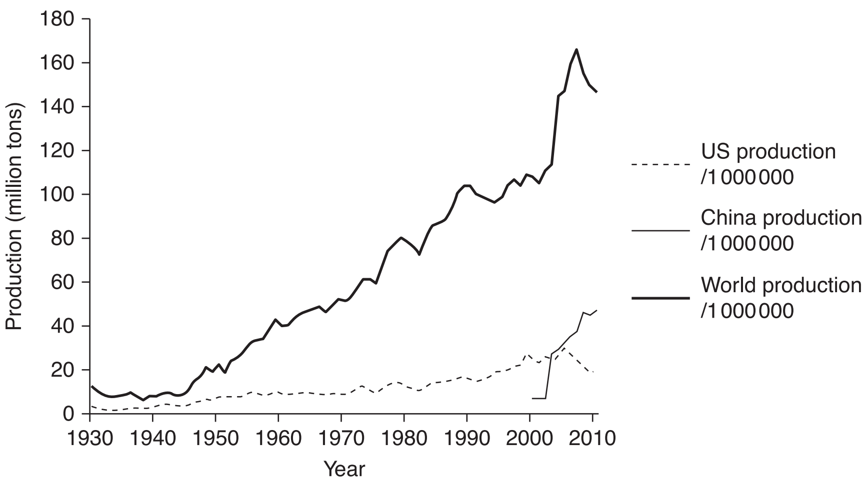
Gypsum board, also known as drywall, plasterboard or wallboard, is used to form panels made of gypsum plaster pressed between two thick sheets of paper.

Gypsum board is used as partitions and linings of walls, ceilings, roofs and floors. It possesses many attributes that make it an attractive construction material. Its important properties (Euro gypsum) are described as follows.

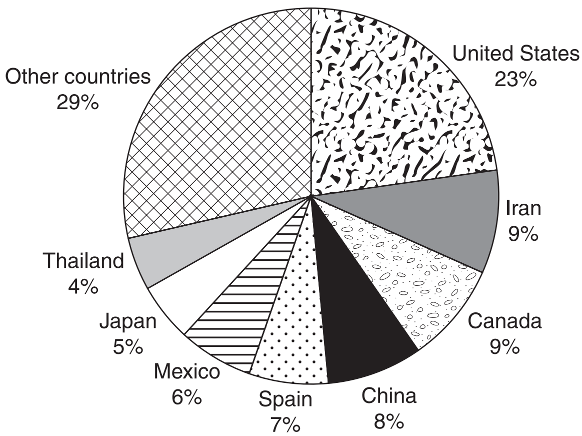
Gypsum board is used as interior wall surface because of ease of installation. It is porous, soft and lightweight, and can be cut by a utility knife and nailed to a structure. Gypsum provides fire protection. Gypsum *per se*, made of $CaSO_4$ and crystal water, is a non-combustible substance. Furthermore, crystal water is also fire-resistant. When gypsum board is exposed to heat or fire, the water is vaporized, retarding heat transfer. Gypsum also equilibrates humidity and heat peaks. Gypsum board is capable of storing humidity when a room is humid and automatically releasing humidity if the indoor air is too dry. It acts as a thermal insulator because of its low thermal conductivity. Furthermore, it also has 'heat-storing' abilities. Small temperature increases are absorbed and radiated back later when the temperature in the room decreases.

19.2.3 Past and future usage of gypsum

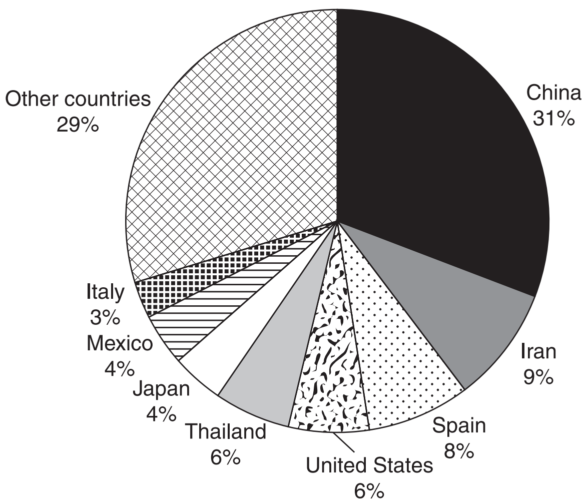
The time course of global gypsum production is shown in Fig. 19.1. Production was maintained at around 10 million tons from 1930 to 1945, followed by a linear increase from 1945 to the early 2000s, finally reaching 108 million tons in 2000. Production increased drastically after 2004. Gypsum production by country in 2000 and 2010 is shown in Figs 19.2 and 19.3, respectively. In 2000, the US accounted for 23% of the world's gypsum production, Iran and Canada, each 9%, and China, 8%. In 2010, China accounted for 31%, Iran, 9%, Spain, 8%, and the US, 6%. As shown in Fig. 19.1, gypsum production in China has shown a dramatic increase in recent years.



19.1 Gypsum production (US Geological Survey, 2011a).



19.2 Gypsum production by country in 2000 (US Geological Survey, 2001).



19.3 Gypsum production by country in 2010 (US Geological Survey, 2011b).

The gypsum market is expected to sustain its growth rate due to the construction boom in developing countries. China remains the key demand region for gypsum and today the growth of the construction materials market, the development of infrastructure in China and other developing economies, and heavy investment and support by governments are driving the gypsum market (Merchant Research & Consulting Ltd, 2011).

19.3 The problem of contamination of construction and demolition waste (C&DW) by gypsum

C&DW generated from a building in which gypsum board was used may be contaminated with gypsum. In Japan, *The Construction Material Recycling Law* partially requires contractors to sort out and recycle waste generated in building demolition work. C&DW generated in demolition work without on-site sorting may be contaminated with gypsum board. In addition, contamination also occurs due to the inability to identify or distinguish gypsum, incomplete removal and/or marked size reduction of gypsum board in the demolition process.

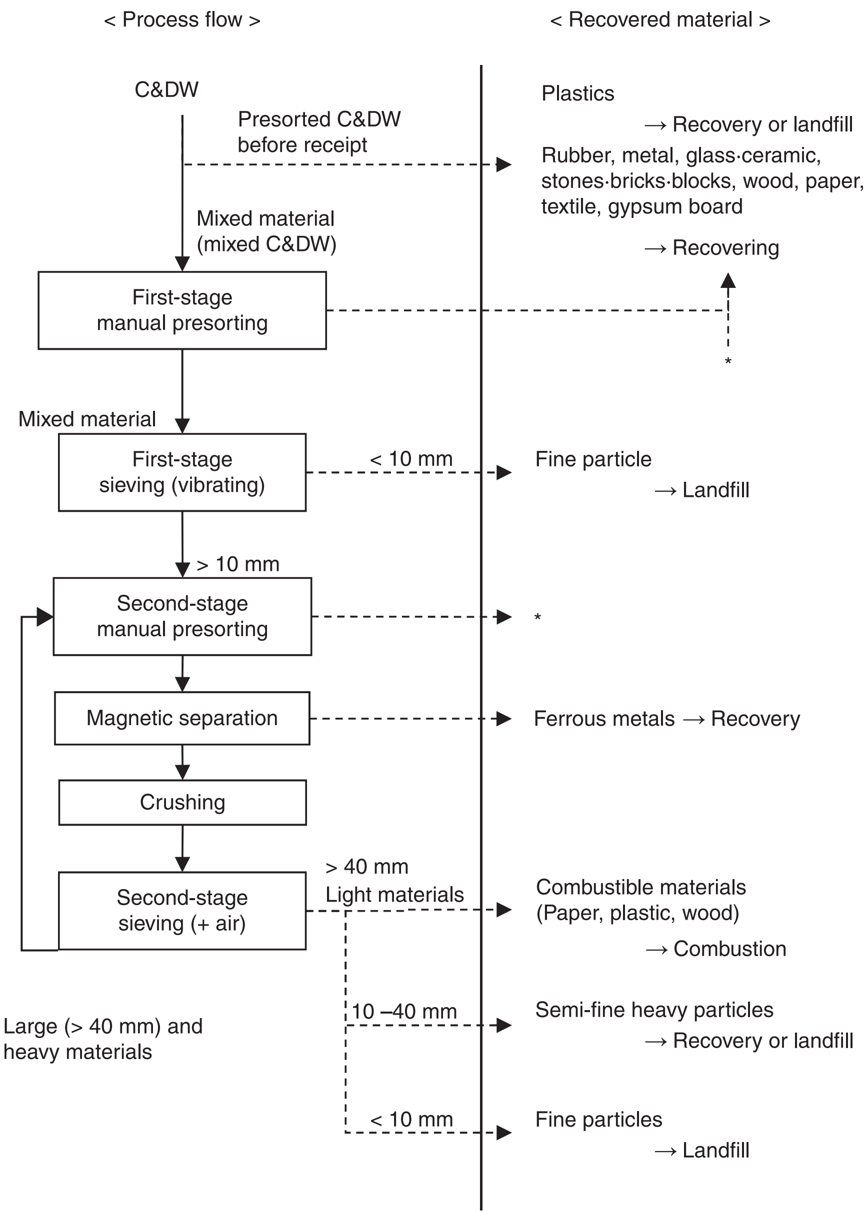
19.3.1 Contamination pathways of gypsum

Figure 19.4 shows a typical treatment flow of an intermediate treatment facility for C&DW. Material separation is performed by manual presorting (Fig. 19.5), sieving, magnetic separation and density classification (air separation). Materials that are generated in the demolition work by on-site sorting or distinguishable coarse particles are separated manually. After that, composite and/or undistinguishable materials are crushed and subsequently the mixed material is separated into combustibles, incombustibles and metals by manual presorting, sieving and magnetic separation. Gypsum board (Fig. 19.6) tends to become fine particles by crushing because it is fragile. Because of this, gypsum can find its way into semi-fine (Fig. 19.7) and fine (Fig. 19.8) particles generated by on-site demolition work and/or the crushing process at an intermediate treatment facility.

19.3.2 Gypsum content in C&DW

Mixed C&DW contains heterogeneous matter, such as inorganic matter, including concrete, gypsum board, stones, bricks, blocks, glass, ceramic and metals, and organic matter, including wood, paper and plastic (Cochran *et al.* 2007). In Japan, although residential and non-residential structures are mostly made of wood, the actual wood content in C&DW was 6 wt% in 2002, which was lower than that in the US (10 wt%; Table 19.1). Gypsum board content in Japan (5.6 wt%) was also lower than that in the US (12 wt%; US EPA, 1998; Huang *et al.*, 2002; Kartam *et al.*, 2004; Japan ML, 2006; Mamiya *et al.*, 2006; Cochran *et al.*, 2007; Japan GA, 2008). However, the composition of C&DW is not markedly different.

The contents of gypsum in processed C&DW are summarized in Table 19.2. The median values for gypsum were 13.4% for C&DWf (S) (generated at construction and demolition work) and 1.1% for C&DWf (M) (generated at construction work). Although waste gypsum board could be separated at construction work, gypsum board at demolition work might be easily crushed, resulting in the higher than 10% content in the processed C&DW. Gypsum content in concrete and mixed rubble was 0.98 to 1.20% in Spain (Vegas *et al.*, 2008) and 8.6 to 22% in the US (Musson

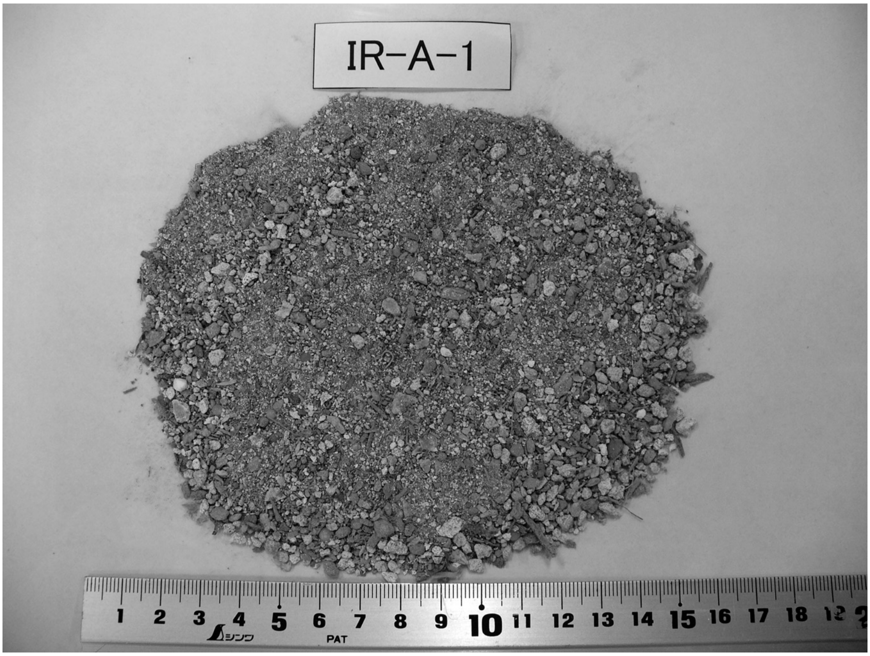


19.4 Typical treatment flow of an intermediate treatment facility where mixed C&DW is crushed and recyclable resources are recovered.

19.5 Manual presorting.

19.6 Waste gypsum board.

19.7 Semi-fine particles.



19.8 Fine particles.

Table 19.1 Composition of C&DW (weight ratio)

	Japan in 2002[a]							USA						
	Residential (±1%)			Nonresidential (±1%)			Total	Residential in 2000[b]			Nonresidential in 2000[b]			Total in 1996–2000[bc]
	Const.	Ren.	Dem.	Const	Ren.	Dem.		Const	Ren.	Dem.	Const	Ren.	Dem.	
Plastic	14	15	9	13	17	14	–	–	–	–	–	–	–	–
Paper	5	3	2	5	3	2	–	3	1	0	1	–	–	0
Wood	5	19	19	13	9	12	6[d]	21	22	7	10	13	0.2	10
Rock, concrete, brick, block, ceramic, and asphalt	49	43	28	49	38	30	–	56	54	79	64	36	82	64
Gypsum and drywall	6[f]	6[f]	6[f]	6[f]	14[f]	27[f]	5.6[ef]	14[g]	9[g]	5[g]	11[g]	38[g]	–	12[g]
Metal	5	10	2	6	15	5	–	2	1	0.2	3	3	5	3
Miscellaneous, Fine	16	4	34	8	4	10	–	4	13	9	11	10	13	11

Notes:
Const., construction; Ren., renovation; Dem., demolition.

[a] Mamiya *et al.*, 2006, [b] Cochran *et al.*, 2007, [c] US EPA, 1998, [d] Japan ML, 2006, [e] Japan GA, 2008 [f] Gypsum, [g] drywall (sheetrock, gypsum, plaster).

Unit: wt%.

Table 19.2 Properties of C&D-Wf (S&M)

			Min.	Median value	Ave.	Max.	Standard deviation
C&D-Wf (S) (n = 11)	Particle density	g cm^{-3}	2.2	2.4	2.4	2.5	0.1
	Moisture content	wt%	9.1	14.3	15.0	20.7	3.8
	LOI	%	9.0	15.3	14.5	21.4	3.8
	Gypsum	%	1.1	13.4	11.5	22.2	6.6
	Wood	%	0.2	2.1	2.5	9.3	2.5
C&D-Wf (M) (n = 66)	Moisture content	wt%	0.8	18.8	19.3	63.9	12.1
	LOI	%	1.3	9.7	14.5	73.2	15.2
	Gypsum	%	0.0	1.1	4.7	38.0	8.0
	Wood	%	0.0	2.0	6.9	72.7	15.9

et al., 2008). The measured values had a wide range and higher than 20% gypsum content was also reported.

19.3.3 Environmental and technical problems arising from contamination

Gypsum-containing materials can generate toxic H_2S gas when used as underground construction material or deposited in landfill sites. The mechanism by which gypsum generates H_2S gas is described as follows. First, sulfate in gypsum is dissolved on contact with rainwater or groundwater. Second, gas and/or dissolved oxygen are consumed when there is a large amount of biodegradable organic matter. In addition, the condition of the layer under the ground changes to anaerobic when the layer is submerged in water. Finally, sulfate-reducing bacteria multiply and convert sulfate into H_2S gas in the presence of sulfate and organic matter under appropriate anaerobic conditions for bacterial growth. H_2S gas is harmful to living bodies and generates black water by reacting with dissolved iron. The gas can kill humans immediately at 1000 ppmv (Ono and Tanaka, 2003). In Japan, H_2S gas generated from a landfill site killed the operators (Ono and Tanaka, 2003).

Gypsum board is fragile and tends to produce fine particles by crushing. In addition, fine gypsum particles become pasty through the treatment process for wet C&DW or when exposed to rain. Normal treatment is hindered because any drive parts that come in contact with waste, such as a sieve, a conveyor or a crusher, tends to become entangled in the paste. The paste on the sieve hinders passage of waste particles.

19.4 Current methods of removing gypsum from C&DW

The major method for treating mixed C&DW at intermediate treatment facilities is crushing (Brunner and Stämpfli, 1993; Townsend *et al.*, 1998; Ono, 2005).

Manual, magnetic, air and sieving separation processes are usually performed before and/or after the crushing process, to accomplish smooth size reduction and recover resources.

19.4.1 Current method of removing gypsum from C&DW in demolition site and intermediate treatment facilities

With regard to recovering materials from C&DW, gypsum board should be removed as impurities, except when the gypsum itself is to be reused. Gypsum is an impurity in reusable combustibles (paper, plastic and wood) or incombustibles (metals, glass, ceramic, stones, bricks and blocks). In addition, gypsum may generate toxic H_2S gas when incombustibles contaminated with gypsum are used as backfill materials. This phenomenon should also occur when C&DW contaminated with gypsum is deposited in landfill sites. Clearly, contaminated gypsum in C&DW should be removed. However, the separation of gypsum is also important when aiming for the reuse of gypsum itself.

In general, separation of the objective material before mixing with other materials is easier than separation from a mixture. In Japan, *The Construction Material Recycling Law* partially requires contractors to sort out and recycle waste generated in building demolition work. In addition, as the disposal expense in an intermediate treatment facility is higher for a mixture than for sorted C&DW, demolition work with on-site sorting is recommended. Distinguishing and removing gypsum board from a building demolition site is the simplest way to remove gypsum from C&DW. Gypsum particles have to be distinguishable for them to be removed by manual presorting. Gypsum board from demolition work with on-site sorting or distinguishable coarse particles can be separated. Non-gypsum materials generated by on-site sorting or recovered by manual presorting can be considered gypsum-free material.

C&DW without on-site sorting may be contaminated with gypsum particles. There is no process for removing gypsum board in an intermediate treatment facility for C&DW, except manual presorting. In addition, the removal of fine gypsum particles is almost impossible, as the fine particles are visually indistinguishable. The current process carried out in an intermediate treatment facility is not suitable for contaminated gypsum particles, namely, it is suitable for gypsum board alone, for example, sorted and collected waste gypsum board is crushed and separated into gypsum, paper and other components. Therefore, manual presorting is currently the only method of removing gypsum from C&DW in an intermediate treatment facility (Fig. 19.4).

19.4.2 Gypsum content in C&DW after treatment

As described in Section 19.4.1, there is no process for removing gypsum board in an intermediate treatment facility for C&DW except manual presorting. Therefore, fine gypsum particles cannot be removed from C&DW at present. However, is it possible to reduce gypsum content in processed C&DW by means of current separation techniques, such as manual presorting, sieving or others? The author examined a method to obtain processed C&DW having low gypsum content. The removal of organic matter, such as wood, was also investigated.

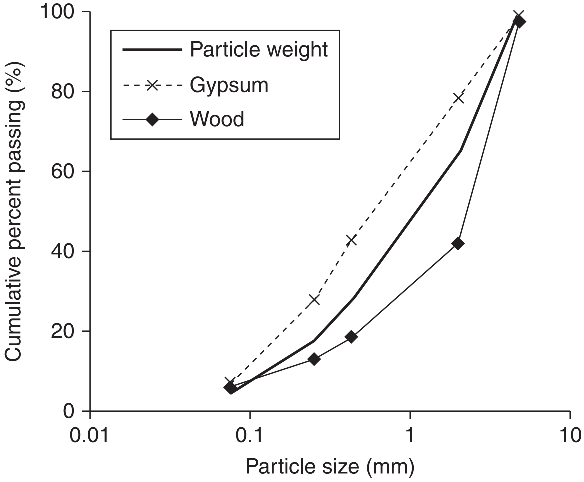
Processed C&DW was sampled from seven intermediate treatment facilities in Japan (A–G), where mixed C&DW was crushed and recyclable matter recovered. At every facility, mixed C&DW was processed by manual separation, sieving, crushing, magnetic separation and air classification. At facilities A–F, processed C&DW for deposition in a landfill was sampled. The source was construction and demolition work (mixture), and the total number of samples was 11. At facility G, processed C&DW after the first stage of manual presorting was collected and sieved through a 5.6 mm mesh laboratory sieve. At the first stage of manual presorting, the presence of gypsum board in mixed C&DW, which was confirmed by visual observation, was checked. The total number of samples was 66 and all waste sources were from the construction work of a new building. Hereafter, samples from facilities A–F are called 'C&DWf (S)' (f, fines; S, sieved), and samples from facility G are called 'C&DWf (M)' (M, manually separated).

Removability by additional sieving

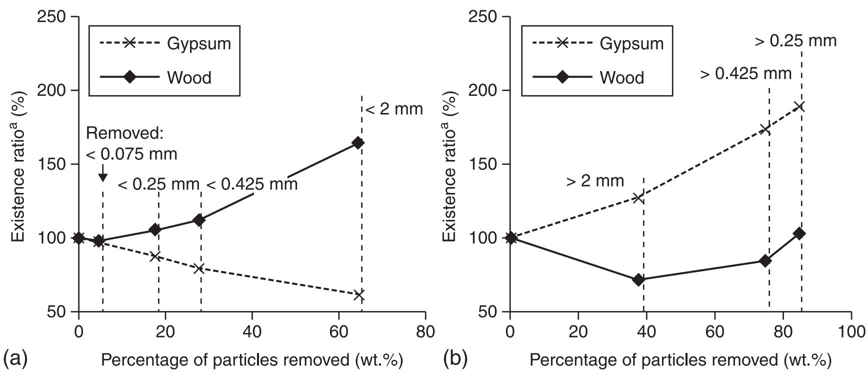
Mixed C&DW was separated with a sieve, whose mesh size was adjusted according to weather conditions (moisture content of mixed C&DW) at the intermediate treatment facility. Organic matter, such as wood, paper and plastic, remained in the coarse fraction after crushing and fine particles (processed C&DW in this study) were generated by crushing brittle inorganic matter. Then the removability of gypsum and wood by additional sieving of processed C&DW was investigated.

Particle weight, gypsum content and wood content in the sieved C&DWf (S) fractions of 0.075, 0.250, 0.425 and 2 mm particle size were measured. The ratio of each content to the total content in the initial sample, that is before sieving C&DWf (S), was calculated and particle size distribution of each component was obtained (Fig. 19.9). Gypsum tended to be present in fine particles and wood tended to be present in coarse particles. However, no obvious deviation of particle size was noted.

The contents of gypsum and wood in the fraction newly generated from C&DWf (S) by sieving at a certain mesh size to remove fine (Fig. 19.10a) or coarse particles (Fig. 19.10b) were calculated (contents in recovered fraction/contents in initial sample (%), using particle size distribution (Fig. 19.9)) and assuming 100% content in the initial sample. Gypsum content decreased by removing fine particles, whereas wood content decreased by removing coarse particles. The minimum contents were 61% (gypsum) and 72% (wood), compared with those in the initial sample. The contents could not be reduced to half in any case.



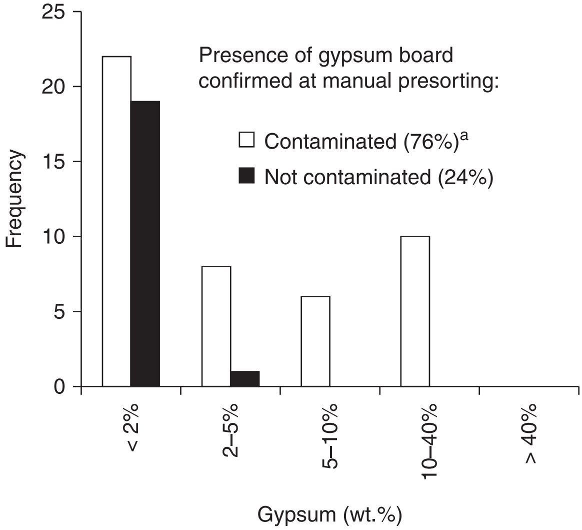
19.9 Particle size distribution of substances in C&DWf (S) (Median value, n = 11).



19.10 Change of existence ratio of substances in C&DWf (S) when (a) fine particles and (b) coarse particles were removed (Median value, n = 11) [a]Contents in recovered fraction/contents in initial sample [%].

Separation by manual presorting according to the presence of gypsum board

Simplified methods to estimate gypsum contents in mixed C&DW or processed C&DW were investigated. The relationships between the presence of gypsum board in mixed C&DW at the first stage of manual presorting and the content of gypsum in C&DWf (M) generated by removing gypsum board by manual presorting and sieving are summarized in Fig. 19.11. The percentage of samples having more than 2% gypsum content was higher in the samples that were contaminated with gypsum board than in those not contaminated with it. Therefore,



19.11 Gypsum contents in C&DWf (M) separated by manual presorting according to the presence of gypsum board (n = 66) [a]Weight ratio [%].

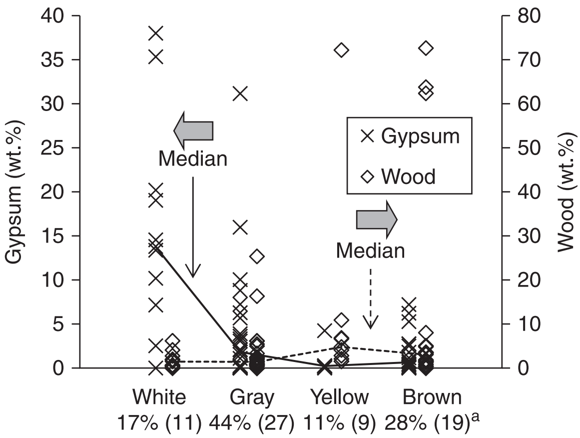
gypsum content in processed C&DW could be estimated from the presence of gypsum board in mixed C&DW.

Separation by color

However, the method for content determination mentioned above was insufficient at the intermediate treatment facility, because it was not known whether waste gypsum board was generated or not at construction and demolition work, and fine gypsum particles generated by crushing mixed C&DW could not be discriminated. Wood was also crushed into fine particles and therefore contaminated mixed C&DW. However, as the color of processed C&DW varied from white to brown, the presence of gypsum board and wood was possibly responsible for the color. Therefore, the contents of gypsum and/or wood in processed C&DW could be determined according to color. The relationship between color of C&DWf (M) and contents of gypsum and/or wood is summarized in Fig. 19.12. The colors were white, gray, yellow and brown. Gypsum content was higher in the white samples than in the other samples. Gypsum contents in the white and gray samples varied widely. Wood contents were high in the yellow and brown samples.

Separation efficiency

From the above, processed C&DW having low gypsum and wood contents could be determined by evaluating the presence of waste gypsum board contaminating



19.12 Gypsum and wood contents in C&DWf (M) separated subjectively by color (n = 66) [a]Weight ratio [%] (number).

mixed C&DW at the first stage of manual presorting and the color of processed C&DW. Using the above results, the weight ratio of recovered C&DWf (M) and the contents of gypsum and wood are summarized in Fig. 19.13. Contents were calculated using the equation:

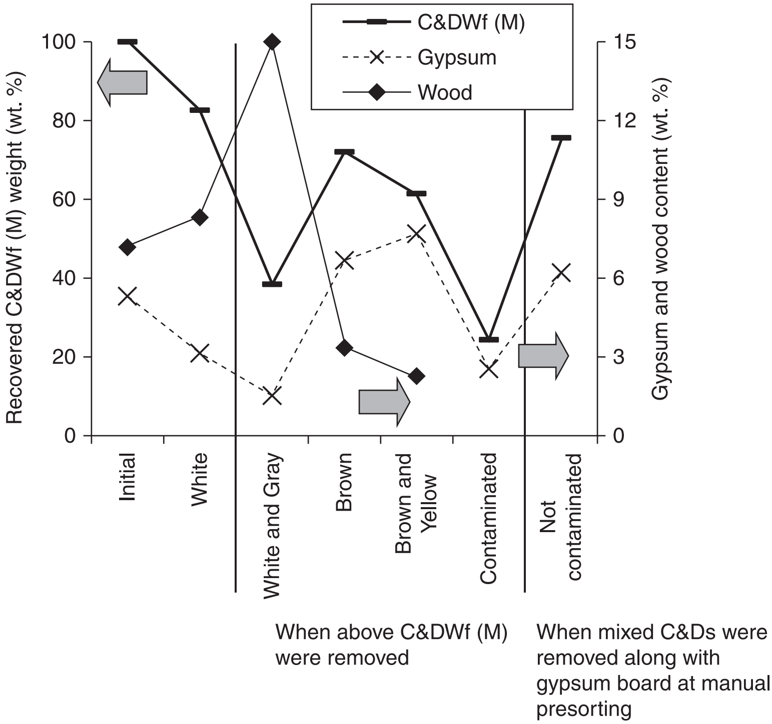
$$(\text{Weight of each sample} \times \text{Content in each sample})/(\text{Total weight of samples}) \quad [19.1]$$

Therefore, the averages and median values listed in Table 19.2 were different from the initial contents shown in Fig. 19.13. To obtain processed C&DW, whose contents of gypsum and wood were less than half of that in the initial state, C&DWf (M) from mixed C&DW in which the presence of gypsum board was observed at the first stage of manual presorting or white and gray processed C&DW should be removed to eliminate gypsum, and brown or brown and yellow processed C&DW should be removed to eliminate wood, without mixing with processed C&DW containing other colors at stockyards.

However, as the recovery weight ratios of the fractions varied with each of the treatments mentioned above, separation efficiency could not be evaluated. Then the overall separation efficiency (OSE) to obtain the fraction (objective component) that was free from gypsum or wood (non-objective component) was calculated with the following equation:

$$\text{OSE} = \text{Recovery ratio of objective component} - \text{Recovery ratio of non-objective component}$$

$$= \{W_r\,(1-0.01C_r)\}/\{W_{ini}\,(1-0.01C_{ini})\} - (W_r\,C_r)/(W_{ini}\,C_{ini}) \quad [19.2]$$



19.13 Gypsum and wood contents in C&DWf (M) after sorting by color and presence of gypsum board (n=66).

where W is fraction weight (kg), C is content of non-objective component (%), and subscripts '*r*' and '*ini*' indicate recovered fraction and initial state, respectively. The OSE and weight of recovered fraction are summarized in Table 19.3. The OSE for additional sieving is also summarized. OSE of gypsum from white C&DWf (M) was maximum at 0.36 (content, 59% of initial sample) and recovery weight of processed C&DW was 83 wt%. When white and gray fractions were removed, the gypsum contents were lowered further (Fig. 19.13). However, the OSE was low (0.29) because of the low recovery weight of the fraction (38 wt%). The OSE was low (0.13) for mixed C&DW contaminated with gypsum board for the same reason, that is low recovery weight. With regard to wood, OSE of brown and yellow C&DWf (M) was maximum at 0.45 (content, 32% of initial sample). The OSE was 0.42 (recovery weight of the fraction, 72 wt%), even if only the brown fraction was removed (data not shown). OSE by sieving was lower than that by color.

From the above, to obtain processed C&DW having low contents of gypsum and wood, white processed C&DW should be removed to eliminate gypsum, and brown or brown and yellow processed C&DW should be removed to eliminate wood, without mixing with processed C&DW containing other colors at stockyards.

Table 19.3 Overall separation efficiency to obtain processed C&DW free from gypsum

	Removal	Recovery weight of particle	Overall separation efficiency free from gypsum
		wt%	–
Color	White	83	0.36
	White and gray	38	0.29
	Brown	72	−0.19
	Brown and yellow	62	−0.29
Presence of gypsum board	Contaminated	24	0.13
	Not contaminated	76	−0.13
Fine particle	<0.075	95	0.04
	<0.25	83	0.15
	<0.425	73	0.20
	<2	37	0.18
Coarse particle	>0.075	5	−0.04
	>0.25	17	−0.15
	>0.425	27	−0.20
	>2	63	−0.18

19.5 Minimum contamination levels for various uses of recovered aggregate

In the case of cement production, powdered gypsum is added at the final process.

19.5.1 Cement and concrete

Guo and Shi (2008) reported that the content of SO_3 was 2.53 wt% in Portland cement. If the content of SO_3 in cement did not increase markedly, recovered aggregates containing gypsum could be used as raw material for cement or aggregate for concrete. The production of Portland cement using waste gypsum was reported by Chandara *et al.* (2009).

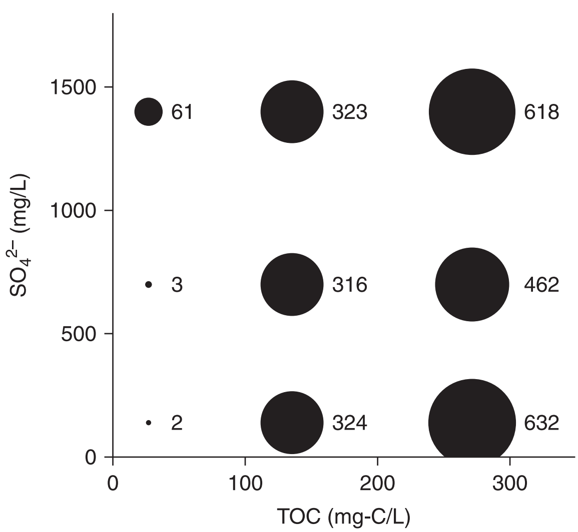
19.5.2 Backfill material

Considering the usage of recovered aggregates from C&DW as backfill material, which is porous, the material can generate H_2S gas when it is submerged. As the solubility of calcium sulfate dihydrate is 0.24 g/100 cm^3 (~1300 mg- SO_4^{2-}/L), the concentration of sulfate in pore water should not exceed saturation concentration, even if a large amount of gypsum exists in the backfill material. For instance, considering that an inert porous material (porosity: 0.5, bulk density: 1 t/m^3) is filled with water, the minimum amount of gypsum required for saturation is 1.2 kg-(calcium sulfate dihydrate)/m^3-porous material, that is only 0.12 wt%-gypsum.

Naturally, such a small amount of gypsum can be flushed out with rainwater or groundwater.

In fact, what should be the desirable or allowable content of contaminated gypsum in backfill material? Should it be below the saturation concentration? It should be noted that organic matter is also required by sulfate-reducing bacteria in order to generate H_2S gas from sulfate. Then, the author investigated the relationship between concentrations of generated H_2S gas and sulfate and organic matter. Solutions having various concentrations of sulfate and organic matter were added to a 50 mL glass bottle and the bottle was sealed (solution volume: 27 mL). The solutions were as follows: fish extract (Wako Pure Chemical) as organic matter; powdered gypsum as sulfate; and nutrients (HACH). Air in the headspace of the glass bottle was purged with N_2 gas. Bottles containing the solutions were incubated at 35 °C for 5 days. Then, the concentrations of H_2S gas in the headspace were measured (Fig. 19.14). More than 300 ppmv H_2S gas was generated when total organic carbon (TOC) was above 140 mg-C/L, regardless of sulfate concentration, such as low (130 mg-SO_4^{2-}/L) or high (1300 mg-SO_4^{2-}/L). However, the concentration of H_2S gas was below 60 ppmv when TOC was 30 mg-C/L.

Therefore, it is difficult to determine the standard gypsum content to prevent the generation of H_2S gas, as the required amount of gypsum is small for saturation and even a low concentration of sulfate can generate H_2S gas. However, the concentration of generated H_2S gas is low when the amount of organic matter is also small, regardless of the existence of sulfate. This phenomenon was also



19.14 Concentration of H_2S gas generated from solutions having various concentrations of TOC and SO_4^{2-} (5 days, 35 °C).

described by Ono and Tanaka (2003) and Miyawaki *et al.* (1996). Naruoka and Ono (2004) and Ono (2010) proposed mixing iron with gypsum as a trap for the generated H_2S.

From the above, the author suggests that the following points be kept in mind when using recovered aggregates from C&DW as backfill material. The content of calcium sulfate dihydrate should be below several wt% and the concentration of organic matter in pore water should be below several tens mg-C/L (as TOC). Iron in an amount equivalent to that of H_2S gas expected to be generated should be mixed. Submerging backfilling material in water and inflow of organic matter to backfilling material should be prevented on site.

19.6 Current research and future needs

Gypsum content in C&DW was extensively investigated by the laboratories of Townsend (Jang and Townsend, 2001; Lee *et al.*, 2006; Musson *et al.*, 2008), Ono (Ono and Tanaka, 2003; Ono, 2005), Tojo (Montero *et al.*, 2008, 2010) and the author (Asakura *et al.*, 2010). The removability of gypsum from C&DW was reported by Tojo's group (Montero *et al.*, 2010) and the author's group (Asakura *et al.*, 2010). Both allowable and desired contamination levels were investigated by the laboratories of Ono (Ono and Tanaka, 2003) and the author. Techniques for preventing H_2S gas generation from waste gypsum board were suggested by Ono's group (Naruoka and Ono, 2004; Ono, 2010) and Yanase's group (Masamoto *et al.*, 2012). In order to utilize recycling aggregates recovered from C&DW containing waste gypsum board, further investigation of removal techniques and the allowable and desirable contamination levels is required.

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Recycling asbestos-containing material (ACM) from construction and demolition waste (CDW)

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Abstract: Asbestos containing materials (ACMs) are a class of hazardous waste whose management has become a matter of great concern. A major open issue is whether ACM wastes, produced after reclamation, should be destined to landfill burial or recycling. After a short history of asbestos as building material, classification, properties and health effects of asbestos minerals, this chapter will describe the methods for the reclamation and disposal of ACMs and attractive recycling solutions as a safe alternative to landfill disposal. Amongst them, recycling of thermally treated cement asbestos for the production of concrete and geopolymers will be illustrated in detail.

Key words: asbestos, asbestos containing waste, landfill disposal, recycling, concrete, geopolymer.

20.1 Introduction

The term *asbestos* derives from the Greek word *ἄσβεστος* or *asbestinon*, which means *unquenchable*/*inextinguishable* and refers to a mineral known and used for millennia (Skinner *et al.*, 1988). The commercial term of *asbestos* applies to a family of six silicate minerals (five amphibole species and one serpentine phyllosilicate) notable for their fibrous (micro)structure and useful attributes (Alleman and Mossman, 1997). There is evidence that chrysotile asbestos was discovered and utilized for the first time in Cyprus, perhaps as long as 5000 years ago (Dilek and Newcomb, 2003). Over the years, asbestos attracted the attention of kings, alchemists and magicians from Western Europe to China, including Georg Agricola (1490–1555), the founder of mineralogy as a scientific discipline (Alleman and Mossman, 1997).

The modern asbestos industrial age began in the second half of the 19th century, with the opening of large-scale asbestos industries in Scotland, Germany and England for the manufacture of asbestos containing (AC) products. Chrysotile asbestos mining in Quebec (Canada) dates back to 1878, and by 1885 seven mines were active. Asbestos rapidly became an invaluable resource all over the world. It is not surprising that asbestos' contribution to humanity was celebrated in the New York World's Fair in 1939 (Alleman and Mossman, 1997).

Since the early 1960s, asbestos began to lose its reputation (Alleman and Mossman, 1997) when it was found that inhalation caused lethal lung diseases. Asbestos is now one of the most feared contaminants. All asbestos minerals are classified as carcinogenic substances and their use is consequently restricted or banned in 55 out of the 195 countries (28%) (*www.ibasecretariat.org*). In the other countries, only amphibole species are considered to induce mesothelioma, the major asbestos related lung disease, whereas *safe* use of chrysotile asbestos is allowed. Despite the concern about asbestos minerals, the top five producers in 2011 alone marketed more than 2 000 000 t/year of chrysotile asbestos.

In countries where asbestos is banned, it has been progressively removed from the environment and replaced by non-toxic synthetic fibres. Those countries where asbestos is banned dispose of it as hazardous waste. There are two disposal solutions, both with advantages and disadvantages: disposal in landfills for toxic wastes or remediation and recycling of the secondary raw material (SRM). According to European Directive 2008/98/EC, recycling should be the preferable solution, assuring a lower environmental impact and a reduction in the consumption of the primary raw materials. This is the basic guideline (*end of waste* concept) of the European waste framework criteria, which specify when certain waste ceases to be waste and obtains a status of a product (or a SRM).

This chapter will describe the process of thermal transformation of ACMs to produce a SRM to be recycled and used for various industrial applications. Although the process of inertization of asbestos via recrystallization at high temperature has been known for about 30 years, there are only a few examples of applications in large-scale industrial plants. The pioneering INERTAM-Europlasma industrial plant in Morcenx (France), operational since 1999, is the most important example. It uses a plasma torch system for the vitrification of ACMs at 1600 °C (Borderes, 2000).

The slow development of this technology is largely due to the perception of the environmental risks posed by such treatment plants, similar to the concerns about solid waste incinerators (Reams and Templet, 1996). Opposition to such waste disposal technology is often for more than technical reasons. The greater availability of information to the public does not necessarily improve understanding of risk (Krimsky, 2007).

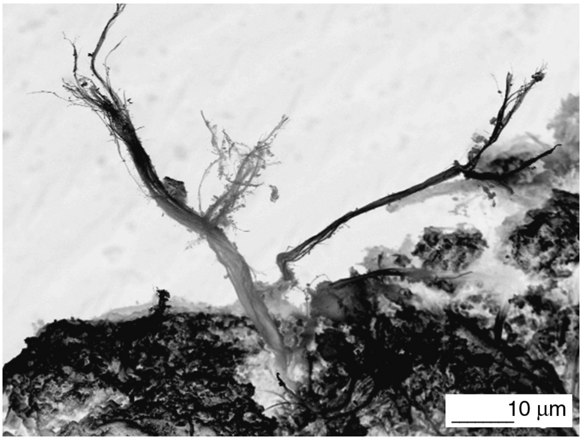
The environmental benefits of such waste disposal systems are undisputed but few would permit an industrial plant for the thermal destruction of asbestos to be built next door. This is the so called ‘Nimby’ (*not in my backyard*) syndrome (McAvoy, 1999). This attitude is aggravated by the combined fear of asbestos and thermal treatment, which is erroneously assumed to mean incineration. In fact, incineration is just the opposite of thermal treatment of asbestos. Incineration of solid waste at high temperature produces airborne particulate whereas the high temperature recrystallization of asbestos promotes the formation of safer, large crystals from newly-formed phases.

Among the technical arguments that slow the progress of the ACMs inertization technology is the absence of an *ad hoc* regulation dealing with such industrial plants. In countries such as Italy, only decrees (DM 29/07/04 n. 248) with general recommendations and commitments have been promulgated, but no specifications have been given to delineate the technical characteristics of the plants, to reduce environmental impact, and to set a protocol of quality control assessment to favour a standardized production and marketing of the SRM. With this case, little help comes from the European Community, because the recent REACH (Registration, Evaluation, Authorisation and Restriction of Chemical) substances regulation on chemicals and their safe use (EC 1907/2006) has been specifically intended for primary raw materials and its application to SRMs is more difficult.

It is the opinion of the author that the market of SRMs will have a bright future and ACMs represent a great social and economic opportunity. In this frame, the core of this chapter deals with a description of recycling of thermally treated cement-asbestos for the production of concrete and geopolymers.

20.2 Classification of asbestos minerals, health effects and use of asbestos as a building material

The modern term 'asbestos' was conceived in the 18th century in Germany to indicate natural materials (minerals) that occur as bundles of flexible fibres and that can be separated into thin durable threads. This unique crystal habit is called *fibrous-asbestiform*. Fibres are composed of smaller segments (Fig. 20.1) called *fibril*s (Skinner *et al.*, 1988). The length of a single fibril typically ranges from a few microns up to decimetres, while the length of a fibre ranges from a few microns to decimetres with a diameter typically smaller than 0.5 μm.



20.1 High resolution SEM image of a tree-like aggregate of chrysotile asbestos fibres in a cement asbestos matrix. The gray tones have been intentionally inverted.

Asbestos minerals are divided into two major categories: serpentine and amphibole asbestos. The fibrous-asbestiform variety of serpentine is called chrysotile. Chrysotile and amphibole asbestos are both silicates sharing fibrous-asbestiform crystal habits but holding very different structural units at a molecular scale (Whittaker, 1956; Yada, 1971; Bailey, 1988; Devouard and Barronet, 1995).

Chrysotile, the most common asbestos species, is a triocathedral hydrous phyllosilicate based on a 1:1 layer structure with a Si-centred tetrahedral sheet and an Mg-centred octahedral sheet. The ideal chemical formula is $Mg_3(OH)_4Si_2O_5$. The lateral dimension of an ideal octahedral Mg-centred sheet ($b = 9.43$ Å) is larger than the lateral dimension of an ideal Si-centred tetrahedral sheet ($b = 9.1$ Å). To a first approximation, this misfit is compensated in chrysotile by the curvature of the layer (Wicks and Whittaker, 1975; Wicks and O'Hanley, 1988), which results in an overall cylindrical lattice (Whittaker, 1956).

The group of amphibole asbestos minerals includes five species: actinolite $Ca_2(Mg,Fe)_5Si_8O_{22}(OH)_2$, tremolite $Ca_2Mg_5Si_8O_{22}(OH)_2$, anthophyllite $(Mg, Fe^{2+})_7Si_8O_{22}(OH)_2$, crocidolite (a fibrous variety of riebeckite) $Na_2(Fe^{2+},Mg)_3Fe_2^{3+}Si_8O_{22}(OH)_2$ and amosite (a fibrous variety of grunerite) $(Fe^{2+},Mg)_7Si_8O_{22}(OH)_2$. Amphiboles are double-chain silicates with the oxygen atoms of the chains coordinated to both Si and Al and to a variety of other cation sites (Veblen, 1981). In the general formula, $A_{0\text{-}1}B_2C_5T_8O_{22}(OH,F,Cl,O)_2$, T are the tetrahedral sites within the silicate chain, C are fairly regular octahedral cation sites (Mg^{2+}; Fe^{2+}, Mn^{2+}, Al^{3+}, Fe^{3+}, Ti^{3+}, Ti^{4+}; T = Si^{4+}, Al^{3+}), B are less regular octahedral or 8-fold coordinated cation sites (Na^+, Li^+, Ca^{2+}, Mn^{2+}, Fe^{2+}, Mg^{2+}), and A are irregular cation sites having coordination in the range 6 to 12 (Na^+, K^+).

The six asbestos minerals exhibit outstanding properties that have been greatly exploited in building materials. The major chemical-physical and technological properties of commercial chrysotile, amosite and crocidolite asbestos minerals are resistance to abrasion, resistance to heat (non-flammable even at very high temperatures), resistance to chemicals, flexibility, resilience, low sound transmission coefficient, high surface area, extremely high tensile strength and low thermal conductivity (Gualtieri, 2012). With such outstanding properties, asbestos minerals have been used for a number of industrial applications.

ACMs can be basically divided into *friable* and *compact* asbestos materials. Friable asbestos designates any ACM that can be easily crumbled or powdered when dry. Loose asbestos fibres can easily be scratched off the surface by hand. This ACM typology is composed of 70 to 95 wt% asbestos fibres. In general, friable asbestos in building materials can be found in: artificial ashes and embers for gas-fired fireplaces; fire-door gaskets in furnaces and wood stoves; cavities, partitions of floors and ceilings, the panels to lift shafts; patching and joint compounds for walls and ceilings; corrugated paper; textured paints/coatings, fireproofing spray; gaskets in pipes and vessel joints; insulating boards; insulation and covering of ventilation and air conditioning systems, freezers, clothes dryers and insulation of electrical wires and panels.

Compact asbestos is a composite material with asbestos fibres embedded in a cement or polymeric matrix. This ACM typology is not prone to release fibres, unless it is sawn or scratched by mechanical tools. In general, compact asbestos in building materials can be found in: bonding and finishing cement, masonry filler, mortars, mastics, caulk; ceiling tiles; asbestos-cement products with generally 4 to 15 wt% chrysotile asbestos and/or 0 to 6 wt% amphibole asbestos: planar or corrugated slates, pipes, insulating blocks; chimney tops; plasters; fire bricks; floor tiles, textiles and composites (linoleum, vinyl asbestos, flooring backing vinyl finishing, asphalt and rubber); wallboards; water tanks.

Unfortunately, the unique fibrous-asbestiform crystal habit and surface activity, responsible for asbestos' excellent technological properties, also seem to be the cause of its potential hazards. The history of the epidemiological reports of asbestos-related diseases is well described in Skinner *et al.* (1988). Although many epidemiological studies have provided evidence that amphibole asbestos minerals are more hazardous than chrysotile (Hodgson and Darnton, 2000), all six asbestos mineral species are assumed to be equally harmful to human health. Exposure through inhalation of asbestos minerals is supposed to provoke asbestosis, lung cancer or carcinoma, mesothelioma and pleural plaques (Skinner *et al.*, 1988; Dilek and Newcomb, 2003). In the 1980s, asbestos minerals were declared proven human carcinogens by the US Environmental Protection Agency, the International Agency for Research on cancer (IARC) of the World Health Organization and the National Toxicology Program (Nicholson, 1986; IARC, 1977; Collegium Ramazzini, 2010) and later many countries worldwide banned or restricted the use of ACMs.

The use of asbestos is restricted or banned in only 55 of 195 countries of the world. In the remaining countries, it is assumed that only amphibole asbestos minerals are carcinogens, whereas chrysotile asbestos is not (the so-called *amphibole hypothesis*). This position is based on the proviso that chrysotile asbestos has little potential for provoking mesothelioma (Liddell *et al.*, 1997; McDonald *et al.*, 1997; Camus, 2001) and that lung diseases are actually due to amphibole minerals (especially tremolite) contaminating chrysotile fibres.

In the pro-chrysotile countries, its use is permitted if it is controlled by technology or by regulations of work practices (*safe use*). Hence, chrysotile is still by far the most used natural fibre (constituting 94% of the world's production). The 2011 asbestos trade data (source US Geological Service) has reported that the top five producers of asbestos (t/year) are Russia (1 000 000), China (440 000), Brazil (302 300), Kazakhstan (223 100) and Canada (50 000), while the top five users of asbestos (t/year) are China (637 735), India (321 803), Russia (251 427), Brazil (185 332) and Kazakhstan (155 166). The largest user of chrysotile fibres is the asbestos-cement industry (85% of the total use) and more than 95% of the commercially developed asbestos ore deposits are chrysotile (Ross *et al.*, 2008). In the countries where all asbestos species are banned, man-made mineral fibres (MMMF) or man-made vitreous fibres (MMVF) are used as substitute materials in buildings.

20.3 The reclamation, disposal and recycling of asbestos-containing material (ACM)

In contact with atmospheric agents, ACMs may experience physical-chemical degradation and release airborne asbestos fibres. For this reason, the countries that banned asbestos gradually remove ACMs from the environment, following specific reclamation procedures.

20.3.1 Reclamation of ACM from outdoor and indoor environments

Reclamation of ACMs (mostly cement-asbestos) in outdoor environments is carried out by the following methods (Gualtieri, 2000):

- **Abatement**: this is the most applied method because asbestos is entirely removed from the contaminated site. However, hazardous waste is produced and has to be disposed (Cecchetti *et al.*, 2005).
- (b1) **Encapsulation *sensu lato***: spray-coating of the ACM with an acrylic based substance to form a thin coating that isolates the material from the environment.
- (b2) **Encapsulation *sensu stricto***: spray-coating of the ACM with a two-component epoxy resin that penetrates the matrix and fixes the fibres there (Gualtieri, 2000).
- **Isolation**: placing chemically inert, light, rigid panels (generally made of aluminium sheets) over the ACM to isolate the contaminated area (D'Orsi, 2007).

With encapsulation and isolation, asbestos is still present *in situ* and the status of the site should be monitored by following a protocol of periodical inspection and maintenance.

The methods for the ACMs reclamation in indoor environments are equivalent to those applied in outdoor environments. In addition to the methods described above, a viable method is chemical inertization by a specially designed foam that is sprayed onto the exposed area, which selectively decomposes the asbestos fibres (Raloff, 1998) and does not alter the cement matrix.

In both outdoor and indoor reclamation, workers are obliged to wear head to toe protection, such as overalls, and a high-efficiency particulate air (HEPA P3) respirator. With respect to the outdoor environment, the indoor reclamation site must be totally sealed by dividing the area into smaller lots, separated by temporary rigid gypsum boards. All the doors, windows, openings, sockets and air-conditioning systems must be sealed using polyethylene wraps. A dynamic containment applied by a negative air pressure unit keeps the building recovery site in negative pressure with respect to the external area and preclude fibre release (D'Orsi, 2007).

Whenever ACM is removed from the environment, hazardous waste is produced. Its final destination is an environmental issue with two disposal solutions, both presenting advantages and disadvantages: (a) disposal in landfill; (b) inertization via chemical-physical transformation (mechano-chemical, hydrothermal, recrystallization, vitrification, and others) and recycling of the transformation product as SRM.

20.3.2 Disposal of ACM

Although recycling in perspective seems to be more promising, to date ACM has been almost entirely disposed of as hazardous waste in specially commissioned landfills. In the last decades, active mines such as open pits and underground mines have been used as disposal sites for ACM waste, especially in European countries such as Austria, France, Germany, Italy, Slovenia, Sweden, Switzerland and the United Kingdom (Gidarakos *et al.*, 2008). Nevertheless, ACM landfills are a concern for the environment, as it cannot be guaranteed that there will be no risk of fibre dispersion in air and water.

Fibre dispersion during disposal operations may occur because, although they are carefully handled, the sealed packages of ACM frequently break or come in contact with water during the disposal operations. Fibre dispersion in the leachate occurs in the medium-long term (after 5–10 years). It was demonstrated that the polyethylene packages protecting ACMs are decomposed with time by percolating fluids which, in turn, slowly dissolve the cement matrix and induce fibre concentration in the leachate (Paglietti *et al.*, 2002). If the leachate is not collected and disposed of as hazardous waste itself, fibre dispersion in the open hydrological system surrounding the landfill is possible.

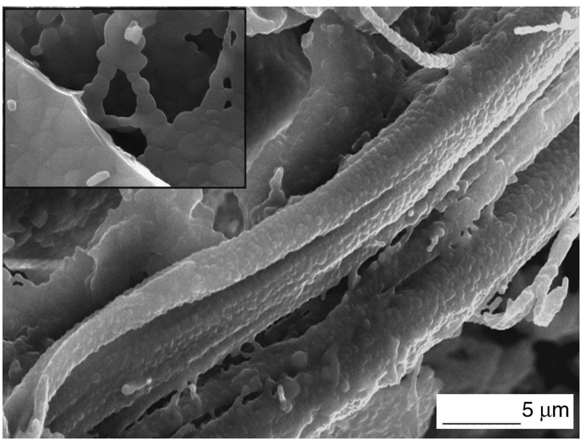
20.3.3 Recycling of ACM

The preference for recycling with respect to landfill disposal is specified in the Directive 2008/98/EC (19 November 2008). An alternative to landfill disposal of ACM is inertization and recycling of the transformed product as a SRM. Although thermal transformation at high temperature is the most common transformation process, other methods have been developed in the last two decades. A mechano-chemical process was conceived by Plescia *et al.* (2003), in which part of the mechanical energy transferred to solid systems is converted into heat and part is utilised to cause fractures, compression and slips at macro-meso- and microscopic levels, which affect the crystalline structure of solids including asbestos minerals.

The complete transformation of the asbestos fibres was achieved by a synergetic action of both high energy milling (HEM) and chemical attack. The concept of mechano-chemical treatment of ACMs in a ball mill was later applied by Inoue *et al.* (2007). HEM was used by Colangelo *et al.* (2011) for the treatment of both pure asbestos minerals and ACM. With this treatment, the asbestos minerals completely disappeared after two hours. By adding the transformation product to concrete mixes, materials with good pozzolanic activity were obtained.

Another way in which asbestos minerals can be destroyed is by dissolution in acidic media. The decomposition of chrysotile asbestos in fluorosulfonic acid (FSO_3H) aqueous solution was investigated by Sugama *et al.* (1998). From the equilibrium of FSO_3H in an aqueous medium ($FSO_3H + H_2O = HF + H_2SO_4$), the resulting H_2SO_4 displayed a strong affinity for the external brucite layers of chrysotile and led to its dissolution via leaching of the Mg^{2+} ions.

Thermal transformation of ACMs relies upon scientific evidence that all asbestos minerals are transformed into stable crystalline silicates at high temperatures (Martin, 1977; MacKenzie and Meinhold, 1994; Cattaneo *et al.*, 2003; Gualtieri and Tartaglia, 2000; Gualtieri *et al.*, 2008). In the range 650 to 750 °C, dehydroxylation of chrysotile asbestos occurs and is followed by solid state recrystallization into forsterite (Mg_2SiO_4) and subsequently enstatite ($MgSiO_3$). During the course of the reaction, chrysotile fibres preserve the same outer crystal habit, although the structure is completely changed at a molecular level. This phenomenon is called *pseudomorphosis* (Giacobbe *et al.*, 2010). Figure 20.2 is a high magnification SEM picture of a chrysotile fibre which has been fully recrystallized into forsterite and enstatite. Crocidolite asbestos is decomposed at around 1050 to 1100 °C through a complex reaction path involving iron oxidation (Gualtieri *et al.*, 2004), which leads to the formation of pyroxene ($NaFe–Si_2O_6$), enstatite ($MgSiO_3$), hematite (Fe_2O_3) and cristobalite (SiO_2).



20.2 A high magnification SEM micrograph of original chrysotile fibres fully recrystallized into forsterite and enstatite after a high temperature treatment at 1200 °C. The top left box is a picture zoom, curiously showing a transformed fibre aggregate with an ‘A’-like shape, evocating the ghost of (A)sbestos.

Tremolite is decomposed at 1050 °C into diopside ($CaMgSi_2O_6$), enstatite and cristobalite (Gualtieri and Tartaglia, 2000).

The same mineral assemblage is reported for actinolite in metamorphic systems, with the exception of quartz in place of cristobalite (Lledo and Jenkins, 2008). The presence of iron in the actinolite structure results in the formation of hematite at a high temperature in an oxidizing atmosphere. Amosite in air decomposes into spinel, hematite, pyroxene and amorphous phase in the temperature interval 800 to 1100 °C. Amorphous silica recrystallizes into cristobalite in the range 1100 to 1350 °C (Hodgson *et al.*, 1965). However, according to Jeyaratnam and West (1994), amosite decomposition leads to the formation of spinel, hematite, magnetite and cristobalite. According to Day and Halback (1979), anthophyllite high temperature products are forsterite and enstatite, eventually accompanied by cristobalite.

The number of research projects at laboratory and pilot scale and patents based on the thermal transformation of ACM found in the existing literature is so vast that illustrating them all is impossible. Vitrification is one of the most successful treatment processes. Roberts and Stuart (1989) patented a method for the inertization and re-use of ACMs by vitrification. Since ACM does not contain heavy metal, the benefit of vitrification derives from the complete destruction of the fibrous structure and the formation of a glass-forming mixture, which can be recycled as secondary glass material. Inaba *et al.* (1999) and Inaba and Iwao (2000) demonstrated that plasma torch vitrification converted asbestos into a rock-like structure.

In the INERTAM-Europlasma process, plasma torch vitrification of ACMs at 1600 °C takes place in a cylindrical furnace (Borderes, 2000). At the moment, this is the only method of conversion of ACMs that has been successfully converted from the lab scale into a fixed large-scale industrial plant. This industrial plant opened in Morcenx, France in 1999. In the industrial process, the subjection of the wastes to a plasma torch jet ensures complete vitrification of the products, whilst stirring ensures the bath homogeneity. The gases emitted during the thermal process are sent to a post-combustion chamber, operating at 1200 °C, where they are fully oxidized (Poiroux and Rollin, 1996). The vitrified product, termed *Cofalit*, is kept in a storage area for cooling and has been shown to be non-hazardous. It has the appearance of an obsidian glass and is generally recycled for use in road foundations. The INERTAM-Europlasma technology for the treatment of ACMs has also been successfully undertaken in Japan and in the UK by Tetronics Limited (Deegan *et al.*, 2006). It is expected to have a wider distribution in the future.

A valuable recycling solution for the product of plasma vitrification of ACMs was reported by Bernardo *et al.* (2011). Dense sintered crystalline materials from vitrified AC waste are obtained by fast heating processes and constitute the basis of glass ceramics and a new type of stoneware, with waste glass replacing conventional feldspar fluxes.

Recently, Min *et al.* (2008) proposed a melting furnace operating at 1450 to 1550 °C that uses a mixture of hydrogen and oxygen (Brown's gas) as a fuel to vitrify ACMs. The *GeoMelt* vitrification was designed by Finucane *et al.* (2008), while a method based on Joule-heating vitrification was recently developed by Dellisanti *et al.* (2009). In the latter case, the vitrification technology on a pre-pilot scale was applied to cement-asbestos pipes containing both chrysotile and crocidolite asbestos. The progressive heating up to 1600 °C led to the complete melting of fibrous minerals; the rapid cooling of the melt formed a monolithic glass.

Melting has the disadvantage of being extremely energy consuming and expensive due to the very high temperatures required. Lower recrystallization (ceramization) temperatures (750–1300 °C) are sufficient to convert ACMs, reducing the energy consumption and making the process economically more competitive. The CORDIAM project developed by Abruzzese *et al.* (1998) for the production of cordierite refractories is one of the pioneering ceramization processes of ACMs. Similarly, the Asbestex process (Johannes, 2003) employed a rotary kiln to thermally convert ACMs. The A.R.I. process (Downey and Timmons, 2005) proposed a thermochemical treatment for destroying asbestos and for treating other hazardous and radioactive wastes.

Microwave thermal treatment of ACMs (Leonelli *et al.*, 2006; Boccacini *et al.*, 2007) has also been reported. A combination of microwave, infrared heating and chemical treatment is the core of the system installed on a mobile asbestos decontamination unit, which was devised to disintegrate the ACMs on-site (Gerdes *et al.*, 2007). An industrial process for the direct thermal conversion of cement–asbestos, named KRY•AS, has been recently developed (Gualtieri *et al.*, 2008). Sealed packages of cement–asbestos materials are thermally treated in a tunnel kiln in the range 1200 to 1300 °C. It was observed that the transformation sequence of the asbestos and cement phases assemblage is totally different with respect to that of the pure asbestos minerals, due to extra-crystalline reactions. The resulting product is mainly composed of SiO_2 and CaO and shows a chemical nature comparable to that of a Mg-rich clinker (Gualtieri *et al.*, 2008).

A method based on hydrothermal conversion combined with acidic attack of friable asbestos has been developed by Yanagisawa *et al.* (2009). The acidic gas was generated by the decomposition of $CHClF_2$ with superheated steam. Chrysotile, crocidolite and amosite were decomposed by the reaction of $CHClF_2$-decomposed acidic gas at much lower temperatures than the traditional melting method. The same research team proposed thermal treatment of asbestos minerals in a water vapor atmosphere at 800 °C for 2 h (Kozawa *et al.*, 2010). On the same lines, hydrothermal treatment of ACMs in a temperature range of 300 to 700 °C and a pressure range of 1.75 to 5.80 MPa is described by Anastasiadou and Axiotis (2010).

ACM converted into SRM offers attractive recycling solutions. The literature contains plenty of examples describing recycling opportunities for SRM derived from ACM inertization. The product of transformation of cement-asbestos has been successfully recycled for the production of clay bricks, glass, glass-ceramics, ceramic frits, ceramic pigments and plastic materials (Gualtieri *et al.*, 2010). Similar studies on the thermochemical inactivation of AC wastes and the recycling of the mineral residues in cement products were done by Yvon and Sharrock

(2011). They found good parameters of tensile stress and mechanical strength by incorporating the SRM into mortars up to 10 %wt. Recently, magnesium sulphate whiskers were prepared by sulphuric acid hydrothermal leaching of the product gained from the calcination of chrysotile asbestos tailings (Cunjin *et al.*, 2012). The product of calcination of chrysotile asbestos tailings was also used for the preparation of diopside-based glass ceramics (Ding *et al.*, 2012). In the next section, recycling thermally treated cement-asbestos for the production of concrete and geopolymers is described.

20.4 Recycling cement asbestos for the production of concrete

Recycling SRMs in concrete, one of the basic ideas of the green concrete concept, is becoming a field of research that has growing interest worldwide, due to the environmental impact of the concrete industry (Bertolini *et al.*, 2004). The recycling of construction and demolition waste (CDW) became a socio-economic power within the European Union (EU). This was prompted by the results of the research by RILEM, the international union of testing and research laboratories for materials and structures (Hansen, 1985). In this scenario, there are several examples of recycling SRMs for the production of concrete. A few notable examples are:

- recycling demolished masonry rubble to create coarse aggregate (Khalaf and DeVenny, 2004);
- production of concrete bricks and paving blocks using recycled aggregates obtained from CDW (Poon *et al.*, 2002);
- mineral stone slurry used to replace fine aggregates (Almeida *et al.*, 2007);
- concrete reinforcement obtained by introducing ground tire rubbers (Bignozzi and Sandrolini, 2006);
- replacing part of Portland cement (PC) with fly and bottom ash recovered from municipal solid waste incinerators (Ferreira *et al.*, 2003; Bertolini *et al.*, 2004);
- plastics and glass in place of fine aggregate (Batayneh *et al.*, 2007);
- recycled glass fibre that is reinforced with plastic waste powder and fibre (Asokan *et al.*, 2009); and
- recycled marble cutting wastes (Mashaly *et al.*, 2012).

Recycling of cement–asbestos for the production of cement has also been attempted (Ambrosius *et al.*, 1996; Sprung *et al.*, 1998) and a process for the use of ground cement–asbestos as a raw material for the production of cement has been patented (Gleichmar *et al.*, 1997).

This chapter describes the recycling process of the product of thermal transformation of cement-asbestos in concrete, an application that is fully described by Gualtieri and Boccaletti (2011). The SRM was prepared by heating a standard sealed pack of 61 cement–asbestos slates, removed from an industrial site, at

1200 °C. The thermal treatment was carried out in an industrial tunnel kiln, with a heating ramp of 40 h, a 21 h long isothermal step at 1200 °C, and a cooling ramp of 8 h to return it to room temperature. The process and the product of transformation are described in Gualtieri *et al.* (2008) and Gualtieri and Boccaletti (2011).

The high temperature transformation of cement–asbestos promoted the crystallization (*retro-clinkerization* process) of cement phases such as C_2S and ferrite (ideally $Ca_4Al_2Fe_2O_{10}$) and Al-,Ca-,Mg-rich silicates such as akermanite (ideally $Ca_2MgSi_2O_7$) and merwinite (ideally $Ca_3MgSi_2O_8$). The chemical composition (wt% oxides) of the SRM used for this application is: SiO_2 = 26.44; Al_2O_3 = 4.17; Fe_2O_3 = 3.53; TiO_2 = 0.23; CaO = 50.10; MgO = 10.69; Na_2O = 0.37; K_2O = 0.43; MnO = 0.08; loss of ignition (H_2O + SO_3 + CO_2) = 3.95. The quantitative mineralogical composition (wt%) is: C_2S = 67.2(2); ferrite = 7.8(3); akermanite = 2.0(3); merwinite = 2.5(4); quartz (SiO_2) = 0.3(1); periclase (MgO) = 8.7(2); amorphous = 11.5(1.0).

It should be noted that the critical aspect of this SRM is the assessment of the mean chemical and mineralogical composition, due to the compositional variability of the original ACM. Packages of ACMs from different plants undergoing the same thermal treatment may yield products with different phase compositions. One of the products shows a prevailing amount of β-larnite (C_2S, Ca_2SiO_5). The main difference between cement and this SRM is that the latter is higher in Mg and C_2S, which makes it comparable to a low temperature cement aggregate (Taylor, 1990). The SRM replaced cement in commercial concrete mixes. Five different compositions were prepared:

1. cement 15 wt%, water 8.6 wt%, aggregate 76.4 wt% (standard);
2. cement 14.25 wt%, SRM 0.75 wt%, water 8.6 wt%, aggregate 76.4 wt%;
3. cement 13.5 wt%, SRM 1.5 wt%, water 8.6 wt%, aggregate 76.4 wt%;
4. cement 12.75 wt%, SRM 2.25 wt%, water 8.6 wt%, aggregate 76.4 wt%;
5. cement 12.0 wt%, SRM 3.0 wt%, water 8.6 wt%, aggregate 76.4 wt%.

Nothing was added to the cement paste. The cement used was a commercial Normal Cement CEM II/A – L, Class 42.5 N. The aggregate was a commercial, natural Italian product with 53 wt% sand and 47 wt% gravel.

The rheological properties of the pastes were determined with the Abrams cone test (slump test: UNI EN 206-1). All of the hardened concrete products were subject to chemical and mineralogical analyses and electron morpho-chemical (SEM + EDS) observations, and they were studied by applying the standard concrete technological tests (compressive strength UNI EN 12390-3:2003 after 3, 7, 28, 60, and 90 days; flexural strength UNI EN 12390-5:2002 after 28 days; depth of penetration of water under pressure UNI EN 12390-8:2002 after 28 days). From the SEM and mineralogical study, it was observed that the addition of the SRM changed the setting behaviour of the pastes. The mineralogical evolution (formation of ettringite, protlandite, C–S–H. . .) accompanying the setting/hardening process is slowed down in the SRM-rich pastes. This is due to the presence of C_2S in the product of transformed cement-asbestos, and there is a slower rate of hydration with respect to C_3S (Brunauer and Greenberg, 1962; Menetrier *et al.*, 1980).

Figure 20.3 consists of selected SEM pictures of both the standard and the 20 wt% SRM-rich concrete mix. The different mineralogical association after 7 days is shown; the standard already contains well-formed portlandite crystals and C–S–H phases, while the 20 wt% SRM-rich mix still shows the ettringite acicular crystals. Figure 20.4 (modified from Fig. 5 in Gualtieri and Boccaletti, 2011) shows the variation of the weight fraction of C_2S, C_3S and amorphous phase, with the setting/hardening time of the standard and the 20 wt% SRM-rich concrete mix showing that the addition of SRM determines an increase in the amount of C_2S, at expense of the C_3S phase.

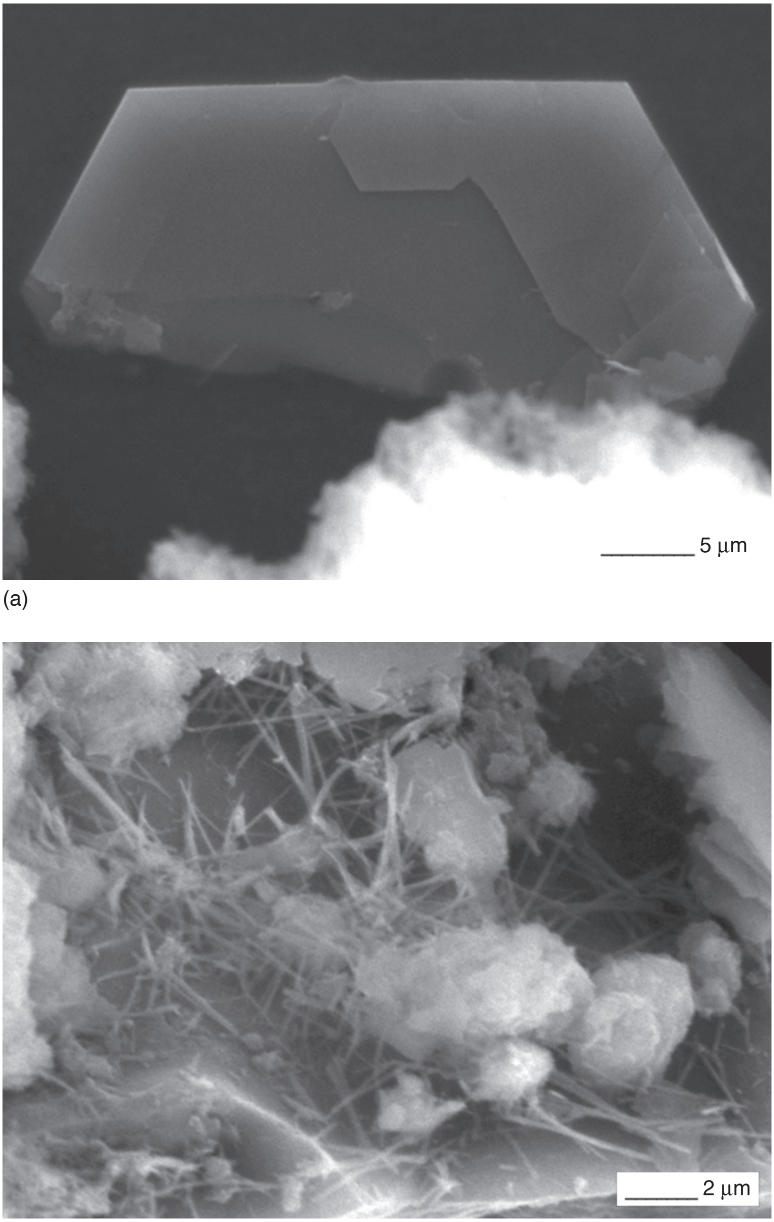
The slump tests, which indicate the rheological properties of the pastes, showed that all the samples are fluid or semi-liquid and possess good workability. It was observed that the compressive strength reaches values as high as 30 MPa after 90 days. After 28 days, the SRM-rich samples have a lower resistance to compression than the standard samples. Nevertheless, they are all classified as *ordinary* cements, according to the UNI 6132 tests. With time, the SRM-rich samples regain their compressive strength, showing values comparable to those of the standard after 90 days (Fig. 20.5, modified from Fig. 7 in Gualtieri and Boccaletti, 2011).

Again, this behaviour is explained by the presence of C_2S in the SRM-rich compositions, which decelerates the kinetics of setting/hardening. The slower kinetics of hydration of C_2S, with respect to C_3S, has a direct consequence on the trend of compressive strength increasing with hardening time (Taylor, 1990). This shows that the compressive strength of C_2S at an earlier point in the hardening period is very low with respect to that of C_3S, but becomes comparable after longer hardening times. This model fits with the results gained from observing the C_2S-rich samples which, after longer hardening times, display a compressive strength identical to that of the standard (Fig. 20.5).

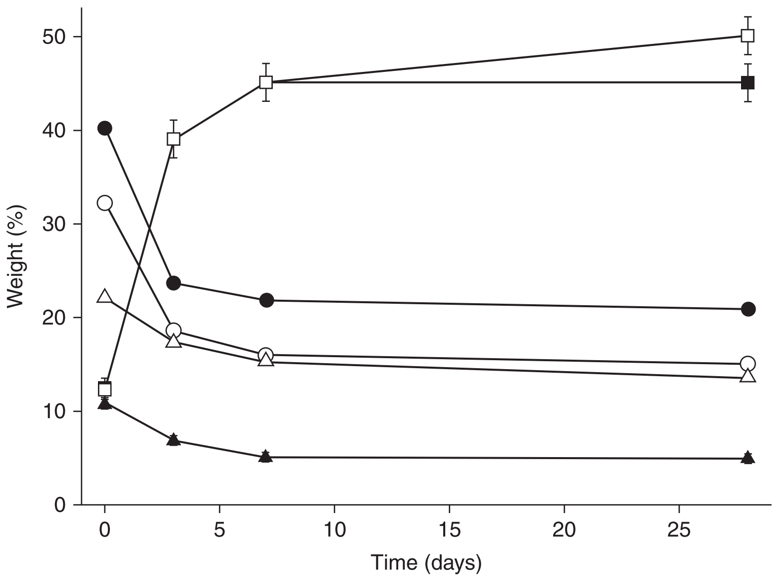
The same explanation is used to interpret the results of the flexural strength test. After 28 days, the property of C2S as a hardening retardant agent is evident in the samples diluted with the SRM, and it shows lower values compared with the standard. The average modulus of flexural resistance (N/mm^2) is 3.75(5) for the standard, and 3.30(39), 3.16(17) and 2.80(39) in the 10, 15 and 20 wt% SRM-rich samples, respectively. According to the classification of concrete based on the values of the flexural strength, the sample with the highest content of SRM (20 wt%) belongs to the T2.5 class.

The results of the penetration depth of water in the hardened concrete samples are all acceptable, although the penetration depth increases linearly with the SRM addition.

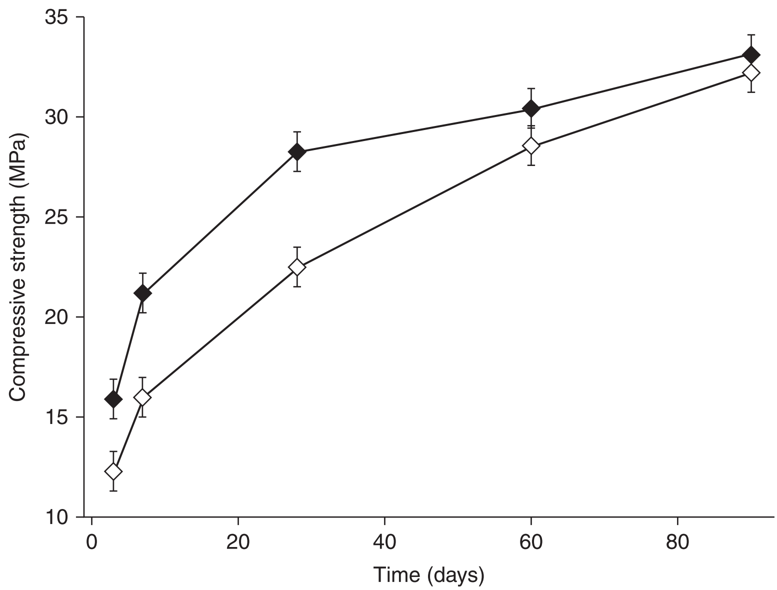
The overall results of this recycling investigation are very promising. According to the UNI EN 206-1:2006 standards, a concrete product with as much as 20 wt%



20.3 SEM micrographs of the standard: (a) and the 20 wt% SRM-rich concrete mix; (b) after 7 days with the standard exhibiting exceptionally well-formed portlandite crystal, whereas the 20 wt% SRM-rich mix still showing ettringite acicular crystals.



20.4 The evolution of C_2S, C_3S and amorphous phase with setting/hardening time for the standard and 20 wt% SRM-rich concrete mix. Legend: ● = C_3S standard; ▲ = C_2S standard; ■ = amorphous standard; ○ = C_3S 20 wt% SRM-rich mix; △ = C_2S 20 wt% SRM-rich mix; □ = amorphous 20 wt% SRM-rich mix.



20.5 Trend of the compressive strength for the standard concrete mix and 20 wt% SRM-rich concrete mix, after 90 days. Legend: ♦ = standard; ◊ = 20 wt% SRM-rich mix.

of the high temperature product of transformed cement-asbestos may be utilized for:

- indoor environments with very low moisture content;
- reinforced concrete injected in chemically inert soil or unaggressive water;
- unreinforced concrete under periodical dry/wet cycles, but not subject to abrasion, frost or chemical attack; and
- liquid storage tanks or foundations.

Recycling this SRM for use in the place of cement in concrete entails two major advantages for the environment:

1. reduction of the request for natural raw materials such as calcium carbonate, which then limits the environmental impact caused by the mining activity;
2. substitution of commercial cement in favour of a SRM in concrete decreases the emission of carbon dioxide, which is unavoidably formed during the clinkerization step of the cement production.

20.5 Recycling cement asbestos in geopolymers

Geopolymers are a class of synthetic Si/Al-based materials classified as chemically bonded ceramics (Davidovits, 1999). They are an alternative to cements or mortars as *green building* materials, as they do not emit carbon dioxide when they are produced. Ideal geopolymer precursors (e.g. metakaolinite, volcanic ash, tuff or pozzolanic materials) are alumino-silicate and their amorphous or paracrystalline structure assemblage includes SiO_4^{4-} and AlO_4^{5-} groups. If these materials are finely milled and put in strong basic ambient, they are partially solubilized and, in the presence of a catalyst (i.e. a sodium-silicate solution), polymerize so as to create a three-dimensional (3D) structure with covalent links (Davidovits, 1999).

Geopolymerization is a two-step process, involving:

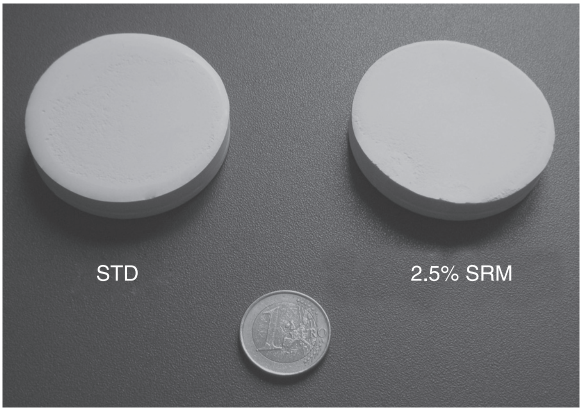
1. Dissolution of the precursor (generally metakaolinite) in strong basic ambient obtained with NaOH or KOH solutions, with the rupture of the original crystalline structure and the formation of monomers, constituted by the tetrahedral Si,Al-centred blocks.
2. Elimination of water linking the tetrahedral groups, causing polycondensation or geopolymerization (Davidovits, 2008). The structure of the inorganic polymers is typically a Si–Al 3D gel, highly coordinated, with the negative charges in the Al-centred tetrahedral sites balanced by the metal alkaline cations (Davidovits, 2008).

Geopolymers are considered green materials because, unlike cements, they do not need clinkerization at 1300 to 1400 °C, but rather thermal activation of the Si/Al precursor at 500 to 750 °C. Their technological properties include low viscosity and easy cast forming, good workability, sealing power, high hardness, absence of thermal stresses caused by thermal gradients and dimensional stability (Medri, 2009). Today, recycling SRMs for the production of geopolymers is a widely debated topic. Recently, Pacheco-Torgal *et al.* (2008) recycled tungsten mine waste mud to design geopolymeric binders. The effect of Ca-rich compounds on the technological properties of fly ash geopolymers has been studied (Temuujin *et al.*, 2009). Yip *et al.* (2005; 2008) focused their research on the recycling of seven different Ca-based silicate materials (e.g. blast furnace slag and others) in geopolymers, in an attempt to understand the role of Ca in the geopolymerization process.

Gualtieri *et al.* (2012) investigated a family of geopolymers prepared using meta-halloysite (the 750 °C activated product of halloysite $Al_2(OH)_4Si_2O_5 \times 2H_2O$) and the SRM obtained by the thermal transformation of cement-asbestos at 1200 °C. The chemical composition (wt% oxides) of the SRM used for this application is: SiO_2 = 30.8; Al_2O_3 = 5.4; Fe_2O_3 = 3.8; TiO_2 = 0.3; CaO = 48.5; MgO = 7.5; Na_2O = 0.4; K_2O = 0.3; loss of ignition ($H_2O + SO_3 + CO_2$) = 2.8. The quantitative mineralogical composition (wt%) is: C_2S = 71.2(2); ferrite = 4.8(3); akermanite = 1.0(3); merwinite = 7.5(4); periclase (MgO) = 8.7(2); amorphous = 6.8(1.0).

Geopolymer compositions containing different amounts of SRM (0, 2.5 and 5 wt.%) were prepared by adding the SRM to meta-halloysite, a water solution of sodium hydroxide (>7.5 M) and sodium trisilicate. The starting (STD) geopolymer mix was prepared following a standard procedure described by Davidovits (2008): 8.1 mol water, 0.47 mol sodium oxide, 1.65 mol SiO_2 and 0.41 mol Al_2O_3. The suspensions were homogenized for 5 min and poured into dies, where they were kept at 25 °C for 24 h. The best obtained formulation contained 2.5 wt% SRM (Fig. 20.6).

In alkaline environment, the Ca-rich crystalline silicates (akermanite, C_2S, ferrite) and amorphous phase of the SRM dissolve and make elements such as Ca and Si available for the geopolymerization reaction. Ca plays a major role in geopolymerization. Exclusively in geopolymer systems at low alkalinity, Ca reacts with Si and water to form C–S–H phases, the hydration products commonly

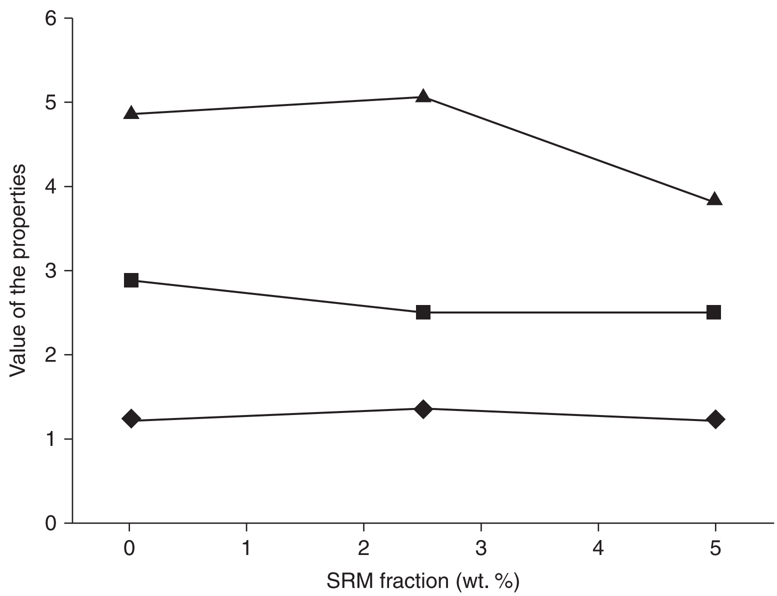


20.6 The geopolymer realized with 2.5 wt% SRM obtained from the thermal transformation of cement-asbestos compared to the pristine geopolymer.

observed during the hardening/setting process of cements (Yip *et al.*, 2008). With high NaOH (>7.5 M) concentrations, a geopolymeric gel is the predominant phase formed, with small calcium precipitates scattered within the binder. In the investigated systems, with a high degree of alkalinity (NaOH > 7.5 M), calcium precipitates are formed, although it is not possible to rule out the presence of C–S–H since, when it is present in low concentrations, it is nearly invisible to X-rays.

The value of the SiO_2/Al_2O_3 molar ratio, of paramount importance for the geopolymeric synthesis, changes with the SRM additions. The standard composition STD with a molar ratio SiO_2/Al_2O_3 of 1.39 holds a geopolymeric network with 95.4 wt% amorphous phase. In the mix with 2.5 wt% SRM and a SiO_2/Al_2O_3 molar ratio of 1.42, the crystalline phases are dissolved in the high pH ambient, and the final geopolymer contains 96 wt% of amorphous phase. In the mix with 5 wt% SRM and SiO_2/Al_2O_3 molar ratio of 1.44, the content of amorphous phase decreases to 93.6 wt%, due to an excess of the unlinked calcium in the liquid system, which reacts with atmospheric CO_2 to form calcite.

The results of the mechanical tests (Gualtieri *et al.*, 2012) indicate that an amount of SRM higher than 2.5 wt% degrades the performance of the geopolymer because of an excess of free Ca, which does not favour either geopolymerization or C–S–H formation. The values of density, water absorption and flexural strength of the designed composition (Fig. 20.7) are in agreement with those found



20.7 Major technological properties (density, water absorption and flexural strength) of the investigated geopolymers with different SRM contents (standard, 2.5 and 5 wt%). Legend: ▲ = flexural strength (MPa); ■ = water absorption (%) divided by 10 for clarity; ♦ = density (g/cm^3).

by other authors (Duxson *et al.*, 2007). In comparison with the standard (STD), the mix with 2.5 wt% SRM allows the obtainment of geopolymers that are characterized by lower water absorption and maximum value of density. The denser sample with 2.5 wt% SRM also displays a high value of flexural strength.

20.6 Future trends

This decade will hopefully bring the solution of the global issue of asbestos. It is expected that safe, novel synthetic fibres will be used as substitutes to asbestos; an Italian group recently managed to synthesize a safe form of chrysotile asbestos, the so-called nano-chrysotile (Falini *et al.*, 2004). The potential toxicity of asbestos substitute fibres that are used as building materials is another widely debated topic.

As far as ACM wastes are concerned, innovative technological solutions are ready to transform ACM into SRM (*end of waste concept*). One of the advantages is that the recycling of ACMs in building materials saves primary raw materials and decreases the overall emission of CO_2. New technologies for the transformation and recycling of ACMs have been developed recently and corresponding large-scale plants are expected in the coming years.

It should be highlighted that, depending on the chemical nature of primary ACM, the high temperature product of transformation contains up to about 70 wt% of phases possessing good hydraulic properties. However, this SRM contains magnesium oxide, considered an undesirable phase in PC because of the expanding properties of its product of hydration. In recent years, the need for a sustainable alternative to traditional PC (i.e. the so-called ‘green cements’) renewed interest in ancient building materials, which reduce carbon dioxide emissions.

The SRM produced after the thermal treatment of ACM wastes can be a suitable source of magnesium oxide, relevant in this respect because it is the main constituent of magnesium based cements. Magnesium cements have been developed for many years and comprise magnesia oxychloride cements (MOC), magnesia oxysulphate cements (MOS) and magnesia phosphate cements (MPC). Early magnesium cements were made with soluble phosphate from animal faeces or fermented plants, magnesia and clays. They were used in the Great Wall of China as well as in buildings around the world over many centuries. When energy was cheap and environmental concerns were not an issue, magnesium cements were progressively abandoned. They possess higher compressive and tensile strength compared to PC, and show a natural affinity for cellulose materials such as plant fibres or wood chips, in contrast to PC, which repels cellulose. This means that wood chips and cellulose materials can be used without special additives as aggregates to achieve a lighter weight and increased insulation.

To promote the formation of a magnesium phosphate cement fraction from the magnesium oxide found in thermally treated ACM, the key factors to be monitored are the reactivity and the weight fraction of the magnesium oxide that is formed. The knowledge of the latter allows for a correct determination of the proportion of phosphates to be added to the mix. The amount of magnesium oxide may be further increased by employing starting materials high in asbestos (i.e. loose asbestos or sprayed coatings) and by introducing appropriate fractions of calcium-rich components.

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The information contained in the selected web sites below must be cautiously considered and eventually cross-checked with peer reviewed scientific literature data, as it may be biased by personal views, exploited or distorted:

http://www.allaboutasbestos.co.uk
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21

Remediation processes for wood treated with organic and/or inorganic preservatives

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Abstract: Treated wood wastes disposal is becoming a challenge because of increasing amounts of treated wood wastes and restricted regulations regarding solid wastes landfilling or burning. Appropriate disposal options have been developed in recent years based on treated wood wastes recycling. However, treated wood wastes recycling options are limited because of organic or inorganic preservative agents. Recovery of these compounds is sometimes required to allow treated wood wastes recycling. This chapter reviews remediation technologies based on inorganic and organic compounds removal by physical, biological or chemical processes. It then discussed future trends concerning treated wood wastes recycling options.

Key words: pentachlorophenol (PCP), creosote, copper-based preservative-treated wood wastes, remediation, recycling.

21.1 Introduction

Wood preservation industries have impregnated wood products with oilborne and waterborne preservatives since the 1830s, to protect wood against insects, fungi and alteration from weathering. Preservatives help to extend the wood service time by 25 to 50 years (Cooper, 2003; McBain *et al.*, 1995). Among oilborne preservatives, creosote and pentachlorophenol (PCP) have been widely used to preserve wood products such as railroad ties, utility poles and timbers used for bridge or building construction. Creosote, a coal tar product, is the oldest commercial preservative system used worldwide for its fungal, insecticidal and molluscicidal properties (Schultz *et al.*, 2008). Creosote is still in use today in North America, mainly for railroad ties. In Europe, its use for industrial and commercial applications has been regulated and preservative industries need specific derogations to produce and sell creosote-treated products (OJEU, 2001). PCP has been widely used in wood preservation for its bacterial, fungal and algaecidal abilities (Kao *et al.*, 2005). In North America, PCP has been commercialized since 1941 (Lorber *et al.*, 2002).

Due to its toxicity, uses of PCP-treated wood are now restricted to commercial and industrial applications. Chromated copper arsenate (CCA), a waterborne preservative, has been widely used to treat exterior lumber for industrial applications over the past 60 years and for residential applications for 40 years. Some researchers have studied the potential risk of dispersion of the three components of CCA-preservative agents into the environment (Hasan *et al.*, 2010; Hingston *et al.*, 2001; Stirling and Morris, 2010). Results revealed that arsenic, chromium and copper can possibly be leached from treated wood products, resulting in metal dispersion in aquatic environments. Exposure to arsenic and chromium can cause inflammation of the larynx and liver, damage immune and nervous systems and can be responsible for skin, liver and kidney cancers (Cheng *et al.*, 2009; Kotas and Stasicka, 2000). Copper is an essential nutrient for plants, animals and humans; however, high concentrations can be responsible for liver and kidney damage, nausea, muscle pain, and mental disorder such as Wilson's disease and Parkinson's disease (Gaetke and Chow, 2003; Kamdem, 2008).

Since January 2004, CCA-treated wood products have been voluntarily replaced by alternative copper-based preservative-treated wood for most residential uses (Barnes, 2008; Freeman *et al.*, 2003; Pederson *et al.*, 2005). Among the new copper-based preservative agents, alkaline copper quaternary (ACQ), copper azole (CA) and micronized copper systems (MCQ and MCA) are the most common preservatives in the wood protection industry.

Large volumes of treated wood products are removed from service each year; however, quantities of discarded treated wood are different from one country to another. In the western United States, among the 800 000 utility poles discarded each year, 50% were treated with PCP (Langoodri *et al.*, 2012). According to Felton and DeGroot (1996), more than 6 million m^3 of creosote, PCP and CCA-treated wood are removed from service each year in the US and this amount could reach 19 million m^3 by 2020. Among creosote and PCP-treated wood discarded each year in the US at the end of the 1990s, 1.3 million m^3 came from railroad ties replacement and 2 millions m^3 came from utility poles (Felton and DeGroot, 1996).

In France, 1 million m^3 of creosote-treated railroad ties and 500 000 CCA-treated poles are removed from service each year (Mateus *et al.*, 2010; Sierra-Alvarez, 2007). In North America, approximately 3 to 4 million m^3 of CCA-treated wood is discarded each year and projections reveal that amount may increase to 16 million m^3 by 2020 (Sierra-Alvarez, 2007). The substantial amounts of treated wood in use today will certainly be discarded in the coming years. According to Cooper (2003), amounts of CCA-treated wood removed from service annually are estimated to increase from 1 million m^3 to 9 million m^3 in 2020 in the US. In Finland, an estimation revealed that 35 088 m^3 of creosote-treated wood will be discarded each year from 2010 to 2014 and 83 785 m^3 of CCA-treated poles will be removed from service each year at the same time.

Current treated wood wastes classification and management practices throughout the world are governed by national or local regulations. Treated wood wastes are classified as 'hazardous wastes' in Europe (decree no. 2002/540, 18 April 2002), whereas CCA-treated wood wastes are classified as 'non-hazardous wastes' in the US and Canada, according to exclusions from Resource Conservation and Recovery Act (RCRA 40 CFR § 264–4) in the US and from Hazardous Substance Regulation (RMD article 2.18) in Quebec. According to treated wood wastes classifications and management regulations, treated wood wastes can be disposed of in sanitary or lined landfills or burned in waste-to-energy or cement plants. In the majority of European countries, incineration is the preferred option for the disposal of treated wood wastes, whereas in North America, landfilling is the most common disposal option.

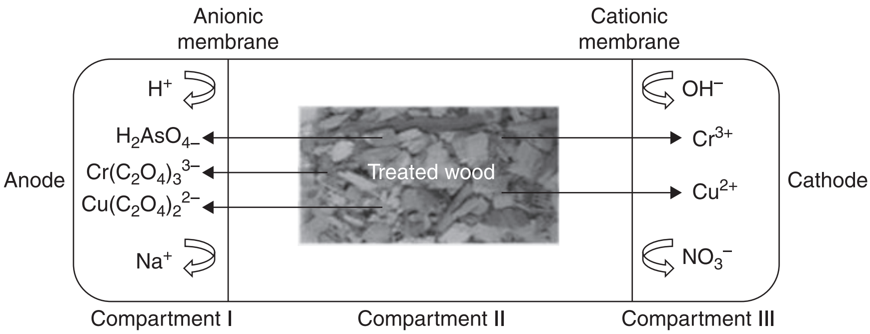
None of the existing disposal management practices is ideal, because treated wood wastes still contain toxic components at concentrations close to the original retention level. Indeed, creosote-, PCP- and copper-based treated wood wastes have to be carefully disposed of in landfill sites, because there are some risk of soil and groundwater contamination by polycyclic aromatic hydrocarbons (PAHs), PCP, dioxins and furans (PCDD/F) and metals (Illman *et al.*, 2002; Jambeck, 2004). Moreover, government strategies intend to reduce the volume of wood wastes entering landfills and prevent congestion of the landfill sites (*Landfill Directive 1999/31/EEC* in Europe and *Sustainable Development Act* in Canada). The most important issue regarding treated wood wastes incineration is potential As, PCDD/F and/or PAHs emissions, which can be partially resolved using appropriate and usually expensive filter systems. Production of highly concentrated metallic ashes, which should be classified as hazardous wastes, is another issue of environmental concern.

Actual disposal options are costly and are becoming increasingly less practical in some countries, because of stringent regulations related to treated wood wastes landfilling or burning (air emissions). The development of new sustainable options, based on treated wood wastes recycling, is therefore encouraged. Several remediation processes have been developed to remove organic and/or inorganic compounds from creosote-, PCP- and copper-based treated wood wastes. This chapter examines physical, biological and chemical approaches to remediate treated wood wastes, by extracting preservative elements from wood wastes and to convert them into value added products such as compost, bio-ethanol or heat.

21.2 Physical remediation processes for treated wood wastes

21.2.1 Electrodialytic remediation process (EDR)

Electrodialytic remediation (EDR) is an emerging remediation technology based on the application of a low-level direct current as a 'cleaning agent' to allow heavy metal removal from a contaminated matrix (Christensen *et al.*, 2006). This technology is based on the displacement of charged particles towards electrodes according to their charges by electro-migration, when an external current is



21.1 Schematic representation of an electrodialytic cell.

applied to the system (Ribeiro *et al.*, 2007). An electrodialytic cell is made of three compartments (Fig. 21.1): the anode compartment (I), the cathode compartment (III) and the central compartment (II), where treated wood wastes are placed. Ion membranes are used as selective barriers to separate the central compartment and the electrolyte compartments. The composition of these membranes allows the charged particles to migrate from the central compartment towards the anolyte or catholyte solution, but it prevents ions movement from the electrolyte solutions to the central compartment (Christensen, 2004).

Electrodialytic treatment was used for creosote-treated wood remediation. The sawdust initially contained 135 g/kg of creosote compounds, and electrolytic treatment allowed 40% of creosote removal after 8 days. Mateus *et al.* (2002) showed that polycyclic aromatic hydrocarbons (PAHs) and phenols mainly moved towards the anode, possibly due to electro-osmosis, whereas the positively charged N-heterocycles moved towards the cathode due to electro-migration.

Studies carried out in the laboratory (Ribeiro *et al.*, 2000; Velizarova *et al.*, 2002; Virkutyte *et al.*, 2005) and a pilot plant (Christensen *et al.*, 2004; Pederson *et al.*, 2005) with CCA-treated wood sawdust (Ribeiro *et al.*, 2000; Virkutyte *et al.*, 2005) and chips (Christensen *et al.*, 2004; Ribeiro *et al.*, 2007; Velizarova *et al.*, 2002) reveal that EDR can be successfully applied to CCA-treated wood wastes remediation. Ribeiro *et al.* (2000) examined As, Cr and Cu removal from CCA-treated wood sawdust by electrodialytic treatment (30 days) with various pre-treatments (demineralised water, oxalic acid (OA) solution at 2.5, 5 and 7.5%). The results highlighted that pre-treatment of CCA-treated wood in OA at 2.5% allowed better metal removal yields (99% As, 95% Cr and 93% Cu removal) than pre-treatment in demineralised water (27% As, negligible amount of Cr removed and 91% Cu removal).

EDR of CCA-treated wood chips, after several pre-treatments using demineralised water, sodium chloride, OA, formic acid, a mix of oxalic and formic acids and EDTA, has been also explored by Velizarova *et al.* (2004). Pre-treatment using OA solution at 2.5% for 24 h seemed to be the most efficient remediation process, with respectively 87% of Cr and 81% of Cu removed from treated wood. In the presence of OA, Cr and Cu ions were mainly found in the anolyte or trapped in the anionic membrane, whereas As ions ($H_2AsO_4^-$, $HAsO_4^{2-}$) were transported to the catholyte (Ribeiro *et al.*, 2007; Virkutyte *et al.*, 2005). Carboxylate groups of OA were identified as a ligand, which bound with Cr and Cu and formed water-soluble charged complexes ($Cr(C_2O_4)_3^{3-}$ and $Cu(C_2O_4)_2^{2-}$). These negatively charged complexes were then able to migrate towards the anode. According to Isosaari *et al.* (2010), a 3-step process, including pre-treatment using a solution of OA at 0.8% for 6 h followed by an electrodialytic treatment for 7 days and a post-treatment under the same conditions of pre-treatment, allowed removal of 81% of arsenic, 64% of chromium and 67% of copper.

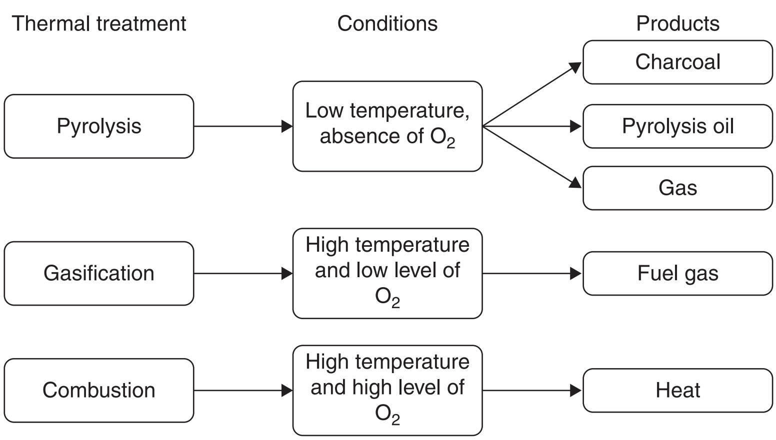
Metal removal efficiencies from CCA-treated wood wastes as well as the operating costs are affected by several parameters, such as wood particle size, applied intensity, electrodialytic retention time and distance between electrodes. Christensen *et al.* (2004) observed that 96% of As, 90% of Cr and 95% of Cu were removed from CCA-treated sawdust (<0.84 mm), while only 85% of As, 85% of Cr and 90% of Cu were removed from CCA-treated chips (1 to 5 cm in the longitudinal direction and 0 to 1 cm in the transversal direction). However, using chips instead of sawdust can be safer for workers who will be less exposed to airborne particles. Moreover, wood grinding is less expensive when using chips instead of sawdust, as the grinding usually requires huge amounts of electricity and time, which can be expensive (Christensen *et al.*, 2004). Ribeiro *et al.* (2007) found that pre-treatment of the wood chips with OA at 2.5% for 36 h allowed reduction of the electrodialysis treatment time from 32 to 25 days, without reducing the arsenic and chromium removal efficiencies. Copper extraction was greatly reduced because of formation of low soluble copper-oxalate complexes. They obtained respectively 99, 97 and 49% removal efficiency for As, Cr and Cu (Ribeiro *et al.*, 2007).

The research team in the University of Denmark designed a pilot plant for PCP of 2 m^3 of treated wood wastes (Christensen *et al.*, 2004; Pederson *et al.*, 2005). The electrodialytic cell was 3 m long, 1 m wide and 1 m high. Seven pilot-scale experiments were carried out with variable experimental conditions (amount of wood, distance between electrodes, pre-treatment solution, electrodialytic retention time, etc.). They identified the best experimental conditions for CCA-treated wood chips ($2 < \times < 4$ cm) with pre-treatment with phosphoric acid at 0.5 M for 24 h, followed by soaking in OA at 5% for 18 h. Then, the electrodialytic treatment was carried out with an applied intensity between 2 and 5 A for 21 days. The overall EDR process led to respectively 95, 81 and 87% of As, Cr and Cu removal. Treatment costs, including operating and capital costs, were estimated at \$275 per ton of treated wood (ttw) (210 €/ttw), whereas revenues, including remediated wood recovery for energy production, metal solutions re-use for wood treatment and saved costs related to treated wood landfilling, were estimated at \$255/ttw (195 €/ttw).

21.2.2 Thermal treatment

Thermal treatment of treated wood wastes is a good way to reduce the volume of solid wastes to be safely disposed of (between 98 and 99% reduction) and, at the same time, to provide energy using wood wastes as a resource. Three main thermo-chemical conversion processes have been developed to convert biomass (wood wastes) into useful energy forms such as heat, fuel gas or charcoal: combustion or incineration, gasification and pyrolysis. These techniques differ from each other, depending on thermal treatment temperature, the amount of oxygen required and the products obtained at the end of the thermal process (Fig. 21.2). Solid wastes combustion produces heat (5000–7560 MJ/m^3) that should be used immediately for heating, whereas pyrolysis or gasification treatments generate secondary energy products (Smith and Bolin, 2010). Gasification produces a fuel gas (CO, H_2 and CO_2) with heating values ranging from 5 to 20 MJ/m^3, depending on the level of oxygen used (Helsen and Van den Bulck, 2006).

Fuel gases produced during gasification can be burned to produce heat or used with an engine or a turbine to generate electricity. However, the pyrolysis allows production of charcoal, pyrolysis oil and gases such as CO, CH_4, H_2, C_2H_2 and C_2H_4. The proportion of each pyrolysis product mainly depends on temperature and heating rate. Between 573 and 823 K, fast pyrolysis takes place in a few seconds and results in high oil production levels, whereas slow pyrolysis is characterized by low heating rates and leads to maximum charcoal production (Helsen and Van den Bulck, 2006; Morrell, 2004).



21.2 Oxygen and temperature requirements of thermo-chemical treatments and thermal products formed.

Organic preservatives

Various pyrolysis and combustion studies have been carried out with creosote-treated wood wastes (Morrell, 2004; Salthammer *et al.*, 1994; Smith, 2005; Smith and Bolin, 2010; Zhurinsh *et al.*, 2005), PCP-treated wood wastes (Freeman *et al.*, 2000; Lee *et al.*, 2012; Morrell, 2004) and Cu-based treated-wood wastes (Helsen and Van den Bulck, 2004). Smith and Bolin (2010) and Smith (2005) carried out studies with creosote and PCP-treated wood wastes and demonstrated that organic compounds were completely degraded under heat, resulting in 99.9% elimination. Hence, they showed that air emissions were cleaner when burning creosote- or PCP-treated wood wastes than untreated wood wastes.

Freeman *et al.* (2000) designed a pilot experiment 'Combustion and Environmental Research Facility' to examine whether creosote- and PCP-treated wood wastes could be co-incinerated at 1253 K with coal (mixture of 10% in energy basis) without affecting air emissions quality. Co-incineration of these treated woods led to a decrease of NOx, SO_2, CO, PAHs, polychlorinated dibenzo-p-dioxins (PCDDs), polychlorinated dibenzofurans (PCDFs) and total hydrocarbon emissions compared to coal burning, whereas emission of acidic vapors (HCl) were increased because of the chlorine content in preservative agents. The authors concluded that PCP- and creosote-treated wood can be successfully co-fired without reducing the quality of air emissions.

These results were in accordance with a study by Salthammer *et al.* (1994), about creosote-treated wood combustion at 1273 to 1523 K. Combustion of PCP-treated wood wastes at 1123 K with excess of oxygen led to a decrease of semi-volatile organic compounds (SVOC) emissions compared to untreated wood; however, bromomethane, chloromethane and PCDD/Fs emission were higher than for untreated wood (Lee *et al.*, 2012). Slow pyrolysis treatment at 823 K for 10 sec were conducted on creosote-treated wood and revealed that organic compounds can be concentrated in the charcoal produced. Unfortunately, PAHs content in the produced charcoal were twice as high compared to untreated wood pyrolysis treatment. Benzo(a)pyrene content was seven times higher in creosote-treated wood charcoal than untreated wood charcoal (Zhurinsh *et al.*, 2005). High toxic organic content could restrict the use of treated wood charcoal. Combustion of creosote-treated wood is straightforward, whereas burning of PCP-treated wood wastes can be more challenging because of the risks of PCDD/Fs emissions and acidic gases (HCl). Hence, pyrolysis of oilborne-preservative treated wood at temperatures lower than 1373 K is restricted, due to accumulation of organic compounds in charcoal and PCDD/Fs emissions (Zhurinsh *et al.*, 2005).

Inorganic or organo-metallic preservatives

Combustion of CCA-treated wood has been widely studied over the last few decades (Helsen and Van den Bulck, 2005). Solo-Gabriele and Townsend (1999) carried out experiments on various samples of CCA-treated wood waste, as well as a mixture of CCA- (5%) and untreated wood (95%). As, Cr and Cu content in the ashes was determined, and TCLP tests were conducted in order to identify metal leachability from ash. All TCLP tests exceeded As and sometimes Cr regulations (5 mg/L), indicating that ashes should be disposed of as hazardous wastes. Speciation experiments conducted on ashes revealed that Cr and Cu compounds remained in the ash as water-insoluble solids and that Cr is mainly found in its hexavalent state (Helsen and Van den Bulck, 2004).

Pyrolysis treatment seemed to be a good option to reduce metallic emissions and the amount of Cr and Cu remaining in solid waste. The amounts of metals in ash and charcoal produced during CCA-treated wood incineration (1073–1173 K) and pyrolysis (573–673 K) were compared by Ottosen *et al.* (2004). Remaining metal content in the ash produced during incineration was 35 000 mg/kg for As, 62 000 mg/kg for Cr and 69 000 mg/kg of Cu, whereas metal content in the charcoal produced during pyrolysis was 990 mg/kg for As, 2500 mg/kg for Cr and 690 mg/kg for Cu. Differences observed between metal content in ash and charcoal can be due to a smaller weight reduction during pyrolysis than combustion.

A pyrolysis process called 'Charthem' was developed at an industrial scale (1500 kg/hr, 10 Gt/year). The Charthem process is a three-step process (wood crushing, thermal treatment and separation) able to operate with any treated wood wastes, and all preservative agents and initial concentrations (Hery, 2005). Hot gases (643 K) with a low oxygen rate (1.5%) were introduced at the bottom of the reactor full of crushed wood. A mineral residue with a high content of carbon (95–99%) was recovered at the bottom of the heat reactor and then transferred into a pneumatic centrifuge in order to remove heavy metals and minerals. Pyrolysis treatment of 1 metric ton of CCA-treated wood produced 280 kg of clean carbon, with a heating value of 27 MJ/kg and 50 kg of highly concentrated metallic residues (Hery, 2005).

Whatever the thermal treatment considered for CCA-treated wood waste remediation, As is the element of most concern because it is the most volatile compound. Release of Cr and Cu into the atmosphere was much lower than arsenic with respectively 11 and 15% volatilized for Cr and Cu and between 0 and 95% for As (Helsen and Van den Bulck, 2006). Decomposition of As_2O_5 and As_2O_3 compounds has been studied (Helsen *et al.*, 2003, 2004); however, thermal behavior of As compounds seemed to be different when As is fixed to wood.

Negligible amounts of arsine (H_3As) were released during CCA-treated wood combustion (Helsen and Van den Bulck, 2004). Arsenic release was controlled by reduction of As(V) to As(III), which was promoted by the presence of reducing compounds produced during treated wood combustion or pyrolysis. Arsenic compounds were first reduced as $As_2O_5 \rightarrow As_2O_3 + O_2$ with heating and then gasified according to the equilibrium $2As_2O_3 \leftrightarrow As_4O_{6(g)}$ (Helsen and Van den Bulck, 2004, 2006). Emissions of As in the atmosphere were a function of extended time of roasting of ashes at elevated temperatures. To avoid As emissions, the air flow needs to be lowered when incinerating at high temperatures (>1373 K);

however, these conditions are difficult to reach at industrial scale. According to Helsen and Van den Bulck (2004), pyrolysis at low temperature (653 K) in a moving bed appears to be the best option for CCA-treated wood wastes thermal treatment, in terms of metal content on ash and As emissions.

21.3 Bioremediation of treated wood wastes

Bioremediation processes have been used since the 1950s to remove organic or inorganic pollutants from contaminated material. The principle of bioremediation is to solubilize or to degrade hazardous components from solid or liquid wastes using bacteria or fungi. Toxicity or chemical stability of the toxic components can reduce bioremediation processes efficiencies (Clausen, 2006).

21.3.1 Bioremediation using bacteria

The bacterial degradation of persistent organic pollutants from culture media, soil, groundwater and treated wood wastes has been widely studied (Barbeau *et al.*, 1997; Carriere and Mesania, 1996; Clausen, 1996, 2006; Kao *et al.*, 2005; McBain *et al.*, 1993; O'Neil *et al.*, 1961; Saber and Crawford, 1985; Topp and Hanson, 1990; Zilouei *et al.*, 2008).

Organic preservatives

Bacteria, such as *Bacillus sp.* and *Pseudomonas sp.*, are able to degrade wood treated with oilborne preservatives, including creosote and PCP-treated wood. Bacteria are less mobile than fungi and have more difficulty in penetrating the wood cells; however; McBain *et al.* (1993) showed that mobility could be enhanced by swelling agents. However, bacteria are able to tolerate high retention levels of preservative agents used to prevent fungal attack during wood service time. A creosote-tolerant marine bacterium, *Pseudomonas creosotensis*, was identified by O'Neil *et al.* (1961) for detoxification of creosote-treated pilings. The bacteria *Aeromonas hydrophila, Flavobacterium sp.* and three species of *Pseudomonas* have been successfully assessed for bioremediation of creosote contaminated materials. These bacteria are creosote-tolerant (Clausen, 1996, 2006). Several studies revealed that biodegradation of 5- or 6-ring PAHs is much more difficult than that of 2- or 3-ring PAHs, probably because of the lower water-solubility of 5- or 6-ring compounds (Guerin, 1999; Glaser, 2012; Zhang, 2010). High degree of chlorination and toxicity make PCP more challenging to biodegrade.

Aerobic or anaerobic biological degradation of PCP using bacteria such as *Flavobacterium* sp., *Pseudomonas mendocina, Pseudomonas sp. Bu34, Antrobacter sp.*, *Rhodoccocus sp.* or indigenous bacteria from soil, have been explored (Barbeau *et al.*, 1997; Kao *et al.*, 2005; Nandish, 2005). *Flavobacterium* strains were able to degrade PCP from contaminated soil (Saber and Crawford,

1985) and from PCP-treated wood sawdust or chips within 2 to 3 weeks (McBain *et al.*, 1993). No PCP-degradation using *Flavobacterium* sp. or *Rhodococcus chlorophenolicus* was observed from PCP-treated wood wastes in a solid substrate system, whereas in liquid substrate system, 99% of PCP was degraded within 2 weeks by *Flavobacterium sp*. This observation can be explained by the fact that PCP was inaccessible to bacteria in solid substrate system (McBain *et al.*, 1993).

Inorganic or organo-metallic preservatives

CCA-treated wood waste bioremediation technologies are based on bacterial ability to convert heavy metals into soluble metal complexes. An advantage of using bacteria instead of fungi for CCA-treated wood wastes remediation is that bacteria are tolerant to metal concentrations that inhibit fungal growth, and shorter exposure times are required using bacteria instead of fungi (Chang *et al.*, 2012; Cole and Clausen, 1997). Bacteria such as *Bacillus coagularis, Bacillus licheniformis, Lactobacillus bulgaricus, Pseudomonas putida* and *Streptococcus thermophillus*; produce OA or lactic acid; which allow As and Cr solubilisation from solid wastes by increasing acidity (Chang *et al.*, 2012; Clausen and Smith, 1998; Cole and Clausen, 1997; Humar and Pohleven, 2004). OA is a strong organic acid and a good chelatant agent able to form Cr oxalate complexes ($Cr(C_2O_4)_3^{3-}$) and Cu-oxalate complexes ($Cu(C_2O_4)_2^{2-}$) (Kartal *et al.*, 2006; Sierra-Alvarez, 2007).

Cole and Clausen (1997) observed that *B. lichenformis* was more efficient than *P. putida* and *B. coagularis* to remove metals from a 20-year weathered CCA-treated wood sample. After 3 weeks of bacterial fermentation, 0 to 7% of As, 0 to 9% of Cr and 22 to 46% of Cu were removed from CCA-treated wood chips. Combination of metal extraction using a solution of 1% OA for 24 h followed by a bacterial fermentation using *B. licheniformis* for 10 days significantly increased the removal efficiencies of As, Cu and especially Cr compared to bacterial fermentation without pre-treatment (90 vs. 91% for CuO, 100% vs. 45% for As_2O_5 and 80% vs. 15% for CrO_3) (Clausen and Smith, 1998). Cr is slightly harder to solubilize using *B. licheniformis*, because of its stronger bonding to the wood (Chang *et al.*, 2012). Clausen and Kenealy (2004a) developed a two-step bioremediation process at pilot scale to allow metal solubilisation from CCA-treated wood wastes. Their process involved a combination of OA extraction and bacterial treatment using metal-tolerant bacterium *B. licheniformis*.

This approach to bioleaching clearly increased metal removal, with As, Cr and Cu removal of 88, 70 and 79%, respectively from CCA-treated wood particles and 81, 64 and 65 from CCA-treated wood chips after 7 days. However, OA and nutrient costs were estimated at $2140/ttw, which is expensive and may delay its development at industrial scale. A two-step fermentation process using *L. bulgaricus* and *S. thermophillus* bacteria, which both produce lactic acid, has been developed at laboratory scale by Chang *et al.* (2012). After two bacterial fermentation steps of 4 days each, 98% of As, 87% of Cr and 93% of Cu were removed from wood.

21.3.2 Bioremediation using fungi

Organic preservatives

A wide variety of persistent organic pollutants can be degraded by white-rot fungi (Pointing, 2001), such as *Inonotus dryophillis, Phanerochaete chrysosporium, Phanerochaete sordida, Trametes versicolor* and *Trametes hirsuta* (Alleman *et al.*, 1995), or by lignin-degrading fungi such as *Coriolus versicolor* (Hoshino *et al.*, 2008). Bioremediation of creosote-treated wood wastes can be achieved using white-rot fungal species, because they are able to degrade phenols and PAHs by producing extracellular redox enzymes. A bioremediation process using creosote-tolerant fungi, including *Antrodia radiculosa* and *Neolentis lepideus*, for degrading creosote components from treated wood wastes have been patented by Illman *et al.* (2002).

According to Galli *et al.* (2008), a consistent amount of PAHs (1500 μg/g) has a negative impact on mycelium growth and enzymatic activities related to pollutant degradation. Experiments carried out on mixtures of creosote-treated wood wastes, using *Agrocybe* sp., *Armillaria* sp., *Daedalea* sp., *Trametes* sp., showed a significant reduction (98%) of PAHs and heterocyclic compounds by oxidation of these pollutants (Galli *et al.*, 2008). *Pleurotus ostreatus* has also been identified for its ability to degrade PAHs from creosote-treated wood wastes (Polcaro *et al.*, 2008). Degradation of PCP from contaminated culture media, soil, groundwater and treated wood using white-rot fungi has been widely explored (Walter *et al.*, 2005a,b).

Mechanism of PCP biodegradation and its by-product formation using white-rot basidiomycete fungi has been explained by Reddy and Gold (2000). Studies revealed that *P. chrysosporium, P. sordida* and *T. hirsuta* can metabolize PCP from contaminated soil, reducing PCP content from 800 to 1000 mg/kg to 4 mg/kg after 74 days (Walter *et al.*, 2005a). According to Alleman *et al.* (1995), *T. versicolor* seemed to be the best choice for metabolizing PCP (62% of PCP degraded) compared to *P. chrysosporium* (38% PCP degraded) or *I. dryophillis* (21% PCP degraded). These results were in accordance with observations made by Tuomela *et al.* (1999) and Walter *et al.* (2004, 2005b). *T. versicolor* showed a good potential to biodegrade PCP with negligible amounts of pentachloroanisole (PCA) and 2,3,4,6-tetrachlorinasole produced compared to *P. chrysosporium*, which produces high levels of PCA (Ford *et al.*, 2007).

Even though PCA is less toxic than PCP, formation of this compound is undesirable, because it is still an environmental hazard due to its high bioaccumulation capacity (McBain *et al.*, 1993). A 3-step bioremediation process, including 2 methanol extraction steps of 3 days and 24 h and a sonification treatment in presence of white-rot fungus, *P. chrysosporium*, has been developed by Pal *et al.* (1997) to remove PCP from treated wood. Biological treatment under sonification increases the mass transfer rate and therefore enhances PCP mineralization. Three-stage counter-current extraction using ethanol and bioremediation treatment can decrease PCP content in treated wood from 12 000 mg/kg to less than 1 mg/kg. The fungal degradation of PCP from treated wood wastes by *P. chrysosporium, P. sordida, T. hirsuta* and *Ceriporiopis subvermispora* has been explored. Results indicated that *T. hirsuta* was the best lignin-degrading fungus to remove PCP from treated wood (84% of PCP removed by *T. hirsuta* vs. 59% by *P. chrysosporium*, 57% by *P. sordida* and 37% by *C. subvermisporia*) after 4 weeks (Lamar and Dietrich, 1992).

Inorganic or organo-metallic preservatives

Fungal abilities of brown-rot fungi, including *A. radiculosa, Antrodia vaillantii, Fomitopsis palustris, Gloeophyllum trabeum, Laetiporus sulphureus, Leucoyrophana pinastri, Meruliporia incrassata and Poria monticola* and white-rot fungi including *Polyporus hirsuta* and *T. versicolor*, have been described in several studies for removing metals from CCA-treated wood wastes (Clausen, 2006; Humar and Pohleven, 2004; Illman and Yang, 2004). Fungal production of OA, which highly depends on fungal species, increased brown-rot fungi growth on CCA-treated wood wastes. Metal solubilisation from treated wood wastes seemed to be due to OA production by bacteria or fungi and then, to metal bio-sorption onto cell membranes of fungi (Kartal *et al.*, 2004a; Kim *et al.*, 2010).

Twenty-four brown-rot fungi and ten white-rot fungi were evaluated for their abilities to degrade CCA-treated wood and to tolerate high amounts of As, Cr and Cu. Brown-rot fungi showed higher tolerance towards Cu and produced greater amounts of OA than white-rot fungi (Sierra-Alvarez, 2007). Among 106 fungi isolates, 5 brown-rot isolates, including *Clustoderma sp.*, *A. vaillantii, F. palustris* and 2 white-rot isolates were identified as highly Cu-tolerant and strong CCA-treated wood degraders. *F. palustris* seemed to be highly efficient for As (79% As_2O_5) and Cr (78% CrO_3) bioleaching, but moderately efficient for Cu removal (30% CuO) (Kim *et al.*, 2010).

According to Kartal *et al.* (2004b), 100 and 85% of As were removed from treated wood after 10 days in the presence of *F. palustris* and *L. sulphureus*, respectively. Among 10 Cu-tolerant fungi, *P. funiculosum* and *Aspergillus niger* were identified as the most efficient to degrade CCA-treated wood chips. *A. niger* has been widely studied for its ability to remove metals from treated wood wastes (Kartal *et al.*, 2004a). According to these authors, after 10 days, 97% of As, 55% of Cr and 49% of Cu were solubilised from CCA-treated wood wastes. Further experiments should be made to determine if *A. niger* can be successfully used for ACQ- and CA-treated wood wastes detoxification. *Wolfiporia cocos*, a copper-tolerant fungus, was able to detoxify Cu-based preservative-treated wood and to concentrate Cu in the mycelium (DeGroot and Woodward, 1999).

According to Illman and Yang (2004), *M. incrassata* was the most tolerant fungus, able to degrade CCA-treated wood among various isolates tested. Another brown-rot fungus, *A. vaillantii*, has been identified for its Cu-tolerant ability. Laboratory experiments showed that 70% of As, 50% of Cr and only 5% of Cu can be removed from treated wood after 16 weeks using *A. vaillantii* (Stephan and Peek, 1992). Based on laboratory-scale results, a pilot scale able to remediate simultaneously 1 m^3 of CCA-treated wood has been designed. After respectively 8 and 11 weeks, 70 and 90% of As and Cr were removed from wood. Treated wood wastes sterilization before biological treatment is required, because bacteria can totally inhibit *A. vaillantii* growth and therefore metal solubilisation (Leithoff and Peek, 1997). A recent study indicates that a pre-treatment of CCA-treated wood wastes using a solution of citric acid at pH 3.10 significantly increased Cu solubilisation from wood. This combination of citric acid extraction and fungal fermentation using *A. vaillantii* allowed 100% of As, more than 80% of Cr and 95% of Cu removal (Sierra-Alvarez, 2009).

21.4 Chemical remediation processes for treated wood wastes

Chemical remediation processes can be used to remove organic or inorganic components from treated wood wastes. This technology is based on toxic compounds extraction from a solid material into liquids (compounds leaching), using inorganic or organic acids, chelating agents or oxidizing agents. Some studies have explored HAPs and PCP extraction from soil and treated wood, using surfactant or oxidative conditions (Fenton oxidation) (Gan *et al.*, 2009; Mulligan and Eftekhari, 2009; Tran *et al.*, 2009; Valderrama *et al.*, 2009). According to Murray (2006), PCP can be successfully solubilized from treated wood wastes using sodium hydroxide, since PCP is converted to water soluble chlorophenates. Experiments carried out at laboratory scale revealed that solutions of NaOH at 0.5, 0.75 and 1.0 N removed 32, 58 and 99% of PCP, respectively, from treated wood sawdust (0–2 mm) after 4 h. At pilot scales (2 L and 1.500 L), PCP-treated blocks (1–3 mm thick, 10–30 mm wide and 150–300 mm long) were subject to a pressure process to improve the contact between the leaching solution (NaOH) and wood. The 2-step chemical process developed at pilot scale, including extraction steps of 2 h each using NaOH 1N at 1034 kPa and 91 kPa, allowed more than 99% of PCP removal. The following section focuses on Cu-based preservative treated wood wastes using chemical extraction technologies.

21.4.1 Remediation process at laboratory scale

Several studies have been carried out at laboratory scale to detoxify treated wood wastes by removing As, Cr and Cu. Metal solubilisation from wood is strongly influenced by extracting agent, chemical concentration, retention time, temperature and wood particle size. Chelating and oxidizing agents can be successfully used for extracting metals from wood by formation of soluble complexes and by oxidation of Cr(III) into its hexavalent form. Although more toxic, Cr(VI) is more water-soluble. Ethylenediaminetetraacetic acid (EDTA) is used for CCA-treated wood remediation, as a strong chelating agent that binds to Cu through its amino- and carboxylic-groups.

Kartal (2003) examined As, Cr and Cu solubilisation by extraction with EDTA. Cu solubilisation from CCA wood was more efficient than Cr and As extraction in the presence of EDTA. Remediation of CCA-treated wood, using EDTA, nitrilotriacetic acid (NTA) and OA, has been studied by Kartal and Kose (2003). Metal removal efficiencies were similar for NTA and EDTA. Single extraction using 1% EDTA removed 60 and 93% of Cu, 13 and 36% of Cr, 25 and 38% of As from respectively sawdust and chips. Dual extraction processes using EDTA/OA or NTA/OA removed about 100% of Cu and As and 90% of Cr. According to these authors, this remediation process can be successfully applied to ACQ-, CA- and MCQ-treated wood because of strong affinity of Cu for EDTA.

Metal solubilisation from CCA-treated wood by extraction with chelating agents (ethylenediamine-N,N′disuccinic acid (EDDS), EDTA and NTA) was studied by Ko *et al.* (2010). EDDS was more efficient than EDTA for Cr solubilisation (59–66% vs. 44–53%) and more efficient than NTA for Cu removal (85–93% vs. 79–84%). Kazi and Cooper (2006) examined metal extraction from CCA-treated wood using hydrogen peroxide, a strong oxidizing agent. Extraction with 10% H_2O_2 for 6 h at 323 K resulted in removal of 93% As, Cr and Cu. Gezer and Cooper (2009) extracted large amounts of CCA components from treated wood using a solution of sodium hypochlorite (NaOCl) as oxidizing agent to enhance Cr solubilisation. Increasing sodium hypochlorite concentration from 1.0 to 5.0% increased metal solubilisation from 56 to 79% for As and from 58 to 84% for Cr. Because Cr is removed from the wood as Cr (VI) form, it can easily be recycled in wood treatment plants for CCA treatment. Unfortunately, the high cost of chelating and oxidizing compounds for removing metals from treated wood wastes may be prohibitive.

Acid extraction processes have been widely explored due to acid ability to solubilize As, Cr and Cu. In order to prevent Cr and Cu precipitation, pH of chemical solution should be lower than 5; however, strong acidic conditions can damage the wood structure and reduce the number of possible recycling options for remediated wood (Kakitani *et al.*, 2006a). Gezer *et al.* (2006) studied the influence of pH of leaching solution on metal removal efficiencies. Extraction of As, Cr and Cu from CCA treated wood after 3 days using oleic acid decreased with increasing pH of solution from 2 to 5 (94 vs. 51% for As, 78 vs. 26% for Cr and 96 vs. 52% for Cu). OA has a good potential to remove metals from CCA wood and is a cheap raw material. According to Yu *et al.* (2010), As can be easily removed from treated wood by acetic and OA with respectively 33 and 98% removal at 363 K and 90 and 100% removal at 433 K. Among 6 organic acids and 2 inorganic acids tested by Sierra-Alvarez (2009), HCl and H_2SO_4 appeared to be more efficient than organic acids, even if OA enhanced Cr solubilisation and citric acid enhanced Cu solubilisation from treated wood wastes.

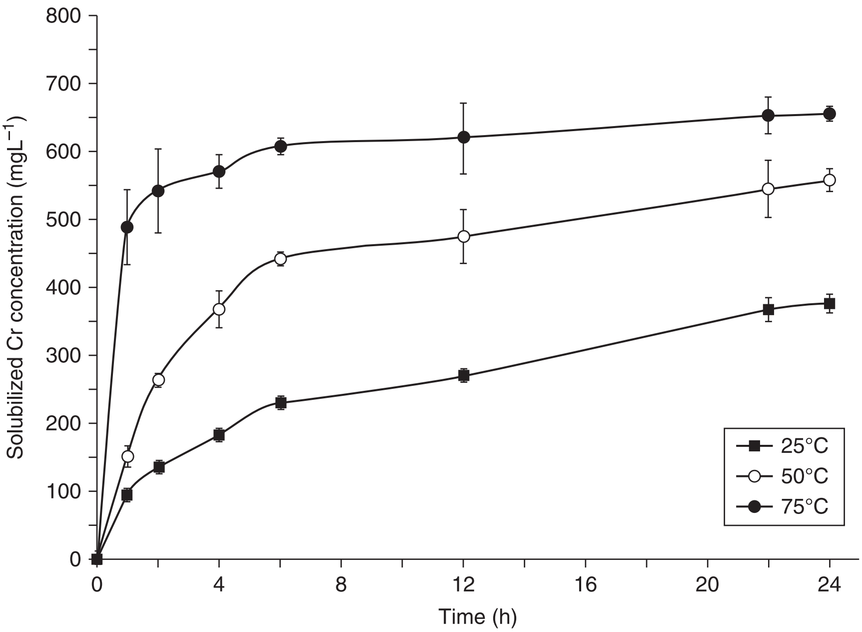
A combination of various acids can significantly increase metal removal efficiencies from treated wood wastes. After 10 min at 403 K using 0.5% acetic acid and 2.75% phosphoric acid, 99.7% As, 93.5% Cr and 98.5% Cu were removed from wood wastes (Yu, 2010). A bioxalate solution made of 0.125 M of OA and sodium hydroxide (pH 3.2) was examined for its ability to remove As, Cr and Cu from Cu-based treated wood. After 6 h of treatment at 348 K, about 90% of As, Cr and Cu were extracted from CCA-, ACQ- and CA-treated chips and sawdust (Kakitani *et al.*, 2006b). Cu removal efficiency using bioxalate solution was not affected by temperature between 298 and 348 K, whereas As and Cr extraction efficiencies were highly dependent on temperature of the leaching step (Kakitani *et al.*, 2007). A combination of citric acid and EDTA has been explored by Kamdem *et al.* (1998). After 18 h at 353 to 373 K, approximately 95 to 100% of As, Cr and Cu was removed from CCA-treated blocks.

Five reagents (H_2SO_4, H_2O_2, H_3PO_4, EDTA and OA) were tested for their ability to remove metals from treated chips and the results are presented in Table 21.1 (Janin *et al.*, 2009a). Sulfuric acid seemed to be a good compromise in terms of As, Cr and Cu removal efficiencies, with respectively 67, 48 and 100% removal with 0.07 N acid concentration. The chemical cost is about $9/ttw. Leaching conditions (acid concentration, temperature, retention time and number of leaching steps) were optimized in order to enhance metal removal efficiencies. Experiments showed that all of these leaching parameters significantly affect metal solubilisation. Indeed, increasing acid concentration from 0.002 to 0.5 N strongly improve extraction of As, Cr and Cu from treated wood, but between 0.5 and 1.0 N, metal solubilisation showed little increase. Temperature and retention time are key parameters that influenced Cu and especially As and Cr extraction efficiencies and rates (Fig. 21.3).

Table 21.1 Influence of chemical reagent on arsenic, chromium and copper removal efficiencies (%) and associated chemical costs

Chemical agents	H_2SO_4	H_2O_2	H_3PO_4	EDTA	Oxalic acid
Reagent concentration	0.07 N	10%	0.06 N	20 g L^{-1}	0.07 N
As (%)	67.3	71.2	31.1	19.7	79.9
Cr (%)	48.2	57.7	11.0	3.5	61.2
Cu (%)	100	82.7	92.6	99.7	49.3
Chemical costs ($/ttw)	9.1	4 616	166	960	84.0

Notes:
Wood content = 50 g/L^{-1}, T = 25 °C, reaction time = 22 h, particle size = from 0.5 to 2 mm.



21.3 Kinetic of chromium solubilisation from CCA-treated chips during sulfuric acid (0.2 N) leaching remediation process at various temperatures (25, 50 and 75°C).

Finally, 3-times two h leaching steps at 348 K with an acid concentration fixed at 0.2 N seemed optimal in terms of metal removal efficiencies and operating costs. Under these conditions, up to 99% of As and Cu and 90% of Cr were removed from treated wood chips for a total cost of $115/ttw (chemical and capital costs). Acid leaching process produced highly concentrated effluents, which need to be treated before discharge. Precipitation, electrodeposition and ion exchange were evaluated to decrease the metal and organic matter content of the leachates. The authors identified the appropriate treatment conditions to reach regulation levels for effluent discharge in sewers for Quebec City and Montreal (Janin *et al.*, 2009b,c).

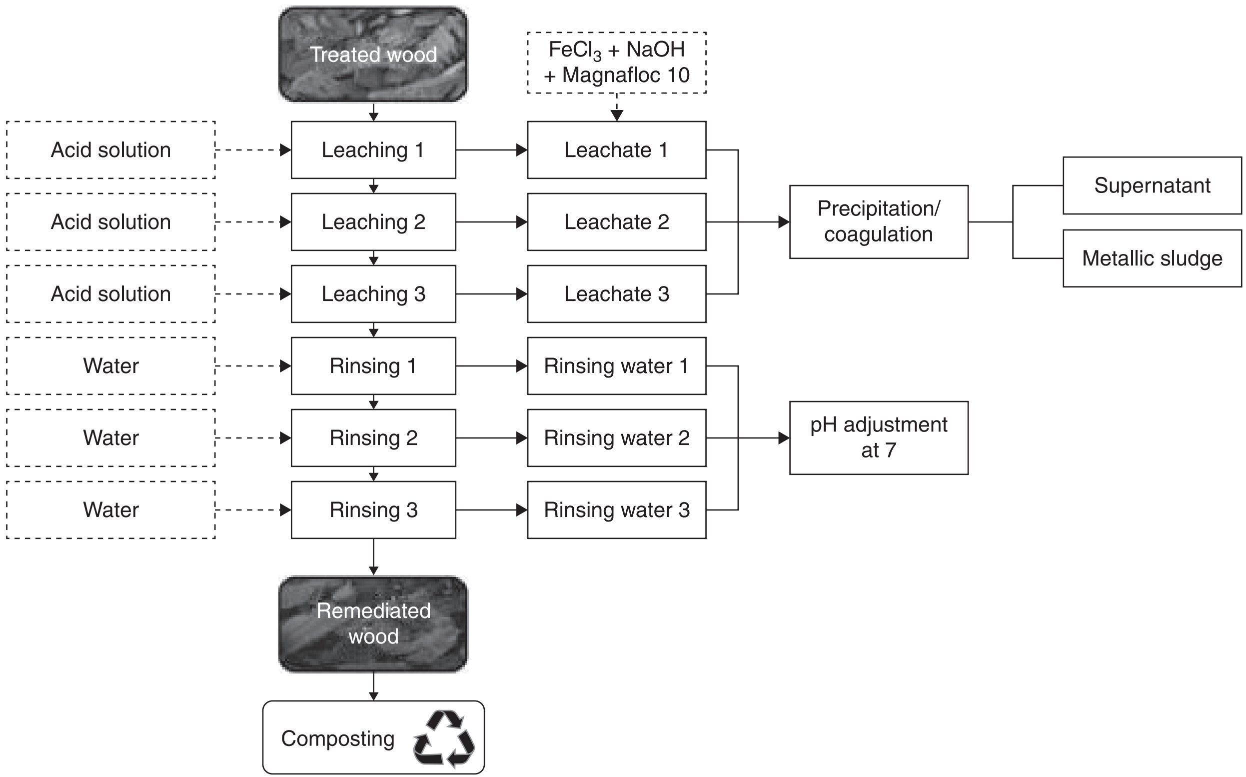
A recent study explored performances of this leaching process for removing Cu from alternative Cu-based treated wood including ACQ-, CA- and MCQ-treated timber, as well as Cu removal from the effluents by precipitation or electrodeposition (Janin *et al.*, 2011). Approximately 98% of Cu was extracted from ACQ-, CA- and MCQ-treated timbers. Therefore, the authors concluded that this acid leaching process can be successfully applied to alternative preservative-treated wood wastes recycling. However, operating conditions seemed to be too stringent and acid concentration or temperature could be reduced without affecting Cu removal efficiency.

In order to optimize leaching conditions for extracting Cu from ACQ-, CA- and MCQ-treated wood, Cu solubilisation from wood was modeled using a response surface methodology (Box Behken Desing: BBD) by Coudert *et al.* (2013). Acid concentration is the main parameter influencing Cu solubilisation from treated timbers followed by the number of leaching steps and then by the temperature and retention time. Optimal leaching conditions were identified as follows: 3 leaching steps of 2 h 40 min each, at room temperature with an acid concentration fixed at 0.13 N, followed by 3 rinsing steps. Approximately 90% of Cu was recovered from ACQ-, CA- and MCQ-treated wood. The use of the Box Benhken Design allowed the reduction of total costs (direct, indirect and capital costs) from \$305 to \$178/ttw without affecting Cu removal (Coudert *et al.*, 2012, 2013). According to these results, a chemical leaching process appeared to be an interesting solution to resolve Cu-based treated wood wastes disposal problems.

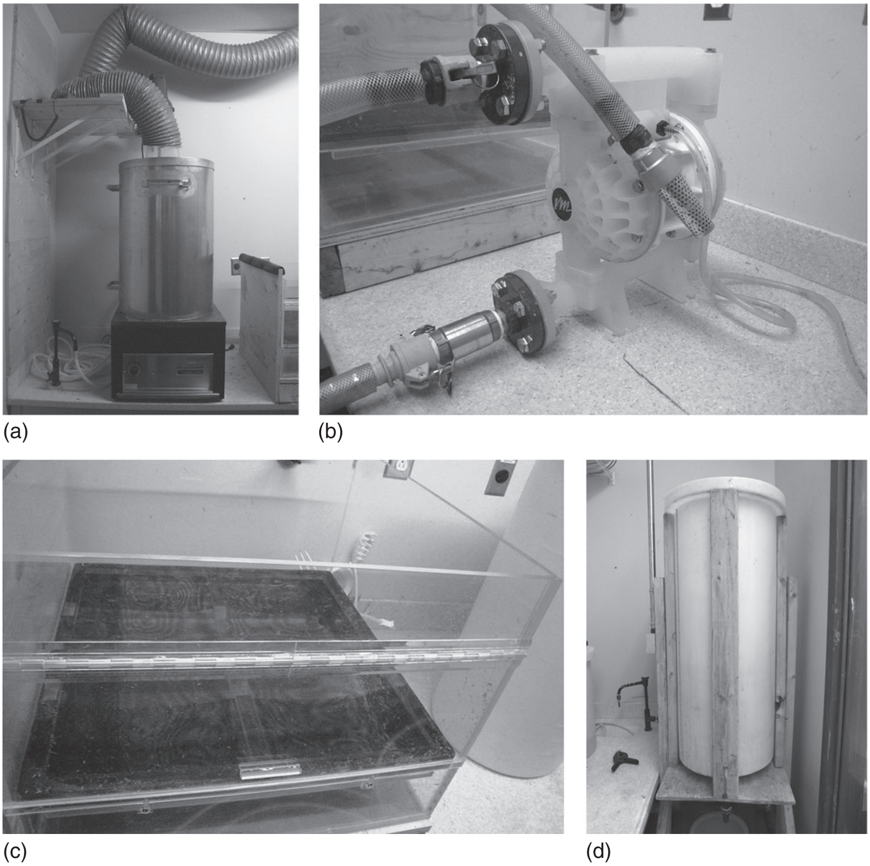
21.4.2 Remediation process at pilot scale

Following promising results obtained at laboratory scale for CCA-, ACQ-, CA- and MCQ-treated wood wastes recycling using a chemical leaching process, a pilot plant was designed to assess the feasibility of the process. The pilot plant allows simultaneous treatment of 12 kg of wood in a 130 L tank. The overall scheme of the process is shown in Fig. 21.4. A recent study explored the performances of this leaching process for removing metals from CCA-, ACQ-CA- and various mixtures of Cu-based treated wood and then from leachates by precipitation-coagulation at pilot plant scale (Janin *et al.*, 2012; Coudert *et al.*, 2012). CCA-samples (7) came from utility poles with a much greater CCA loading than residential timbers whereas ACQ-sample (1) and CA-sample (1) came from freshly treated timbers. For pilot-scale experiments, treated wood was chipped and sieved through a 12 mm sieve. Experiments were conducted in triplicate on CCA-, ACQ-, CA- and various mixtures of Cu-based preservative-treated wood to assess reproducibility of this leaching process at pilot scale.

The composition of each mixture was defined in order to represent wood waste stream evolution in the next decades. Leaching and rinsing steps were carried out in a 130 L-capacity tank made of 316-stainless steel (Fig. 21.5). Depending on wood preservative or initial metal content, acid concentration, leaching retention time and temperature should be adjusted (Table 21.2). After each leaching and rinsing step, liquid was removed using a pump and wood was then transferred to a Plexiglas drainer. The leaching remediation process produced effluents containing high concentrations of As, Cr and Cu. Leachates were treated by precipitation-coagulation on a 136-L capacity tank. A solution of ferric chloride (131 g Fe/L) was added to increase arsenic removal. pH was then adjusted to 7 by addition of a solution of sodium hydroxide using a peristaltic pump. In order to enhance solid/liquid separation, a flocculent (Magnafloc 10) was added under slow agitation.



21.4 Overview scheme of leaching process.



21.5 Pilot-scale equipment – leaching and rinsing tank with heating plate (a), solid to liquid separation system: pump (b) and drainer (c), settling tank for precipitation-coagulation (d).

An economic model was developed to determine costs related to CCA-, ACQ-CA- and mixtures of wood wastes recycling and leachate treatment by precipitation. The calculation of the direct operating and capital costs was completed for a 157 500 ttw (ton of treated wood) annual capacity treated plant (operating period: 350 days per year, 500 ttw per day). The estimation of direct operating costs was conducted on the basis of the following unit prices: \$80/t H_2SO_4 (solution at 93% w/w), \$500/t $FeCl_3$, \$500/t NaOH, \$25/t as average labor cost, \$0.05/kWh of electricity, \$0.05/m^3 of leaching and rinsing waters.

The capital costs were evaluated using a 15-yr reimbursement period and a 4% annual interest rate. The capital costs include all the equipment required for wood decontamination (acid leaching reactors, pumps, sieves, settlers, filter-press,

Table 21.2 Initial arsenic, chromium and copper contents on CCA-, ACQ-, CA- and mixture of copper-based treated wood and leaching conditions used for pilot scale experiments

Wood sample (preservative used)	Wood service time (yr)	Acid concentration (N)	Leaching time (h)	Temp (°C)	Initial metals content (mg/kg)		
					As	Cr	Cu
CCA 1991	20	0.5	2	75	7730	9320	5700
CCA 1996	14	0.3	2	75	4860	5430	3020
CCA 1999	10	0.4	2	75	5660	6360	3730
CCA 2005	–	0.2	2	75	5180	5590	3220
CCA 2008	1	0.2	2	75	5400	6480	3450
CCA 2009	1	0.2	2	75	5990	7310	4260
CCA 2010	1	0.4	2	75	6260	7340	4350
ACQ	0	0.13	2.7	20	–	–	1190
CA	0	0.13	2.7	20	–	–	1880
75% CCA, 12.5% ACQ, 12.5% CA	–	0.3	2	75	4320	5690	3670
50% CCA, 25% ACQ, 25% CA	–	0.2	2	75	2770	3730	2920
25% CCA, 37.5% ACQ, 37.5% CA	–	0.2	2	75	1660	2070	2280

dryer, etc.). The revenues were calculated considering an energy value of remediated wood burning for energy production of $13 per GJ.

Initial metal content of CCA-samples (Table 21.2) was very high. This was because CCA samples came from utility poles with a much greater CCA loading than residential timbers. For CCA 1991, high amounts of As, Cr and Cu were measured with respectively 7 g As/kg (dry wood), 9 g Cr/kg and 6 g Cu/kg. For other CCA-samples, the initial metal content ranged from 4 to 6 g As/kg, from 5 to 7 g Cr/kg and from 3 to 4 g Cu/kg. For ACQ- and CA-samples, copper initial amounts varied between 1 and 2 g/kg. Concerning mixtures of Cu-based preservative-treated wood, initial metal content varied between 2 and 4 g As/kg, 2 6 g Cr/kg, and 2 and 4 g Cu/kg. An increase of the percentage of CCA in the mixture led to an increase of initial metal content.

As, Cr and Cu content quickly decreased during the leaching process, and metal removal from treated wood wastes was highly efficient at pilot scale. Performances of this leaching process seemed to be slightly better than at laboratory scale, which can be explained by more efficient mixing system at pilot scale. Remaining metal content in remediated wood was very low, which allow remediated wood recycling by composting or energy production. For CCA-samples, remaining As content ranged from 26 to 60 mg/kg, Cr content ranged from 280 to 730 mg/kg and Cu content ranged from 15 to 100 mg/kg. Final metal content in the mixture of preservative-treated wood after remediation was lower than in CCA-remediated wood with 18 to 43 mg As/kg, 220 to 349 mg Cr/kg and 18 to 48 mg Cu/kg.

Concerning ACQ- and CA-samples, the remaining Cu content varied between 49 and 90 mg/kg, yielding up to more than 95% removal. At the end of the leaching process, whatever the wood preservative or initial metal content in wood wastes, more than 98% of As, 90% of Cr and 96% of Cu were removed. The leaching process produces effluents with high concentrations of As, Cr and Cu, which greatly exceed regulations for effluents discharge in sewer. Leachate treatment by precipitation allowed more than 97% removal of metals. The remaining As, Cr and Cu concentrations of the final effluents satisfied Quebec City municipal effluent discharge regulations.

A disadvantage of leachate treatment by precipitation was the production of highly concentrated metallic sludge (5–25 g As, Cr, Cu/kg of dry sludge), which should be dried and disposed of as hazardous materials. Chemicals consumption (sulfuric acid, ferric chloride, sodium hydroxide) and sludge disposal depends on wood preservative or mixture composition and initial metal content. For CCA-samples, costs were evaluated by taking into account the most representative sample (CCA 2005) of treated wood wastes timber entering wood wastes stream in the future. For CCA-, ACQ- and CA-samples, direct costs were estimated based on a treated wood wastes remediation process in counter-current mode with effluent recirculation.

In other words, leachates 2 and 3 were respectively reused in leaching 1 and 2 for the following loops of treated wood wastes remediation. Effluents coming out from leaching 1 were treated by precipitation and re-introduced in the leaching process in the third rinsing step. The leaching process can be successfully used in counter-current with effluent recirculation without affecting metal removal efficiencies. Counter-current helps reduce chemicals consumption and operational costs. Direct and indirect costs were estimated at $270/ttw for CCA-sample and $60/ttw for ACQ/CA samples. Among direct costs, the most expensive parts are related to leachate treatment by precipitation-coagulation and for CCA-sample to energy consumption for heating leaching solutions.

Revenues were estimated at $240/ttw, considering remediated sales for energy production. Net CCA-treated wood wastes recycling costs were estimated at $30/ttw, whereas ACQ- and CA-treated wood wastes recycling allowed a benefit of $120/ttw. This compares favorably to landfilling or burning costs ranging from $40 to $150/ttw.

21.5 Future trends

Treated wood wastes management is becoming a global and political challenge, due to increasing amounts of treated wood products reaching their end-of-life. At this time, landfilling and incineration of creosote-, PCP- and copper-based treated wood wastes remain the most common options worldwide for disposal of such solid wastes. However, regulatory changes, congestion of landfill sites and controversial public opinion could affect these disposal options in the future. Recycling of the treated wood wastes is becoming of high importance. The recycling options include toxic components solubilisation or degradation using physical, biological or chemical processes.

Several processes show good potential for treated wood wastes recycling and appear as promising solutions for sustainable management of the wood wastes in the future. As discussed below, considerable advancements have been made in the development of treated wood wastes recycling processes over the past few years, especially concerning electrodialysis treatment and chemical remediation processes. Electrodialysis proved to be efficient for preservative removal from the wood and various research teams obtained promising results. However, researchers have to considerably reduce duration of electrodialytic treatment that may cause high operational costs and could possibly be an issue for future development in the industry.

However, the chemical approach is really promising too. A leaching process with sulfuric acid was developed for CCA-treated wood waste and then modified for other copper-based preservatives. Economic analysis and pilot-scale experiments of the chemical leaching processes look promising for development at industrial scale in the near future.

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22

An effective approach to utilize recycled aggregates (RAs) from alkali-silica reaction (ASR) affected Portland cement concrete

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Abstract: This chapter presents the use of recycled aggregate (RA) from alkali-silica reaction (ASR) affected Portland cement concrete (PCC) in hot mix asphalt (HMA), which promote sustainability. This study was performed to determine the effects that existing ASR gel, reactive aggregates and micro-cracks might have on the performance of HMA made using ASR affected recycled concrete aggregate (ASR-RCA). Guidelines were developed for the use of ASR-RCA in HMA, with special emphasis on potential remedial measures to prevent any new ASR in HMA. The test results have indicated that ASR-RCA can safely be used in making HMA airfield pavements, with no threat of expansion due to new ASR.

Key words: alkali-silica reaction (ASR), Portland cement concrete (PCC), recycled concrete aggregate (RAC), alkali silica gel, micro-cracking, hot mix asphalt (HMA), expansion, moisture damage, sustainability.

22.1 Introduction

Alkali-silica reaction (ASR), a reaction between siliceous aggregates and alkali-hydroxides in concrete pore solution, is one of the main chemical distress problems that may occur within Portland cement concrete (PCC). The three criteria required to initiate expansive ASR are:

1. presence of reactive siliceous component(s) in aggregate;
2. liquid medium with sufficient pH (>13.2: Tang and Fen, 1980) and alkalis; and
3. sufficient moisture (≥85%: Chatterjii *et al.*, 1989; Chatterji, 2005).

If any of these criteria are not met, then ASR will not occur. The reaction between alkaline pore solution (OH^-, Na^+ and K^+ ions) and reactive silica in the aggregates produce a reaction product called alkali-silica gel. If this gel absorbs sufficient moisture, it expands, creating internal pressures in the concrete, which can exceed the tensile strength and crack the concrete. If the damage caused by ASR and other associated distress mechanisms is significant, removal or replacement of the concrete is inevitable. Disposal of ASR affected concrete can be problematic, due to decreasing landfill space and increased focus on re-use of materials to promote sustainability. One way to re-use these concretes is to crush them and make recycled concrete aggregate (RCA).

RCA has been used successfully as a base material, as well as making PCC, but exposure to moisture would be undesirable for an ASR-affected recycled concrete aggregate (ASR-RCA). Also, ASR-RCA used in concrete would not only expose it to moisture, but there would likely be fresh faces of the reactive aggregates created during crushing that could facilitate additional ASR to occur. Since the expansion of ASR is a result of the gel absorbing water, it is desirable to re-use ASR-RCA in a manner that will minimize its exposure to moisture. The occurrence of new ASR has been reported in previous studies on using ASR-RCA in PCC (Li and Gress, 2006). As an alternative, the ASR-RCA could be used as an aggregate in hot mix asphalt (HMA) safely and effectively, as moisture is not available in HMA normally. While studies have been performed on the use of RCA in HMA, no studies have been performed on the use of ASR-RCA in HMA.

Studies of regular RCA have indicated that RCA is rougher, more angular and more absorptive than conventional aggregates (Kuo *et al.*, 2002; Saeed *et al.*, 2006; Gonzalez and Young, 2004; Chini *et al.*, 2001; Cuttell *et al.*, 1997; Obla *et al.*, 2007; Topcu, 1997; Topcu and Sengel, 2004; Heins, 1986; Paranavithana and Mohajerani, 2006; Wong *et al.*, 2007). These properties have been attributed to increased water demand in PCC made using RCA (Gonzalez and Young, 2004; Chini *et al.*, 2001; Cuttell *et al.*, 1997; Obla *et al.*, 2007; Topcu, 1997; Topcu and Sengel, 2004).

Several states allow the use of RCA into HMA. Several studies have been performed to determine the properties and performance of HMA containing RCA. These studies have found that, in comparison to mixes made with virgin aggregates, HMA made with RCA will either have a higher air void content for the same asphalt binder content or a higher optimum binder content for the same air void content (Anon, 1984; Azari *et al.*, 2006; Heins, 1986; Paranavithana and Mohajerani, 2006; Wong *et al.*, 2007). This has been attributed to the increased porosity of the cement paste attached to the aggregate, similar to that found in the studies of RCA in PCC. The more absorptive RCA particles usually absorb more asphalt into the larger pores and absorb asphalt more deeply into the particle. Therefore, design asphalt contents, when using RCA, can be expected to be relatively high. RCA lowers abrasion resistance and bond strength of HMA made of RCA more than HMA made of virgin granite aggregate (Wong *et al.*, 2007).

The studies show differing results for the stripping potential of HMA made with RCA. Paranavithana and Mohajerani (2006) found that HMA mixes made with RCA had a greater potential for stripping, while Heins (1986) found that HMA mixes made with RCA had stripping potential similar to mixes made with virgin aggregates. Paranavithana and Mohajerani (2006) also found that HMA mixes made with RCA had significantly lower strength than those made with virgin aggregates. However, Wong *et al.* (2007) found that the use of RCA fines in the mix improved the rut resistance of the HMA mixture. Some studies have found that the RCA particles have a greater tendency to break down during mixing and compacting of the HMA (Paranavithana and Mohajerani, 2006).

The objective of this study was to provide guidance on the use of ASR-RCA in HMA airfield pavements, specifically:

- To determine the effects that existing ASR gel, reactive aggregates and micro-cracks might have on the performance of HMA made using ASR-RCA;
- Determine potential remedial measures to prevent any new ASR and mitigate the damage present in ASR-RCA;
- Develop guidelines for the use of ASR-RCA in HMA with proper consideration of long-term effects of some airfield conditions on pavement performance. These conditions include climate (rainfall, temperature, etc.), traffic loading and de-icer applications.

22.2 Scope of the study

The study was conducted in the following sequence in order to achieve the above objectives:

- Testing both virgin aggregate and ASR-RCAs. Testing virgin aggregate served as a control as well as providing basic understanding of the chemical reaction (if any) mechanisms. However, testing ASR-RCA ensured studying the effects of actual field ASR-RCA specimens on continued ASR in HMA.
- Identify the potential possible mechanisms for continued ASR distress (if any) within HMA; establishing a connection between moisture damage and any new ASR in HMA was the main item to address this task.
- Identify laboratory tests to properly characterize ASR-RCA and investigate the potential for continued ASR within the HMA mixture:
  - **Aggregate testing**: Testing was conducted to determine the basic properties of each aggregate source typically determined for a HMA aggregate. These tests included gradation, specific gravity, absorption, abrasion resistance and durability tests.
  - **Compacted HMA testing**: In addition, testing was performed on HMA specimens made with each aggregate to determine ASR potential and moisture damage. A detailed micro-structural study was also performed to verify the occurrence and nature of ASR gel formed on the virgin aggregate during testing. Lastly, an artificial ASR gel was observed to determine the effects that heating the aggregates in the HMA plant will have on any ASR gel present in the RA.
- Projection of field performance based on accelerated laboratory performance. An attempt was made to project the performance of HMA under airfield

conditions based on lab results. The potential differences between lab testing and actual long-term field performance was evaluated. However, actual field performance evaluation through measurement of field parameters (e.g. level of alkalinity and moisture availability) and test section construction is beyond the scope of the present research.

- Possible remediation measures were recommended and guidelines were developed on the use of ASR-RCA HMA airfield pavements.

22.3 Materials and test methods

Two ASR-RCAs were prepared from removed ASR-affected PCC slabs of two airfield pavements. The ASR-RCAs were characterized by stereo light, polarized light and scanning electron microscopy to confirm the presence of ASR. Testing to determine the potential for re-expansion of existing gel or expansion due to new ASR was performed using the dilatometer device developed at the Texas Transportation Institute. The dilatometer is a device that simulates the reaction between aggregate – pore solution that exists in PCC and measures the solution volume change over time as the ASR proceeds (Mukhopadhyay, 2006; Sarkar, 2004; Shon, 2007). The dilatometer consists of a stainless steel container, a brass lid, a stainless steel tower, a stainless steel float, a cap, a linear variable displacement transducer (LVDT) and a thermocouple.

The LVDTs and thermocouples are connected to a computer data acquisition system. The computer automatically measures the displacement and temperature at the specified time interval. Both ASR-RCAs were tested at 60 °C using 0.5N NaOH + CH as the primary test condition for this study. Additional expansion testing was performed using a modified beam test (based on ASTM C 1293). The petrographic examination (ASTM C 295) of the test samples and analysis of the test solution were used as supporting tools to verify the dilatometer results. Moisture susceptibility of the HMA was performed using the Lottman test (AASHTO T 283, ASTM D 4867) and a micro-calorimeter. The Lottman test is intended to indicate the moisture sensitivity of an asphalt mixture. It consists of comparing the indirect tensile strength of specimens conditioned in water at 60 °C for 24 h to the strength of dry specimens. Higher ratios of indirect tensile strength indicate a lower susceptibility to moisture damage, with most agencies requiring a minimum ratio of 70%.

The micro-calorimeter measures the total energy of adhesion (TEA) between an aggregate and an asphalt binder through the heat produced when the two materials are mixed. An empty reference cell and a cell containing the aggregate are allowed to reach thermal equilibrium, and then a solution of asphalt binder in toluene is injected into each cell. The heat flow between the cells is recorded until it returns to equilibrium and the software integrates the area under the heat flow curve to determine the TEA. Micro-calorimeter tests were performed on samples between 150 and 75 μm (No. 100 to No. 200).

Differential scanning calorimeter (DSC) testing and thermogravimetric analysis (TGA) were used to determine the impact of heating on artificial ASR gel. Micro-deval (AASHTO T 327, ASTM D 6928/ASTM D 7428) and freeze-thaw testing (AASHTO T 103) were used to evaluate the suitability of the RCAs for use in HMA. The micro-deval test measures the mass loss of an aggregate sample abraded by steel balls in the presence of water inside a steel drum. It is anticipated that ASR-RCA will show greater micro-deval mass loss than the virgin unreacted aggregate. The higher loss can be correlated with poor mechanical performance (Meininger, 2004; Rogers, 1991). The freeze-thaw test (AASHTO T 103) measures the mass loss of aggregate specimens subjected to a specified number of freezing and thawing cycles. It is expected that ASR-RCAs will have greater mass loss due to higher porosity and lower strength of these RA.

A dense-graded HMA meeting the gradation requirements of P 401 was used for dilatometer and Lottman tests. The asphalt binder used in the study was a PG76-22 produced by Valero. The mixing temperature of the binder was 163 °C (325 °F) and the compaction temperature was 149 °C (300 °F). Samples of compacted cylindrical HMA specimens (4 inch dia. and 5.5 inch ht.) were produced using the Superpave gyratory compactor. Two different binder contents (i.e. 6 and 8%), representing thick and thin films of asphalt binder, were selected for each material. The SMA gradation with a binder content of 6.5% was used to construct the beams for the modified beam test.

22.4 Results and discussion

The results of aggregate durability testing of the ASR-RCAs, and moisture susceptibility as well as ASR testing of HMA made using the ASR-RCAs, are presented in the following subsections. In addition, a discussion on how these results influence the use of ASR-RCA in HMA pavements is included.

22.4.1 Recycled concrete aggregate (ASR-RCA) durability testing

The results from the micro-deval and freeze-thaw tests for both ASR-RCAs and virgin aggregate are presented in Table 22.1. The results indicate that the RCAs have higher mass losses than chert in both the micro-deval and freeze-thaw tests, as expected. This is likely due to the presence of ASR micro-cracks and softer adhered mortar in the RCAs, which facilitated greater breakdown in RCAs than the virgin aggregate. The high micro-deval loss (13%) in virgin aggregate fines was due to the presence of limestone particles. The ASR-RCA-1 is made of limestone as coarse aggregate, which was responsible for high micro-deval loss in its coarse fraction (15%). The coarse aggregate in RCA-2 was primarily granitic.

It is anticipated that the degree of preferential accumulation of mortar particles and ASR gel primarily determines the percentage loss in fines, whereas the degree

Table 22.1 Micro-deval and freeze-thaw test results

		Chert		ASR-RCA-1		ASR-RCA-2	
	Size fraction	Coarse (12.5–4.75 mm)	Fine (4.75–0.075 mm)	Coarse (12.5–4.75 mm)	Fine (4.75–0.075 mm)	Coarse (12.5–4.75 mm)	Fine (4.75–0.075 mm)
Micro-deval	Average mass						
	Loss (%)	5%	13%	15%	13%	13%	16%
	COV (%)	2.0%	5.6%	0.1%	0.2%	0.9%	9.2%
	Size fraction	Coarse (19–4.75 mm)	Fine (4.75–0.300 mm)	Coarse (19–4.75 mm)	Fine (4.75–0.300 mm)	Coarse (19–4.75 mm)	Fine (4.75–0.300 mm)
Freeze-thaw	Average mass						
	Loss (%)	0.7%	4.5%	2.8%	8.0%	1.1%	7.0%
	COV (%)	0.2%	5.5%	59.1%	4.4%	0.3%	3.3%

of micro-cracking determines the percentage loss in coarse fraction in ASR-RCA. As a result, the difference in loss (%) between coarse and fine fractions of the two RCAs is not significant. This is an indication that the micro-deval test of the fine aggregate cannot be considered as a diagnostic tool to select or reject the ASR-RCA fines in HMA. However, if the loss in the fine fraction is significantly higher than that in coarse fraction, then it is advisable not to use the fine fraction in HMA. Micro-deval and freeze-thaw testing do not appear able to detect the effect of gel in an ASR-RCA but do appear capable of detecting the effect of pre-existing micro-cracking due to ASR. The freeze-thaw test seems to be more sensitive to detecting the effect of pre-existing micro-cracking due to ASR than micro-deval. Both micro-deval and freeze-thaw tests may still be used as quality indicators for coarse and fine aggregate, using the same guidelines as would be used for a typical RCA.

22.4.2 Hot mix asphalt (HMA) moisture damage testing

The results of the moisture damage testing are presented in Table 22.2. Specimens were conditioned in distilled water at 60 °C for 24 h (representative of Lottman test conditions) as well as in NaOH solution at 60 °C for both 24 h and 6.5 days (representative of dilatometer test conditions). It is reported that TEA is proportional to the strength of the chemical bond between the aggregate and the asphalt binder (Vasconcelos *et al.*, 2008).

Table 22.2 Moisture damage test results

	ASR-RCA-1	ASR-RCA-2	Chert
TSR (24 h water to dry)	96%	100%	80%
TSR_{ND}(24 h NaOH to dry)	101%	104%	82%
TSR_{NW}(24 h NaOH to water)	106%	104%	103%
$TSR_{6.5}$(6.5 day water to dry)	95%	85%	*
$TSR_{ND6.5}$(6.5 day NaOH to dry)	91%	76%	*
$TSR_{NW6.5}$(6.5 day NaOH to water)	95%	85%	*
TEA (mJ/g)	680	425	295

Notes:

* Specimens were too soft to test; TSR – tensile strength ratio of specimens conditioned in water to specimens left dry at 60°C for 24 hours, as required by Lottman test; TSR_{ND} – tensile strength ratio of specimens conditioned in 0.5N NaOH solution to specimens left dry at 60°C for either 24 hours or 6.5 days; TSR_{NW} – tensile strength ratio of specimens conditioned in 0.5N NaOH solution to specimens conditioned in water at 60°C for either 24 hours or 6.5 days TEA – total energy of adhesion between the aggregate and asphalt binder determined by Microcalorimeter test.

None of the specimens after the 24 h testing period showed damage index below the 70% TSR limit (24 h water to dry) prescribed by most agencies that use the Lottman test. It is noteworthy that a relatively high binder grade was selected for this study to reduce stripping in the HMA specimens made with chert during dilatometer testing (described later). With a softer binder, the chert would have showed below 70% TSR. The results of the micro-calorimeter testing support the results of the Lottman testing. The micro-calorimeter results predict that the chert should have the greatest moisture damage (lowest TEA), which was confirmed by the lowest TSR values for chert in the Lottman test. It is interesting to note that the chert specimens failed through the binder and binder-aggregate interface (supportive of lowest TEA) in the Lottman test. At 6.5 days, ASR-RCA-2 had lower TSR than ASR-RCA-1, which agrees with the lower TEA for ASR-RCA-2. ASR-RCA-1, which contains limestone coarse aggregate (non-reactive), is more moisture resistant than ASR-RCA-2, which contains granitic coarse aggregate (reactive). The fine aggregate is reactive in ASR-RCA-1. This would indicate that ASR-RCA containing reactive siliceous coarse aggregate (e.g. ASR-RCA-2) is more likely to be moisture sensitive, since HMA mixtures containing siliceous aggregates tend to have greater moisture susceptibility (Bhason *et al.*, 2007; Birgisson *et al.*, 2007). The failure planes in the Lottman specimens of ASR-RCA pass through the RCA (particularly through the mortar portion), which is likely due to the presence of micro-cracks in RCA (indicated by the freeze-thaw results earlier) and high TEA values.

22.4.3 Alkali-silica reaction (ASR) testing

Dilatometer testing was successfully used to characterize the reactivity of the ASR-RCAs alone and HMA concrete specimens made using ASR-RCAs. The dilatometer tests were mostly conducted at 60 °C and 0.5N NaOH + $Ca(OH)_2$ solution. Additional testing of ASR-RCA-2 was performed using a potassium acetate de-icer solution.

The occurrence of ASR was evident through volume expansion measurement over time from RCA-solution tests. Volume changes over time in compacted HMA specimens containing RCAs appear to be primarily due to an interaction between the asphalt binder and the alkaline solution. The technical term for the binder-solution interaction is saponification (i.e. neutralization by the OH^- ions) of naphthenic acids in the asphalt binder. The standard test to quantify these acids for petroleum products involves neutralization with potassium hydroxide (30–36), which is similar to saponification described above. Therefore, saponification is the reasonable explanation for binder-solution interaction. In support of the occurrence of the binder-solution interaction, the compacted HMA specimens were observed to feel significantly softer after testing and had a tendency to crack during handling. Microstructure and test solution chemistry strongly support:

- the presence of ASR in RCA – solution tests (without binder); and
- mainly binder-solution interaction in all the compacted HMA tests.

However, some ASR was manifested along with the binder-solution interaction, when HMA containing one RCA was tested with concentrated de-icer solution (potassium acetate solution from Cryotech, Inc.).

The possible reaction sequences between HMA made with ASR-RCA and de-icers are as follows:

- OH^- from the mortar fraction of the ASR-RCA diffuses through the asphalt binder into the solution to increase its pH. This initial process occurs slowly.
- The OH^- in the solution reacts with the naphthenic acids in the asphalt binder, causing binder degradation and accelerating moisture damage. The degradation and moisture damage lead to softening of the binder and stripping of the binder from the aggregate surface.
- The exposed mortar increases the rate at which OH^- enters the solution. The OH^- in solution reacts with both the asphalt binder (binder-solution interaction) and the exposed surfaces of reactive aggregates (new ASR).

The binder-solution interaction appears to occur at a greater rate than ASR; however, the combined effects of both reactions appear to be less than that of the mortar's release of OH^- into the solution. The compacted HMA specimens do not have such large decreases in K^+ as ASR-RCA-2 alone, indicating that if any ASR did occur in the HMA tests in de-icer solution, it was not as extensive as with ASR-RCA-2 alone (i.e. asphalt film offered some protection).

22.4.4 Modified beam testing

The modified beam test was developed to measure linear ASR expansion (if any) in HMA concrete. A HMA concrete beam using one of the two RCAs did not show any signs of expansion. However, results of the modified beam test provide some support of the binder-solution interaction. No increase in length of the beams was apparent from the beam tests until the 3-month testing period. Since the volume changes measured in the HMA dilatometer testing (Section 22.4.3) are believed primarily to be a result of binder-solution interaction and not ASR, the lack of change in the beam lengths is consistent. It seems binder-solution interaction was not capable of producing any measurable length change in beam test due to:

- insufficient swelling from binder-solution interaction; and/or
- incompressible nature of asphalt binder.

Since the dilatometer measures volume changes in the whole system, it is expected to be more sensitive than the beam test. From the limited test results, it is expected that any volume change due to binder-solution interaction may not be manifested as any measurable total solid volume change in the field.

22.4.5 Differential scanning calorimeter (DSC) testing on artificial gel

An artificial gel of similar composition to those found in concretes suffering from ASR was tested in the DSC-TGA. Dehydration of the gel was observed to occur at temperature ranges of 100 to 150 °C (i.e. 212–302 °F). Since HMA is typically produced at temperatures above this, normal drying equipment should be capable of drying the gel. However, extended drying (i.e. in a batch plant) is preferable, to ensure that the gel is dried completely.

22.5 Field implications

Although mild, the conditions (0.5N NaOH + $Ca(OH)_2$ and 60 °C) used to accelerate the test process in the laboratory are still more severe than HMA made with ASR-RCA is likely to experience in the field. The possible distress mechanisms that might occur in HMA made using ASR-RCAs in the field are:

- mechanical degradation due to pre-existing ASR micro-cracks;
- rehydration and swelling of dried gels, if sufficient moisture is available through damage (moisture normally not available in HMA);
- weak bond between aggregate-binder due to presence of gel on aggregate surface;
- generation of alkaline solution due to interaction between mortar fractions and moisture – quantity and strength of alkaline solution depends on the degree of moisture availability and de-icer penetration (whenever applies);
- binder degradation through asphalt binder and alkaline solution interaction; and
- new ASR, provided alkaline solution of sufficient strength and quantity is still available to the fresh aggregate surfaces. It is anticipated that solution may still be available at the reactive faces but the pH will not be high enough (~11.5) to cause any measurable new ASR.

Although de-icers can make the solution pH relatively high (~12.0), it is still not favorable to create sufficient ASR that can cause any measurable distress.

Solid volume increase, due to binder-solution interaction (swelling) and expansion of new ASR gel (if any) or re-hydration (expansive) of existing gel, will either not be manifested at all or will take longer than the service life of HMA pavements made using ASR-RCA. However, binder-solution interaction and any ASR may not be a primary cause of failure, but micro-crack formation due to ASR can create a favourable situation for other distress (especially moisture damage).

Although the dilatometer testing indicated that ASR might still occur in HMA made with ASR-RCA exposed to de-icer solution at its original concentration, the dilatometer conditions can still be considered more severe than in the field. In the dilatometer, the compacted HMA specimen was completely surrounded by a large quantity of solution that maintains a constant supply of de-icer, but in the field the de-icer would only be applied at the surface. Therefore, only de-icer that penetrated the HMA pavement through cracks or diffusion would be available for ASR. In addition, the de-icer would be subject to washing away or dilution by rain or melted snow and ice.

22.6 Recommendations

The test results in the current study indicate that ASR-RCA can successfully be used in making HMA pavements. The occurrence of any new widespread ASR in HMA made using ASR-RCAs is a remote possibility. The guidelines for the use of ASR-RCA in HMA have been developed, which are summarized below:

- Any restrictions placed on the use of conventional RCA should also be applied for ASR-RCA.
- It is advisable to store the crushed ASR-RCA outdoors for as long as possible, to increase the carbonation of the gel and the mortar fraction. Increased carbonation of the gel will reduce its potential for future expansion through re-hydration. Increased carbonation of the mortar fractions in RCA will reduce the pH of alkaline solution that may generate through moisture-cement mortar interactions.
- An effective drying at high temperature in the HMA plant is necessary to ensure effective dehydration of gel, which will minimize the availability of additional moisture inside HMA pavements. It would be preferable to produce the HMA in a Batch Plant (if available), as it facilitates long effective drying.
- It is recommended not to use the fines and fillers from ASR-RCAs, as ASR gel may preferentially accumulate in the finer fractions during crushing and sieving as demonstrated by XRD analysis. The use of a good-quality fine aggregate is highly recommended when ASR-RCA is used as coarse aggregate, which will reduce the necessary binder content of the mix, thereby reducing its cost.
- Efforts need to be made to reduce interconnected voids in HMA mixtures and therefore the ability of moisture to infiltrate the HMA.
- Conservative moisture damage criteria based on moisture damage testing can be introduced to reduce the potential for moisture related problems.
- Although a moisture-damage resistant design is desirable, engineers may choose to restrict the use of anti-stripping agents in HMA made with ASR-RCA, as they may provide additional OH^- for reactions. The limited results of the Lottman and micro-calorimeter tests from this study indicate good bonding

between the ASR-RCA and the asphalt binder, negating the need for anti-stripping agents.
- Attention should also be paid to the reaction products and micro-cracks from other distresses, such as delayed ettringite formation (DEF) and alkali-carbonate reaction (ACR) that may present along with ASR in the RCA.
- It is advisable not to use HMA made from ASR-RCA in the portions of airfield pavements prone to receive greater impact due to aircraft landing (e.g. runway) or high traffic frequency. However, placing a quality surface course made using virgin aggregate on top of HMA made using ASR-RCA will probably minimize this issue.
- It is recommended to determine the level of ASR distress and the reactivity of the aggregate before making the RCA from the ASR-affected PCC pavement. This will assist in determining the proper utilization of ASR-RCA in HMA pavements, depending on climate, traffic and de-icer applications, as well as selecting the remedial measures.

22.7 Acknowledgements

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23
Life-cycle assessment (LCA) of concrete with recycled aggregates (RAs)

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Abstract: This chapter focuses on the life-cycle assessment (LCA) of aggregates obtained by recycling of demolished concrete – recycled concrete aggregates (RCA), and concrete made with such aggregates – recycled aggregate concrete (RAC). It includes methodological aspects, such as the treatment of allocation in the case of concrete recycling. The results of LCA case studies on two different RCA applications – as aggregate in structural RAC concrete and as material for road base, are presented. The potentials and limitations of LCA in comparing different waste management scenarios are discussed based on the published research. Recommendations regarding future research are given.

Key words: demolished concrete, recycled concrete aggregate, recycled aggregate concrete (RAC), life-cycle assessment (LCA), allocation, waste management systems.

23.1 Introduction

Over the few past decades, the development of energy and resource efficient technologies and products has become a primary goal in sustainable development. The construction industry is no exception to this rule. It is responsible for 50% of the consumption of natural raw materials, 40% of total energy consumption and almost half of the total industrial waste generated worldwide (Oikonomou, 2005). Within the construction industry, concrete production is regarded as having the most significant environmental, cost and social impacts. The volume of concrete produced and the number of concrete structures built forms a significant part of the global problem of sustainable development. Concrete is the most widely used building material, and its production and utilization is constantly increasing. World concrete production is currently about 6 billion tons per year, that is, 1 ton per person per year (ISO/TC 71, 2005). The level of harmful impact represented by a unit of concrete is, in comparison with other building materials, relatively small. However, due to the high production of concrete, the overall negative environmental impact of concrete structures is significant: large consumption of natural resources (aggregates for cement and concrete and energy) and a large amount of produced construction and demolition (C&D) waste.

Recycling represents one way to convert a waste product into a resource. It has the potential to reduce the amount of waste disposed of in landfills, to preserve natural resources, and to provide energy and cost savings while limiting environmental damage. Consequently, the use of recycled concrete aggregates (RCA) from waste concrete in new buildings could be of crucial importance in achieving sustainable construction. RCA are generally produced by crushing, screening and removing contaminants such as reinforcement, paper, wood, plastics and gypsum (by magnetic separation, water cleaning or air-sifting). Primary and secondary crushing is performed using a combination of compressive-type and impact-type crushers (e.g. jaw crusher and impact crusher, respectively).

When demolished concrete is crushed, a certain amount of mortar and cement paste from the original concrete remains attached to stone particles in recycled aggregate. Because of this attached mortar, recycled aggregates, compared to natural aggregates (NA), have (Marinkovic *et al.*, 2012):

- up to 10% lower density;
- higher porosity and higher water absorption – for coarse RCA, water absorption ranges from 2.2 to 9% and for fine RCA from 5.5 to 13%;
- up to 70% higher Los Angeles abrasion loss.

These properties of recycled aggregates also affect the properties of concrete.

RCA are commonly used in lower-quality product applications such as back-fills and road sub-base and base, where they currently compare favourably with NA in many local markets. However, only a small amount of RCA is used today in higher-quality product applications, such as structural concrete. Although in many countries standards do allow the utilization of RCA in structural concrete, actual application remains limited to less than 1% of the amount of aggregates used in structural concrete (Marinkovic *et al.*, 2012). The potential of demolished concrete recycling to decrease the environmental burdens of concrete can be fully exploited only if RCA replace NA in structural concrete, since this is by far the largest application of aggregates.

Other types of demolished material that can be used as aggregate in concrete include recycled glass or brick. The replacement of NA with recycled glass for replacement ratios higher than 30% leads to a significant deterioration of the basic physical and mechanical properties of concrete, besides concerns about alkali-silica reactions (Topçu and Canbaz, 2004). If ground to a fine powder, recycled glass can be better utilized as a partial substitution for cement, instead of silica fume and flying ash (Shayan and Xu, 2004). Considering the inherent properties of bricks (low density and high porosity), it is obvious that utilization of recycled bricks as aggregate also leads to a significant deterioration of concrete properties (Schulz and Hendricks, 1992). Since higher replacement of NA with such recycled waste is not recommended in structural concrete, this chapter focuses on RCA.

This chapter presents the application of life-cycle assessment (LCA) methodology in the environmental assessment of recycled aggregate concrete (RAC) for structural use. First, the basic properties of this construction material are described.

23.2 Properties of concrete with recycled concrete aggregates (RCA)

Concrete made with recycled concrete aggregate instead of NA is called recycled aggregate concrete (RAC). The replacement of NA with RCA can be total (100%) or partial (<100%). Standard methods used for the mix design of NA concrete (NAC) can also be used for the design of RAC mixes. However, RCA has high water absorption, which affects the workability of the RAC mix. To obtain the desired workability of RAC, it is necessary to add a certain amount of water to saturate recycled aggregate before or during mixing. The quantity of additional water can be reduced if water-reducing admixtures are used to achieve the desired workability.

The available test results on the properties of concrete made with recycled concrete aggregate vary widely and are sometimes contradictory. Based on the analysis of published experimental evidence, it can be concluded that concrete made with recycled coarse aggregate (100% replacement ratio), as compared to NAC with the same water-to-cement (w/c) ratio, has the following properties (Marinkovic *et al.*, 2012):

- decreased compressive strength up to 25%;
- decreased splitting and flexural tensile strength up to 10%;
- decreased modulus of elasticity up to 45%;
- increased drying shrinkage up to 70%;
- increased creep up to 50%;
- increased water absorption up to 50%;
- similar depth of carbonation;
- same or decreased freezing and thawing resistance;
- same or slightly increased chloride penetration.

The values given here are the upper bounds of all the analysed research data, which vary greatly because of the varied quality of the RCA used. This does not mean that it is impossible to obtain RAC with similar properties to NAC (Section 23.2.3). For this to be accomplished, it is necessary to use good-quality RCA and to make small adjustments in the w/c ratio in the mix design. Good-quality RCA should be recycled aggregate obtained from clean concrete waste and with water absorption of up to 5 to 6%. The properties of RAC containing both fine and coarse recycled aggregate are inferior to those of RAC containing only coarse recycled aggregate. Fine RCA has very high water absorption values (up to 13%) and high cohesion, which makes quality control of the concrete very difficult. Some standards and specifications (DAfStb, 2004; BSI, 2006) therefore do not recommend utilization of fine recycled aggregate in RAC for structural use.

European standards for aggregates in concrete EN 12620 (CEN, 2002) and concrete EN 206–1 (CEN, 2000) allow the use of recycled aggregates defined as 'aggregates resulting from the processing of inorganic material previously used in construction'. They can be applied in concrete regardless of their size, type and amount, as long as they comply with these standard requirements for aggregates and concrete. However, some national standards, complementary to European norms, specifically cover recycled aggregates for concrete.

For instance, British standard BS 8500–2 (BSI, 2006) specifies that a maximum of 20% of the natural coarse aggregate can automatically be replaced by RCA conforming to the requirements. Only the use of coarse RCA is recommended for application in new concrete construction. RCA can also be used in concretes of strength classes up to C 40/50 and in most exposure classes, except exposure to salt, severe freeze-thaw or severe aggressive ground. Another example is the 'Guidelines for Recycled Aggregate Concrete' published by the German Committee for Reinforced Concrete (DAfStb) in 2004. Complementing DIN EN 206–1 and DIN 4226–100 (German standards for recycled aggregates), these guidelines state that the permitted replacement ratio of coarse natural with coarse recycled aggregate in concretes of strength classes up to C 30/37 varies from 25% to 45%, depending on the recycled aggregate type and exposure class. Only recycled aggregates with grain size larger than 2 mm are allowed in new concrete.

23.3 Life-cycle assessment (LCA) of concrete: allocation issues

As stated above, concrete has a large impact on the environment because of the enormous scale of its production and utilization. That is why the environmental assessment of concrete is of great importance with regard to efforts to work towards a sustainable society. Recycling demolished concrete is not an 'environmental friendly' option by default. The impacts from RCA production and from the production of concrete with such an aggregate must be evaluated as for any other product or service. Today, the environmental impacts of products and processes are assessed for their whole life cycle: raw materials acquisition, material production and construction, use phase and end-of-life phase.

Until now, many different methodologies for environmental assessment have been developed, but the most acknowledged (and standardized) is life-cycle assessment (LCA). According to ISO standards (ISO, 2006), LCA consists of four steps: defining goal and scope, creating the life cycle inventory (LCI), assessing the environmental impacts (LCIA) and interpreting the results. For detailed information on LCA with regard to concrete see FIB TG3.6 (2008), Hájek *et al.*

(2011), Josa *et al.* (2005), Marinkovic (2013), Purnell and Black (2012) and Van de Heede and De Belie (2012). Only the specific features of LCA application in the environmental assessment of RCA and RAC are discussed here, and the most important among them is allocation.

When dealing with multi-functional processes in LCA, some method of allocation must be applied. In the ISO standards (ISO, 2006), allocation is defined as the partitioning of the input and/or output flows of a process in the product system being studied. Recycling is a multi-functional process: for the product being recycled, it is a waste management service; for the product receiving the recycled material, it forms part of the raw material production process. In these situations, the question is how to allocate the environmental impacts of recycling. ISO 14041 (ISO, 2006) describes a three-step procedure. First, unnecessary allocation should be avoided by dividing the process into sub-processes or by expanding the system boundaries to include all the products involved. This is usually the most labour-intensive step and is rarely applied. Second, unavoidable allocation must be done in a way that reflects an underlying, causal, physical relationship. The third step is about 'other relationships', such as market value (economic allocation).

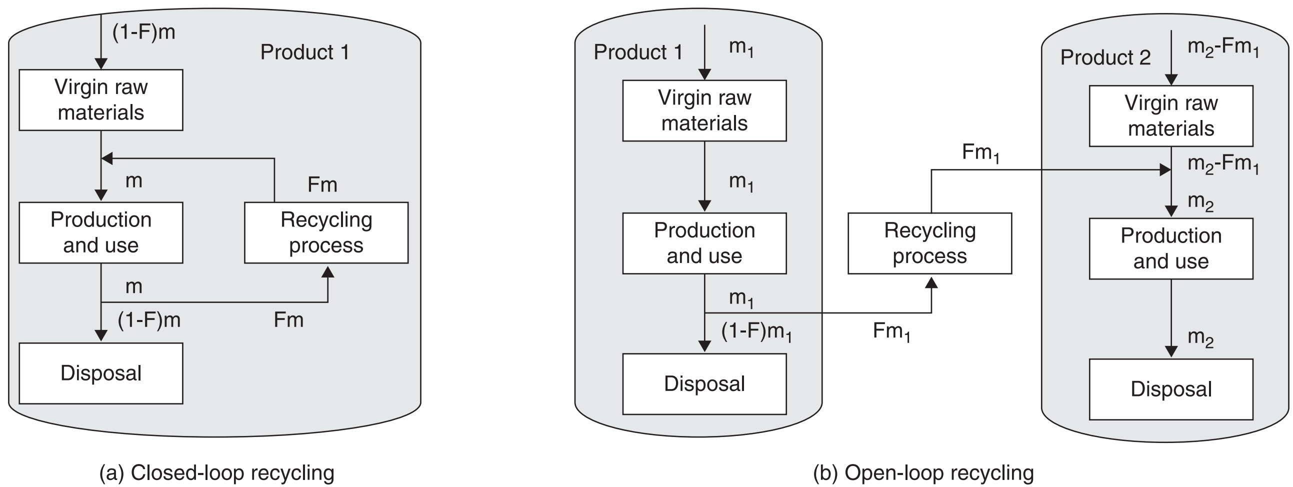
According to ISO (2006), the changes in the inherent properties of materials should be taken into consideration within an allocation procedure. There are two possible distinctive situations:

1. The material's inherent properties are not changed through the product system and the material is to be used in the same application;
2. The material's inherent properties are changed through the product system and the material is to be used in other applications.

In the first case, recycling is considered as a closed-loop system. Closed-loop recycling occurs when a product is recycled into a product that can be recycled over and over again, theoretically endlessly (Fig. 23.1a) (Vigon *et al.*, 1993). In the second case, recycling is considered as an open-loop system. Basic open-loop recycling occurs when a product made from virgin material is recycled into another product, which is not recycled again, but disposed of, possibly after a long-term diversion (Fig. 23.1b) (Vigon *et al.*, 1993).

Demolished concrete is recycled into a recycled concrete aggregate which, if obtained by a traditional two-stage crushing procedure, has inferior properties to NA. Concrete made completely of such an aggregate typically has inferior properties compared to the parent NAC, when made with the same w/c ratio. The possibility of repeated recycling of RAC has not yet been investigated. With the development of research, green procurement and markets, concrete recycling has a good chance of becoming a closed-loop system in the near future; however, for the above reasons it will be understood here as open-loop recycling (Fig. 23.1b).

Product 1 (the product that is recycled) in Fig. 23.1b is NAC. Product 2 (the product that uses recycled material) is, for example, RAC where RCA is used as

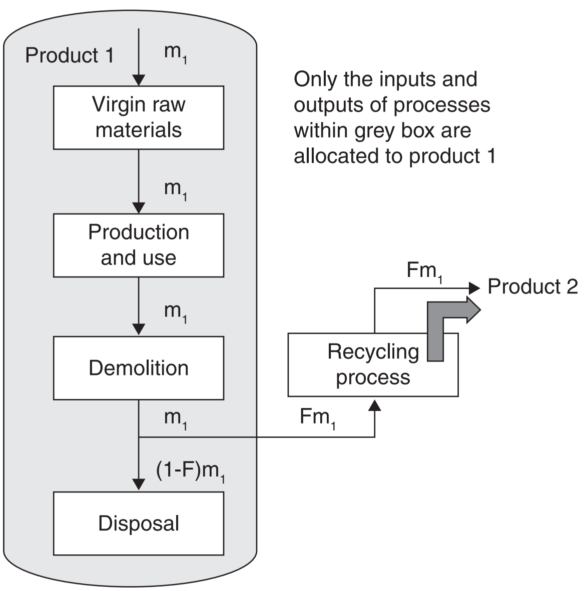


23.1 Closed-loop (a) and open-loop (b) recycling.

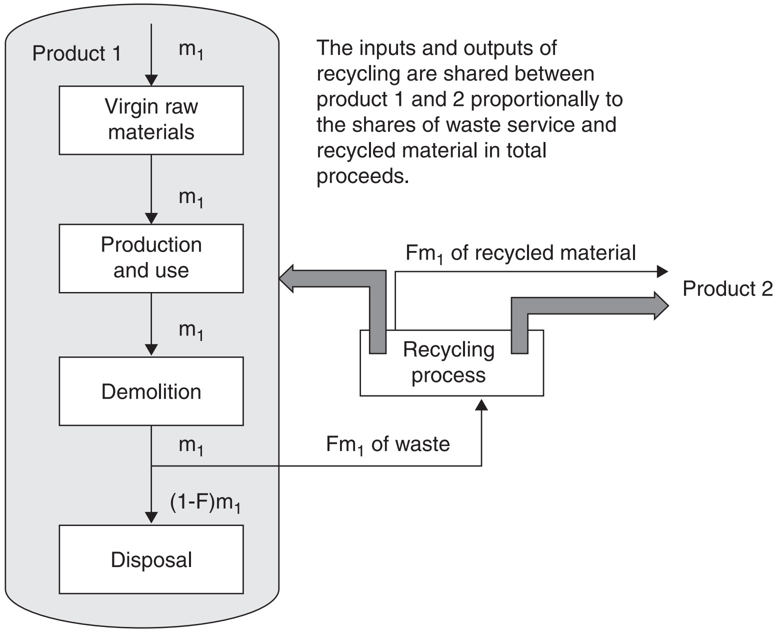
aggregate, or a road where RCA is used as a base material. Some authors argue that economic allocation cannot be avoided in recycling, since neither of the two first steps recommended by the ISO is feasible (Werner and Richter, 2000; Votgländer *et al.*, 2001). The second step, allocation based on physical relationship, is rarely possible in the case of concrete open-loop recycling, since the change in physical or mechanical properties of a material cannot be described consistently using only one parameter.

23.3.1 Economic allocation versus cut-off rule

The most traditional allocation procedure for re-use and recycling, which is applied in most of the LCA software, is not to apply allocation at all. This is a simple cut-off rule, where a product made out of primary materials carries the burdens of those primary materials, while a product made out of secondary materials carries the environmental burdens of the recycling activities of those secondary materials (Ekvall and Tillman, 1997). In the case of concrete, this means that environmental burdens of all stages from raw material production to the disposal of non-recyclable wastes are included in the system being studied. The environmental burdens of recycling are excluded from the system, as they are considered burdens to the next product system (Fig. 23.2) (Gervásio *et al.*, 2009).



23.2 Cut-off rule.



23.3 Economic allocation.

The cut-off method in general has certain advantages: it is well established in practice and relatively easy to apply. Its main limitation is that it does not give credit to the party that made the effort to implement recycling. Although economic allocation gives credit to both involved parties, its main disadvantage is that it is based on market prices which are, in the case of recycled materials and markets, highly unstable and fluctuating. Many different solutions to the allocation problem have been suggested (Ekvall and Tillman, 1997; Ekvall, 2000; Werner and Richter, 2000; Votgländer *et al.*, 2001; Guinée *et al.*, 2002, 2004). The following example of an application of these two allocation procedures presents the method proposed by Guinée *et al.* (2004). According to these authors, the recycling process should be allocated partly to Product 1 and partly to Product 2 (Fig. 23.3), proportionally to the shares of waste management service and recycled material in the total economic proceeds. The economic proceeds are calculated based on the quantity and market price of the service or product.

23.3.2 A case study: cut-off rule and economic allocation in concrete recycling

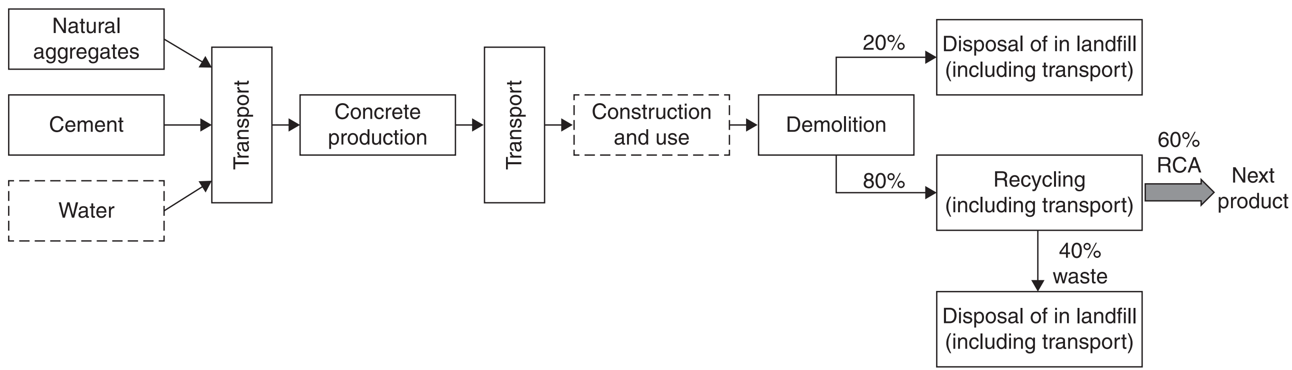
The following case study compares the environmental impacts of ready-mixed NAC production for cut-off rule and economic allocation applied in the context of

LCA of NAC production in Serbia. For comparison, the system boundaries are defined as follows. The analysed part of the life cycle includes the production and transport of raw materials and concrete, demolition and end-of-life phase (Fig. 23.4). The construction and use phases are excluded, because the impacts of these phases significantly depend on the type of concrete structure. Regarding the end-of-life scenario, it is assumed that 20% of demolished concrete is disposed of in landfill, while 80% is recycled. It is assumed that recycling is performed in a mobile recycling plant on a demolition site. The recovery rate is assumed to be 60% (Nagataki *et al.*, 2004), which means that, from 1 kg of demolished concrete, 0.6 kg of RCA can be obtained; the rest is disposed of in landfill. Therefore, 'recycling' includes the recycling process itself, transportation of the mobile recycling plant to the demolition site, and the landfill of the recycling waste that cannot be used as RCA. The functional unit of 1 m^3 of ready-mixed concrete is used in this work.

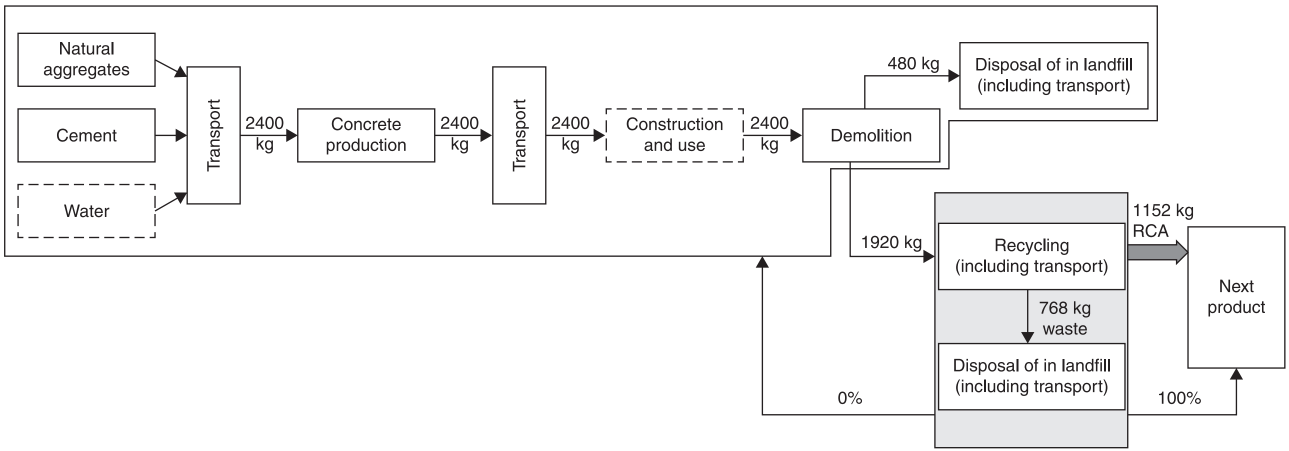
The production of ready-mixed concrete takes place in Serbia; all the LCI data for aggregate, cement and concrete production, and demolition and recycling processes, are therefore collected from local suppliers and manufacturers (Marinkovic *et al.*, 2008). Emission data for diesel production and transportation, natural gas distribution, and transport, which could not be collected for local conditions, are taken from the GEMIS database (Öko-Institut, 2008). The LCA in this case study focuses on energy use and airborne emissions. They are calculated using an Excel-based software made for LCI and life-cycle impacts calculation. For impact categories calculation, CML methodology (Guinée *et al.*, 2002) is used. This software is applied in all case studies presented in this chapter.

Transport types and distances are estimated as typical for the construction/demolition site, which is located in Belgrade, the Serbian capital (Table 23.1). It is usually river aggregate that is used in Serbia for concrete production, transported by medium-sized ships. For each load of 2500 t, the mobile plant (20 t) is transported 200 km. Such a long distance was assumed, since there is only one mobile recycling plant available in Serbia at present. The mix proportion of concrete was determined from two conditions: the target concrete strength class was C25/30 (characteristic compressive cylinder/cube strength equal to 25/30 MPa), nomenclature according to Eurocode 2 (CEN, 2004), and the target slump 20 min after mixing was 6 ± 2 cm.

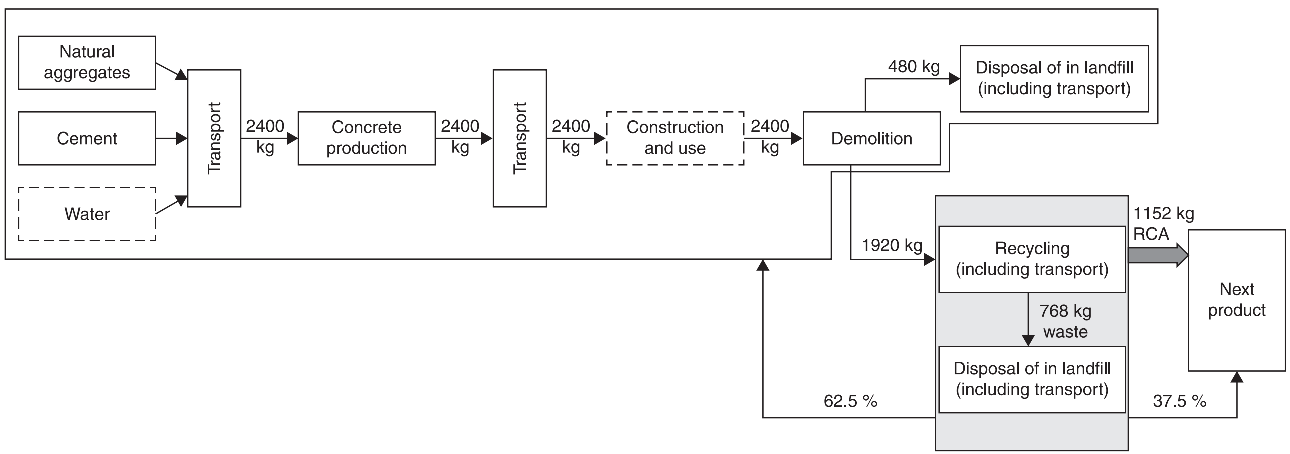
Figure 23.5 illustrates the cut-off rule: all the inputs and outputs from the recycling process are excluded from the NAC system; only the impacts from landfilling are accounted for. Figure 23.6 illustrates the economic allocation, where allocation factors are based on the economic proceeds. The waste has negative economic value, since the waste producer has either to pay a landfill tax for its disposal or a recycler to collect it. It is reasonable to assume that the waste producer will not pay more than the cost of the landfill tax, so the landfill tax is



23.4 NAC life-cycle.



23.5 NAC life-cycle: system boundaries in the cut-off rule case.



23.6 NAC life-cycle: system boundaries in the economic allocation case.

Table 23.1 Transport distances and types

Material	Route		Transport distance (km)	Transport type
	From	To		
River aggregate	Place of extraction	Concrete plant	100	Ship 10 000 t
Cement	Cement factory	Concrete plant	150	Truck 28 t
Concrete	Concrete plant	Construction site	10	Mixer 11 t
Demolished concrete	Demolition site	Landfill	30	Truck 28 t
Waste from recycling	Demolition site	Landfill	30	Truck 28 t
Mobile recycling plant[1]		Demolition site	200	Truck 28 t

Note: [1] For each campaign of 2500 t the mobile plant (20 t) is transported a distance of 200 km.

adopted as waste price. The value of landfill tax for inert waste varies a lot between European countries, from zero (no landfill tax) to a situation where landfilling is partially forbidden (e.g. landfilling of combustible C&D waste is banned in the Netherlands). However, a rough estimate of the average value in Europe is 10 EUR per ton of inert waste (Fischer, 2011; CEWEP, 2011). The average price of RCA is assumed to be 10 EUR per ton (WBCSD, 2009). Quantities, prices, proceeds and allocation factors are presented in Table 23.2, which shows that 62.5% of inputs and outputs of recycling process is allocated to the NAC system, and 37.5% is allocated to the product system which receives RCA, for example RAC.

Calculated cumulative energy requirements and emissions to air (so-called inventory table) are presented in Table 23.3 for the cut-off case, and in Table 23.4 for the economic allocation case. Energy use and airborne emissions are higher in the economic allocation case compared with the cut-off case (Table 23.5). Table 23.5 shows that this increase is below or about 2.0%, except for N_2O

Table 23.2 Allocation factors for recycling of demolished concrete

Functional flow	Quantity (tone)	Price (EUR/tone)	Proceeds (EUR)	Allocation factor
Demolished concrete	1.920	10.00	19.20	19.20/30.72 = 0.625
Recycled concrete aggregate	1.152	10.00	11.52	11.52/30.72 = 0.375
Total	–	–	30.72	1.000

Table 23.3 Inventory table of 1 m^3 of NAC – cut-off rule

	Cement (kg) 315.00	Aggregate (kg) 1879.00	Concrete 1 m^3	Demolition 1 m^3	Transport 1 m^3	Landfilling 480 kg of waste	Recycling (0%) 1920 kg of waste	Total
Energy (MJ)								
Coal	1061.594							1061.594
Diesel	7.676	27.772		89.591	267.187	22.189		414.415
Natural gas	26.201							26.201
Electricity	159.917		20.069					179.986
Emission to air (g)								
CO	1324.016	6.530	0.723	21.064	66.836	4.591		1423.759
NO_x	717.906	29.273	13.224	94.435	175.220	14.175		1044.234
SO_x	1148.789	10.235	98.754	33.016	75.587	6.206		1372.586
CH_4	315.866	2.435	0.433	7.854	20.894	1.784		349.266
CO_2	271278.882	2589.123	5698.210	8352.512	19257.738	1595.088		308771.553
N_2O	0.238	0.103	0.029	0.331	0.557	0.042		1.301
HCl	21.357		2.680					24.037
HC	0.183		0.023					0.206
NMVOC	10.941	0.737	0.071	2.267	32.527	1.796		48.339
Particles	224.274	2.734	11.991	8.819	42.377	2.783		292.978

Table 23.4 Inventory table of 1 m^3 of NAC – economic allocation

	Cement (kg) 315.00	Aggregate (kg) 1879.00	Concrete 1 m^3	Demolition 1 m^3	Transport 1 m^3	Landfilling 480 kg of waste	Recycling (0.625%) 1920 kg of waste	Total
Energy (MJ)								
Coal	1061.594							1061.594
Diesel	7.676	27.772		89.591	267.187	22.189	34.099	448.514
Natural gas	26.201							26.201
Electricity	159.917		20.069					179.986
Emission to air (g)								
CO	1324.016	6.530	0.723	21.064	66.836	4.591	7.308	1431.067
NO_x	717.906	29.273	13.224	94.435	175.220	14.175	25.501	1069.735
SO_x	1148.789	10.235	98.754	33.016	75.587	6.206	10.332	1382.918
CH_4	315.866	2.435	0.433	7.854	20.894	1.784	2.806	352.072
CO_2	271278.882	2589.123	5698.210	8352.512	19257.738	1595.088	2642.353	311413.906
N_2O	0.238	0.103	0.029	0.331	0.557	0.042	0.081	1.382
HCl	21.357		2.680					24.037
HC	0.183		0.023					0.206
NMVOC	10.941	0.737	0.071	2.267	32.527	1.796	2.272	50.611
Particles	224.274	2.734	11.991	8.819	42.377	2.783	4.035	297.014

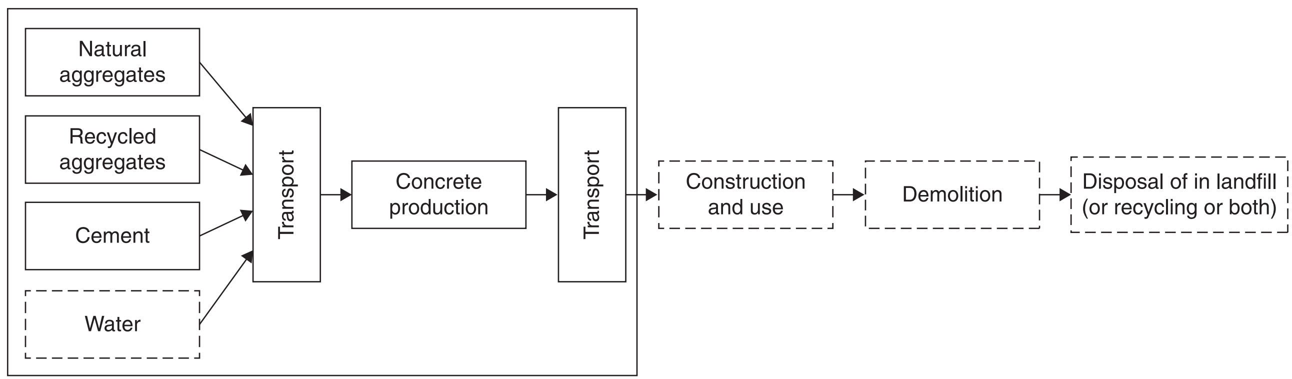
Table 23.5 Economic allocation over cut-off increase

	Cut-off rule	Economic allocation	Economic allocation over cut-off increase (%)
Energy (MJ)	1682.196	1716.295	2.03
Emission to air (g)			
CO	1423.759	1431.067	0.51
NO_x	1044.234	1069.735	2.44
SO_x	1372.586	1382.918	0.75
CH_4	349.266	352.072	0.80
CO_2	308771.553	311413.906	0.86
N_2O	1.301	1.382	**6.22**
HCl	24.037	24.037	0
HC	0.206	0.206	0
NMVOC	48.339	50.611	**4.70**
Particles	292.978	297.014	1.38

emissions and NMVOC, which are slightly larger. The increase can be considered negligible and a simple cut-off rule can be recommended in this specific case of concrete recycling. If the construction and use phase had been brought into consideration, the differences would be even smaller. If the recycling rate was assumed to be less than 80%, which is a more realistic situation in practice, the differences would also be smaller. Such results can generally be expected, because cement production is by far the largest contributor to the environmental impacts of concrete (if the use phase is excluded); the contribution of the recycling phase is comparatively small.

23.4 A case study: LCA of recycled aggregate concrete (RAC) production compared to natural aggregate concrete (NAC) production

In this chapter, the environmental implications of RCA utilization in structural concrete are analysed using LCA. Published research in this area is limited (Braunschweig *et al.*, 2011; Weil *et al.*, 2006; Evangelista and de Brito, 2007). However, Braunschweig *et al.* (2011) have found that the environmental impacts of high-quality NAC and 25% recycled-aggregate RAC are similar, as long as the increase in the amount of cement in RAC is small (up to a few percent). They compared the following impacts: energy use, climate change (global warming), acidification, respiratory effects, land use and gravel use. Weil *et al.* (2006) compared NAC to 35 and 50% recycled-aggregate RAC with different cement contents. Their conclusion is similar to that of Braunschweig *et al.* (2011): if the cement content is the same, the energy use and global warming potential of NAC and RAC are similar; if the cement content in RAC is larger than in NAC, the impacts from RAC are



23.7 Life-cycle of a concrete structure and system boundaries in the case study.

greater than those from NAC. For a 40% higher cement amount in RAC, the energy use and global warming potential of RAC exceed those of NAC by 36 and 39%, respectively. Although some important assumptions regarding system boundaries were not reported in these studies (especially regarding the transport distances), it is obvious that impacts from NAC and RAC mainly depend on cement content.

The specific case study is used to compare the environmental impact of the production of two types of ready-mixed concrete in Serbia: natural aggregate concrete (NAC) made entirely with river aggregate and RAC made with natural fine and recycled coarse aggregate (100% replacement ratio). With this goal in mind, system boundaries in this study are determined (Fig. 23.7). Regarding the production of recycled concrete aggregate, the cut-off rule is applied, which means that no impacts from parent NAC production and all the impacts from recycling are allocated to RCA production. Production of RCA includes the recycling process itself – which takes place in a mobile recycling plant, typical for Serbia – transportation of the mobile recycling plant to the demolition site, and the landfilling of the recycling waste that cannot be used as RCA (assumed recovery rate 60%). For each load of 2500 t, the mobile plant (20 t) is transported a distance of 200 km.

The construction, use and demolition phases are excluded. To compare the environmental impact of different concrete types, both concrete types need to fulfil similar functional requirements. This means they will have approximately the same strength and durability performance: mechanical properties, workability and durability-related properties. The mix proportions of NAC and RAC are therefore determined, so that both types of concrete have the same compressive strength and workability. However, rheological and durability-related properties still may not be the same, and it is likely that RAC will have a lower durability performance than NAC.

To enable a comparative environmental impact assessment, the analysis is limited to a type of concrete structure for which non-aggressive environmental conditions apply (i.e. an indoor environment in residential/office buildings). Otherwise, the exclusion of the service phase would not be acceptable, because of the possible difference in the durability performances of the two concrete types. Under all these assumptions, the impact of the construction, use and demolition phases is expected to be approximately the same for both concrete types, and these phases need not to be taken into consideration in comparative analysis. If the end-of-life scenario is assumed to be the same for both concrete types, its impact will be the same and this phase, also need not to be taken into consideration in comparative analysis.

Due to the above-mentioned assumptions, it was possible to implement 1 m^3 of ready-mixed NAC and RAC as a functional unit in this work. All the LCI data for natural and recycled aggregate, cement and concrete production are collected from local suppliers and manufacturers (Marinkovic *et al.*, 2008). Emission data for diesel production and transportation, natural gas distribution and transport that could not be collected for local conditions are taken from the GEMIS database

(Öko-Institut, 2008). The environmental impact categories included in this work are: global warming (GW), eutrophication (E), acidification (A) and photochemical oxidant creation (POC). They are calculated using the above-mentioned software by multiplying the emission results by their corresponding characterization factors (CML methodology) (Guinée *et al.*, 2002). Cumulated energy requirement from the studied part of the life cycle is calculated and expressed as energy use (EU).

The typical transport distances are estimated for the construction site in Belgrade, as follows:

- cement transported by heavy trucks (28 t) from cement factory to concrete plant: estimated distance 150 km;
- natural river aggregate (sand and gravel) transported by medium-sized ships (10 000 t) from extraction place to concrete plant: estimated distance 100 km;
- recycled aggregate transported by medium-heavy trucks (11 t) from the recycling plant to the concrete plant: estimated distance 15 km;
- concrete transported by concrete mixers from the concrete plant to the construction site: estimated distance 10 km.

23.4.1 NAC and RAC properties

For this comparative analysis, it was necessary to determine the NAC and RAC mix proportions, so that both concrete types have the same compressive strength and workability. Coarse recycled aggregate for the RAC mix was obtained from a demolished reinforced concrete bridge structure and waste laboratory concrete samples. The crushing of the demolished concrete and the screening into three fractions (4/8, 8/16 and 16/31.5) was performed in the mobile recycling plant. The properties of recycled concrete aggregate were tested and the results are shown in Table 23.6. The component materials used for both concrete types were:

- Portland cement EN 197–1 – CEM I 42.5 R;
- fine aggregate, fraction 0/4 mm (river aggregate, Morava river);
- two types of coarse aggregate, fractions 4/8, 8/16 and 16/31.5 mm: river aggregate (Morava river) for NAC and recycled aggregate for RAC;
- water.

The target properties for both types were: compressive cube strength at 28 days equal to 42 MPa and slump 30 min after mixing equal to 8 ± 2 cm. Laboratory tests with various mix proportions of NAC and RAC were performed to obtain these target values. Because of the high water absorption of recycled aggregates, it was necessary to add a certain amount of water to saturate the recycled aggregate before or during mixing to obtain the desired RAC workability. In this case, dried recycled aggregate was used and the additional water quantity was calculated on the basis of recycled aggregate water absorption after 30 min. No water-reducing admixtures were used. The final results, that is, the mix proportions and properties of NAC and RAC obtained, are shown in Table 23.7. The table shows that a

Table 23.6 Recycled concrete aggregate properties

Property	Unit	Fraction		
		4/8	8/16	16/31.5
Crushing resistance (in cylinder)	Mass loss (%)	23.0	26.1	32.7
Volumetric coefficient	(%)	29.0	20.7	28.6
Loose bulk density	(kg/m^3)	1132	1260	1520
Fines content	(%)	0.38	0.38	0.0
Water absorption after 30 min	(%)	4.1	4.0	3.7

slightly larger amount of cement, that is a slightly smaller free w/c ratio (~3%), is required for RAC than for corresponding NAC. Also, the density of RAC is slightly lower (~4%) than the density of NAC, because of the lower density of recycled concrete aggregate. The w/c ratio (Table 23.7) refers to free water content, excluding additional water.

23.4.2 Results and discussion

The calculated cumulative energy requirement and emissions to air for the production and transport of 1 m^3 of ready-mixed NAC and RAC for the assumed transport scenario are presented in Table 23.8. The amounts of component materials (cement, natural and recycled aggregate) are in accordance with tests performed to determine the mix proportions of NAC and RAC with the same compressive strength and workability. Figure 23.8 shows the contribution of various phases of the concrete life cycle to the studied impact categories and a total value of impact categories for NAC and RAC. The results show that cement production is the largest contributor to all impact categories, for both NAC and RAC. It causes approximately 81% of the total energy use, 92% of the total global warming, 79% of the total eutrophication, 85% of the total acidification and 77% of the total POC. The main reason for this is substantial CO_2 emission during the calcination process in clinker production, and the use of fossil fuels. The contribution of aggregate and concrete production is very small, while the contribution of transport lies somewhere in between.

However, the impacts of cement and aggregate production life-cycle phases for RAC are slightly larger than for NAC.

The environmental impacts of RAC over NAC increase for the assumed transport scenario is shown in Table 23.9. As the table shows, the total impact of RAC for each category is slightly larger than the impact of NAC, an increase ranging from 3.2 to 3.9%. As the total environmental impact of RAC and NAC production in terms of calculated impact categories is approximately the same, the benefit from recycling in terms of minimizing waste and natural mineral resources depletion is clearly gained. In this specific case, the landfilling of 1071 kg of

Table 23.7 Mix proportions and properties of NAC and RAC

Type of concrete	Cement (kg/m^3)	Water (kg/m^3)	Aggregate		w/c[1] /	a/c[2] /	Density (kg/m^3)	Slump after 30 minutes (cm)	Compressive strength at 28 days (MPa)
			Fine (kg/m^3)	Coarse (kg/m^3)					
NAC	354	185	600	1164	0.524	4.983	2400	8.0	43.7
RAC	365	180+38[3]	576	1071[4]	0.511	4.512	2310	9.7	42.5

Notes:

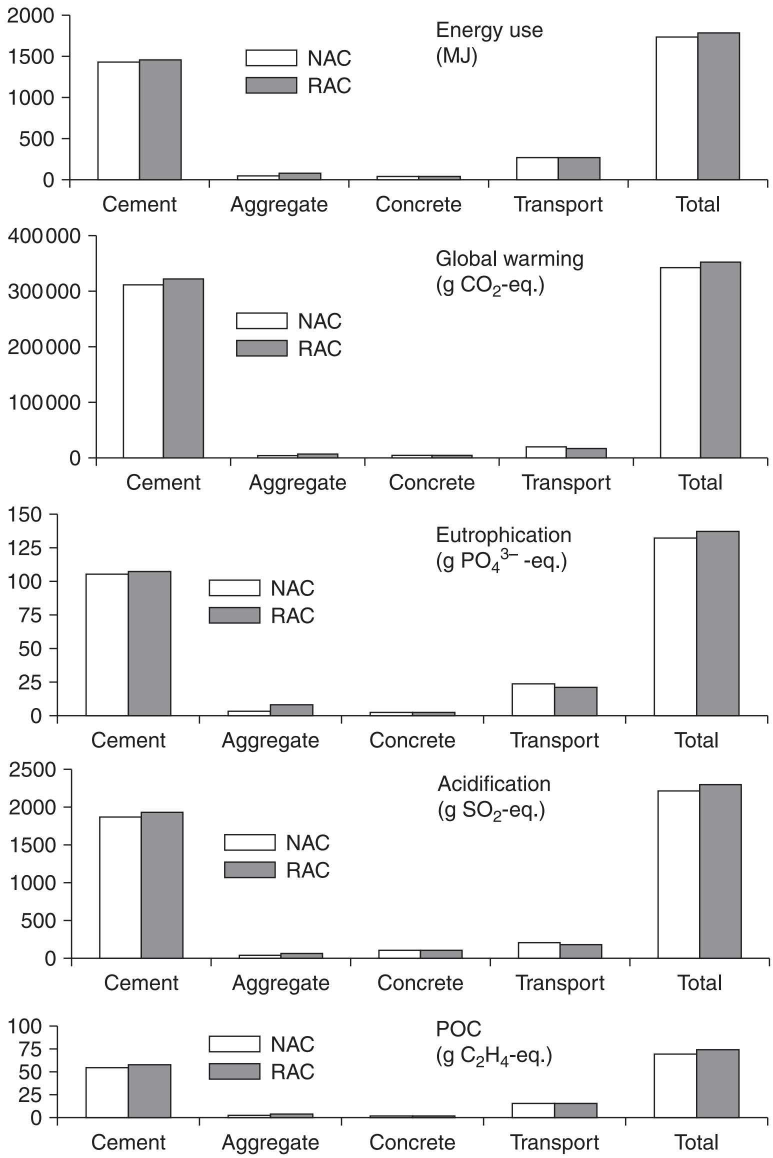
[1] Water-to-cement ratio
[2] Aggregate-to-cement ratio
[3] Additional water quantity
[4] Recycled aggregate

Table 23.8 Inventory table per 1 m^3 of NAC and RAC

	Cement (kg)		Aggregate (kg)			Concrete (1 m^3)		Transport (for 1 m^3 of concrete)		Total (for 1 m^3 of concrete)	
	NAC 354.00	RAC 365.00	NAC NA[1]: 1764.00	RAC NA: 576.00	RCA[2]: 1071.00	NAC	RAC	NAC	RAC	NAC	RAC
Energy (MJ)											
Coal	1193.03	1230.10								1193.03	1230.10
Diesel	8.63	8.89	26.07	8.51	56.90			269.30	253.06	304.00	327.37
Natural gas	29.45	30.36								29.45	30.36
Electricity	179.72	185.30				20.07	20.07			199.78	205.37
Emissions to air (g)											
CO	1487.94	1534.18	6.13	2.00	12.32	0.72	0.72	66.91	63.48	1561.71	1612.70
NO_x	806.79	831.86	27.48	8.97	44.44	13.22	13.22	176.07	158.17	1023.57	1056.67
SO_x	1291.02	1331.14	9.61	3.14	17.64	98.75	98.75	76.14	71.25	1475.52	1521.92
CH_4	354.97	366.00	2.29	0.75	4.72	0.43	0.43	21.09	19.84	378.78	391.74
CO_2	304865.79	314339.02	2430.66	793.69	4506.11	5698.21	5698.21	19406.78	18207.92	332401.44	343544.95
N_2O	0.27	0.28	0.10	0.03	0.14	0.03	0.03	0.56	0.52	0.95	1.00
HCl	24.00	24.75				2.68	2.68			26.68	27.43
HC	0.21	0.21				0.02	0.02			0.23	0.23
NMVOC	12.30	12.68	0.69	0.23	3.55	0.07	0.07	32.39	31.55	45.44	48.07
Particles	252.04	259.87	2.57	0.84	6.61	11.99	11.99	41.76	27.02	308.36	306.33

Notes:

[1] Natural aggregate
[2] Recycled aggregate



23.8 Contribution of different NAC and RAC life-cycle phases to category indicators results.

Table 23.9 Environmental impacts increase of RAC over NAC

Impact category	RAC over NAC impact increase (%)
Energy use	3.88
Global warming	3.36
Eutrophication	3.23
Acidification	3.17
POC	3.93

concrete waste and the extraction of 1071 kg of NA is avoided per 1 m^3 of RAC. It should be noted here that these conclusions are valid only under the assumptions made in this study, especially considering the transport distances of RCA and NA. The ratio of transport distances of RCA to NA was assumed to be 15 km: 100 km, which means that the recycling plant must be located much closer to the concrete plant than the place of NA extraction, if environmental benefits from recycling are to be expected.

23.5 LCA of low-grade applications of RCA

Despite all the research performed in the area of RCA utilization in structural concrete, this type of aggregate has, up to now, been most often used in road construction as a base and sub-base material. As in the comparison of RAC and NAC, the obvious advantages of RCA application are the avoidance of concrete waste landfilling and the saving of NA resources. The question of total environmental impact still remains, however. LCA has already been successfully applied to the environmental impact assessment of different aggregate types for base and sub-base in road construction.

Mroueh *et al.* (2001) concluded that the use of crushed concrete waste decreased the environmental burdens of road construction when compared to NA. Their conclusion is valid only for the construction and transport distances assumed in the specific case study. However, Chowdhury *et al.* (2010) have found that RCA had higher impacts on energy use, global warming and acidification compared to NA. They analysed the influence of transport distances and found that, if the transport distance ratio of RCA to NA was more than 1: 4, the impacts on energy use and global warming of RCA were smaller than those of NA. These contradictory conclusions have been caused by the use of different databases, system boundaries and functional units. However, it can be concluded that transport types and distances significantly affect environmental impacts, since transportation is a high energy consumption phase in the road construction life cycle.

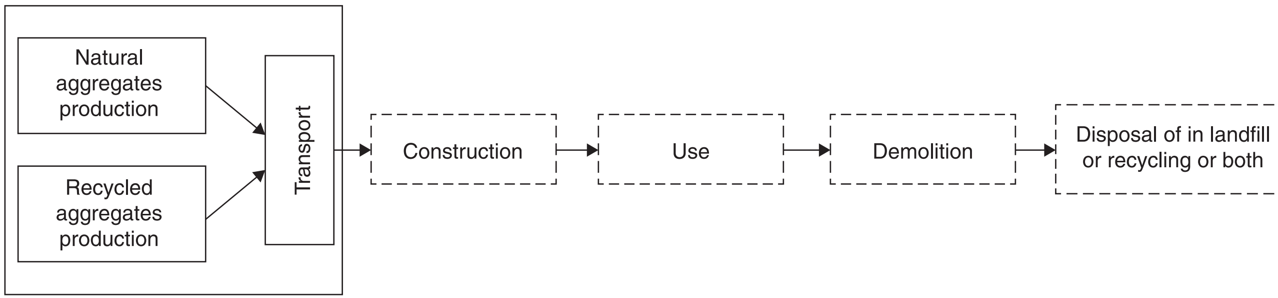
23.5.1 A case study: comparative LCA of recycled and natural aggregate application in road base

This section presents a comparative environmental assessment of the road base application of RCA and NA (river gravel). LCA methodology is applied, with system boundaries (Fig. 23.9). The assumption is made that the required properties of both aggregate types are approximately the same, meaning the impacts from the construction, use and end-of-life phases are also approximately the same. This assumption also allows 1 kg of aggregate to be adopted as a functional unit (the same road structure in both cases). However, this assumption is not always valid. Depending on the type and quality of concrete waste, RCA can have better bearing capacity than gravel or crushed NA, but may need more demanding compaction (Wahlström *et al.*, 2000, Mroueh *et al.*, 2001).

For RCA production, the cut-off rule is applied, which means that no impacts from the parent NAC production and all the impacts from the recycling process are allocated to RCA production. Recycling includes the transportation of the mobile plant (20 t over 200 km), the recycling process itself and landfilling of the recycling waste, which cannot be used as RCA (40%). LCI data for natural (gravel) and recycled aggregate production are collected from local suppliers and manufacturers (Marinkovic *et al.*, 2008). Emission data for diesel production and transportation, natural gas distribution and transport that could not be collected for local conditions are taken from the GEMIS database (Öko-Institut, 2008). The environmental impact categories included in this work are global warming (climate change), eutrophication, acidification and photochemical oxidant creation. They are calculated according to CML methodology (Guinée *et al.*, 2002), using the above-mentioned Excel-based software made for LCI and life-cycle impact calculation. Cumulative energy requirements during the studied part of the life cycle are calculated and expressed as energy use.

It should be pointed out here that leaching behaviour of natural and RCA is also regarded as a significant parameter in the environmental impact assessment of road construction. Both types of aggregate can potentially leach and contaminate groundwater, surface water and soil in the surrounding area. Several researchers have dealt with leaching from natural and recycled aggregates in road construction (Wahlström *et al.*, 2000; Engelsen *et al.*, 2009; Tossavainen and Forssberg, 1999; Mroueh *et al.*, 2001). Although there are many factors that affect leaching behaviour in practice, based on the above-mentioned research, it can be concluded that the leaching of heavy metals from RCA is generally smaller than that from NA. For the purpose of this case study, the data on leaching of heavy metals were taken from Wahlström *et al.* (2000) and Mroueh *et al.* (2001) (Table 23.10).

The toxicity impacts that are affected by the release of heavy metals into the environment are human toxicity, aquatic and terrestrial ecotoxicity potentials. They are calculated by multiplying the characterization factors with the amount of released pollutants per unit and then adding all the multiplied values together to



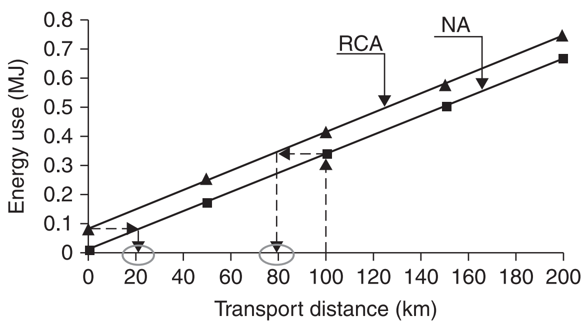
23.9 Life-cycle of a road structure base and system boundaries in the case study.

Table 23.10 Leaching of heavy metals from natural (gravel) and recycled concrete aggregates (mg/kg)

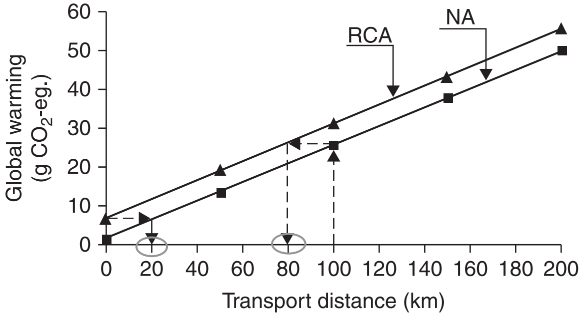
Aggregate	Heavy metal							Reference
	Cd	Cr	V	Pb	Zn	As	Mo	
NA	0.0002	0.004	0.010	0.004	0.040	0.006	0.020	Mroueh *et al.* (2001)
RCA	<0.0002	0.027	–	0.003	0.010	–	–	Wahlström *et al.* (2000)

gain a single indicator result (Guinée *et al.*, 2002). Since the amount of each heavy metal leached from RCA (except for chromium) is smaller than that from NA, it is obvious that all the impacts resulting from leaching are smaller in the RCA case than in the NA case; they were therefore not calculated. Transportation type is assumed to be the same for both types of aggregate – medium-heavy trucks. Transportation distance from the extraction place or recycling plant (in this case, demolition site) to construction site varies from 0 to 200 km, to evaluate the influence of the transport distance on the environmental impacts of the two aggregate types. The results for each impact category are shown in Figures 23.10 to 23.14.

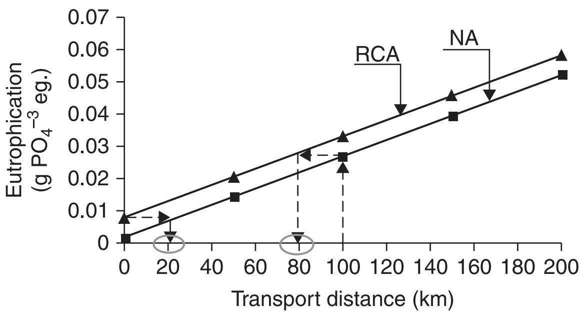
Keeping all other parameters constant and varying only the aggregate transport distance, it is possible to determine the 'limit' transport distance of NA. This is defined as NA transport distance below which environmental impact of RCA is higher than the environmental impact of NA, regardless of RCA transport distance. Figures 23.10 to 23.14 show the relationship between the transport distance of aggregate (natural and recycled) and calculated environmental impacts. It can be seen from these figures that the limit distance for NA is about 20 km: for transport



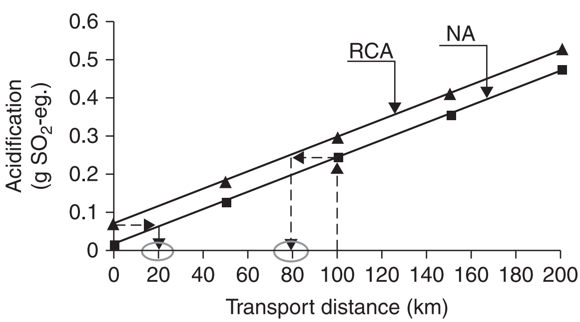
23.10 Relationship between energy use and transport distance of RCA and NA.



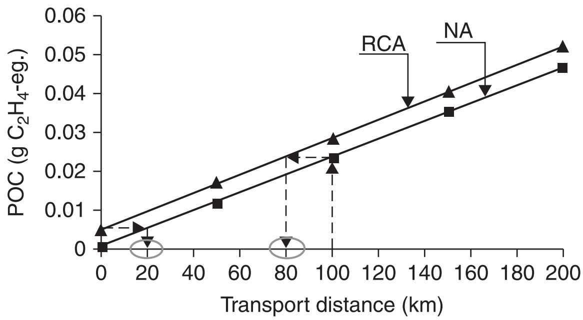
23.11 Relationship between global warming and transport distance of RCA and NA.



23.12 Relationship between eutrophication and transport distance of RCA and NA.



23.13 Relationship between acidification and transport distance of RCA and NA.



23.14 Relationship between POC and transport distance of RCA and NA.

distances below this value, environmental impacts from RCA production and transport are higher than environmental impacts from NA production and transport. For transport distances larger than 20 km, RCA impacts can be equal to or lower than NA impacts. For example, if the transport distance of NA is 100 km, the required recycled aggregate transport distance is about 80 km, for the same impacts. This case study is based on Serbian LCI data and typical conditions in Serbia. It is based on the assumptions regarding the quality of aggregates and, consequently, the system boundaries and functional unit. Within these limits, it is concluded that RCA application in road sub-base and base can be more environmentally beneficial than NA, but this depends on the ratio of transport distances of natural and recycled aggregate.

23.6 LCA of waste management systems

The contribution of various phases to total life-cycle impacts depends on the type of construction. For buildings, life-cycle impacts are often dominated by energy consumption during the use phase: it is estimated that the use phase in conventional buildings represents approximately 80 to 94% of the life-cycle energy use; 6 to 20% is consumed in materials extraction, transportation and production, and less than 1% is consumed in the end-of-life phase (Bragança and Mateus, 2009). Blegnini (2009) found that, when taking into account other environmental impacts (GW, A, E, POC, ozone depletion-OD), the contribution of the use phase is 93%, while the contributions of all the other life-cycle phases make up the rest (end-of-life phase contribution is 2.6%).

For other types of construction work, such as bridges or dykes, the contribution of material production, construction and end-of-life phases is more significant: indeed, the impacts from these phases usually exceed the impacts from the use phase. With the growing interest in the development of technologies for reducing the energy consumption of buildings, the other life-cycle phases are becoming more significant. The material production and end-of-life phases are of special interest in this context, since they are energy-, resource- and waste-intensive. The amount of waste that will be disposed of in landfill and the amount of material that is eventually going to be recovered at the end of a life cycle, depend on the chosen waste management scenario.

Some research regarding the LCA of different C&D waste management scenarios has already been published (Blegnini, 2009; Blegnini and Garbarino, 2010; Coelho and de Brito, 2012; Mercante *et al.*, 2012; Ortiz *et al.*, 2010). Most of these works concern specific case studies, where only the end-of-life phase is considered. This typical simplification, when comparing different waste management scenarios, is sometimes called the 'zero burden assumption'. It suggests that the waste carries no upstream burdens into the waste management system, and the functional unit of waste is assumed to be kg or t. Such LCA models only allow environmental comparisons of different waste management options (Ekvall *et al.*, 2007). Studies where all the life-cycle phases of a building are included, for example Blegnini (2009) or Coelho and de Brito (2012), enable evaluation of the participation of the end-of-life phase in the whole life cycle for different waste management options – a much broader and complex perspective.

Two scenarios are typically analysed when considering concrete waste: landfilling and recycling. Only in Coelho and de Brito (2012) has the re-use scenario also been analysed. In all the studies considering only end-of-life phase, the net impacts from the recycling option were calculated as a difference between avoided impacts and induced impacts from recycling itself and subsequent transportation. Avoided extraction of NA was considered as avoided impact; Blegnini and Garbarino (2010) also included avoided landfilling. This means that net recycling impacts can have positive value (if induced impacts are larger than avoided impacts) or negative value (if induced impacts are smaller than avoided impacts). Methodology like this does not strictly follow the LCA methodology, since the recycling is completely allocated to the product that generates waste.

The results of all case studies point to the same conclusion: recycling is more environmentally beneficial than landfilling, if not in all the calculated impacts then in most of them. According to Blegnini (2010), 13 out of 14 calculated impact indicators for recycling had negative values, meaning that avoided impacts were higher than induced impacts. The largest contributor in the recycling chain was avoided landfill, followed by avoided quarrying and transportation. Moreover, although transportation was a significant contributor, sensitivity analysis showed that assumptions on delivery distances did not affect the overall conclusion. The distance of RA transportation should increase by a factor of 2 or 3 before the induced impacts outweigh the avoided impacts (Blegnini, 2010).

Ortiz *et al.* (2010) compared three different scenarios in their case study: landfilling, recycling and incineration of various types of C&D waste. In the case of concrete waste, the calculated impacts (GW, A, E, toxicity indicators) corresponding to recycling had small positive values, meaning that generated impacts were slightly higher than avoided impacts. Even so, these impacts were always lower than those caused by landfilling: recycling is therefore always the recommended option. Considering the influence of the transport distances, the study concluded that concrete waste should be recycled close to the demolition site, that is, long distances from the demolition site to the recycling plant were not recommended. These conclusions are somewhat different to those of Blegnini (2010), which is probably due to the fact that the study only counted avoided NA production as an avoided impact, not avoided landfilling.

Mercante *et al.* (2012) considered only the recycling option in their specific case study and compared two different types of inert waste sorting and treatment facilities in Spain. Their results were not directly comparable to the previous two studies, since the data for recycling were separated from the transportation data. However, they obtained negative values for all calculated impacts (GW, POC, OD, E and A) for the recycling process itself, although they took only avoided NA production (and not avoided landfilling) into consideration. Coelho and de Brito (2012) compared five different end-of-life scenarios, taking into account:

- material production and end-of-life phase of a building life cycle; and
- all life cycles of a building.

Their study included various types of C&D waste, not only demolished concrete. End-of-life scenarios ranged from uncontrolled demolition and landfilling to total piece-by-piece deconstruction with 50% of material recovered for re-use and the other 50% recycled. The study found that, at the level of the whole life cycle, significant reductions in GW, A, E and POC can only be obtained in the case of full deconstruction, and for re-use and recycling of all the waste, except the hazardous waste. In this rather hypothetical scenario, obtained reductions in the mentioned impacts ranges from 4 to 22%.

The reduction of impacts in a scenario with less intensive demolition techniques and only a small part of the waste material recycled (compared to a scenario with complete demolition and landfilling) was practically negligible (below 2%). When compared to the material production phase, the situation was more favourable. In the case of full recycling and re-use scenarios, reductions ranged from 36 to 81%; where a small part of the waste was recycled, differences ranged from 0 to 7%, depending on the impact category. It was concluded that to achieve any obvious reduction in environmental impact from a complete life-cycle perspective, it is necessary to raise the recycling level above 90% and to incorporate as far as possible the resulting material into new construction. However, it should be kept in mind that scenarios that include recycling and re-use lead to obvious environmental benefits, such as natural resource preservation and waste reduction, which are not expressed through calculated impact categories.

Blegnini (2009) compared two different end-of-life scenarios for an actual building in Turin (Italy): complete recycling of waste (except for a small amount of common waste, which was landfilled) and complete landfilling. He performed LCA of the whole life cycle of a building, with special attention paid to the demolition and recycling phases. For these phases, analysis was based on the actual, field-measured data. Since the building was reinforced-concrete framed, waste consisted mostly of concrete (83%) and steel rebars (4%). The obtained results showed that the reductions in the recycling option were small in comparison to the whole life-cycle impacts (0.2–2.1%, depending on the impact category). When the comparison was restricted to the pre-use phase (material production and construction phases), all the impacts in the landfill option were higher than those in the recycling option, an increase ranging from 17 to 54%, depending on impact category. The impacts from recycling were calculated as the difference between avoided impacts (avoided primary NA production) and induced impacts (recycling process and transportation). All the credits from avoided NA production were therefore allocated to the system that produces waste. Avoided landfilling was not considered.

Despite using different approaches within LCA methodology, most of the research published so far indicates that recycling of concrete waste presents environmental advantages over landfilling. However, special attention must be paid to transportation distances, which can make a substantial difference. For buildings, the impact of the end-of-life phase is small compared to the total life-cycle impacts. Nevertheless, with the development of energy-efficient buildings and the use of less polluting energy sources, the contribution of materials production and end-of-life phases is expected to increase in future.

23.7 Conclusions and future trends

It is impossible to draw general conclusions on the eco-efficiency of RCA compared to NA, either as aggregate in structural concrete or as material for road base, since there are too many parameters that can affect the results. Each case is specific and must be considered separately. However, in both the specific case studies presented, it was found that concrete recycling can bring environmental benefits. This depends on the ratio of transport distances of RCA and NA, and the location of the recycling plant is therefore of crucial importance. With an appropriate ratio, the environmental impact of RAC and NAC (or RCA and NA) production in terms of the studied impact categories can be the same, and the benefit of recycling in terms of minimizing waste and natural mineral resource depletion is clearly gained. Generally for specific aggregate and concrete production technologies, it is possible to determine the transport distances at which RCA and/or RAC are environmentally beneficial over NA and/or NAC.

When considering methodological aspects, it is particularly important to adopt a standardized methodology when applying LCA to the end-of-life phase, such as different waste management scenarios within the whole life cycle. This is especially important when considering the impacts from recycling. Otherwise, it is not possible to compare the results of different researchers and the whole field of research loses its consistency. Recycling is a multi-functional process, always connected to at least two products. It is not consistent with LCA methodology to consider the recycling of a product without taking into account how it affects the next product in the chain, even if this means that a simple cut-off rule is applied. Following LCA methodology, and taking into account Fig. 23.1b (open-loop recycling), the treatment of the end-of-life phase should, strictly speaking, be as follows:

- Landfilling belongs to Product 1, i.e. to the product that generates waste. So, if part of the waste is recycled, Product 1 is credited with avoided landfilling.
- Material production in the recycling process belongs to Product 2, i.e. to the product that receives recycled waste. So, if the recycled waste of Product 1 is used in the life cycle of Product 2, Product 2 is credited with avoided NA production.
- The impacts from the recycling process and corresponding transportation is allocated between these two products, following some of the allocation rules (economic, mass allocation or cut-off).

Product 1 is thus credited with avoided landfilling and with the fact that only part of the recycling impacts is allocated to it. Product 2 is credited with avoided NA production and also with the fact that only part of the recycling impacts is allocated to it.

The second problem is how these credits from the recycling process are accounted for in LCA. Avoided landfilling and avoided NA production should be taken into account through appropriate impact categories with corresponding indicators (landfill capacity depletion and natural resources depletion) and not by subtracting the avoided from the induced impacts of recycling. This is because such an approach represents the real impact on the environment. The real impact from recycling is equal to real energy consumption and real emissions of various pollutants (so-called induced impacts), but will also result in the saving of a certain amount of landfill space and NA.

Unfortunately, most of the proposed methodologies neither include solid waste production/landfill capacity as an impact category, nor consider sand and stone as abiotic resources that can be depleted. Besides the fact that sources of quality sand and stone for aggregate production are not endless, especially at regional level, their unlimited extraction also has a strong impact on the environment and leads directly to the devastation of local natural environments, whether the aggregate is crushed stone or river aggregate. Landfill capacity is also becoming a scarce resource nowadays, in many countries. Future research should be aimed towards development of special indicators for the depletion of natural bulk resources that are used in concrete production (i.e. sand and gravel) (Habert *et al.*, 2010) and landfill space.

23.8 Acknowledgement

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24

Assessing the potential environmental hazards of concrete made using recycled aggregates (RAs)

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Abstract: This chapter starts with widely recommended and used methodologies for environmental assessment, with an overview of their application to construction materials. The concept of pollutant emission by leaching and its dependence on the materials involved is introduced, along with the parameters influencing the leaching process. The manner in which water contact with materials causes transfer, transport and dispersion of the contained contaminants is explained. Then the most relevant experimental tools dedicated to hazard identification and environmental performances are presented. These tools comprise laboratory leaching tests whose suitability for future standardisation in the field of construction materials is considered. Ecotoxicology tests could potentially be adapted to this standard testing method. The second part discusses recent research and discoveries concerning the leaching properties and potential hazards of materials of concern. The chemical behaviour of several pollutants in cement matrix is explained and examples of results obtained by leaching studies on concrete materials are discussed. Examples of studies on recycled aggregates (RA) from demolition as well as new concrete materials containing these wastes are also presented.

Key words: recycled aggregates (RA), concrete, risk assessment, leaching test, toxicity test.

24.1 Introduction

The analysis of the life cycle of concrete-based building materials highlights high energy consumption, greenhouse gas emissions and generation of significant amounts of waste during the construction and demolition stages. The appropriate re-use of these materials will help to reduce the environmental impact, for example by reducing the extraction and processing of mineral resources and therefore the energy consumption associated with these activities (extraction, transport, manufacturing etc.), reducing the greenhouse gas emissions and the production of wastes. Although the value of re-using these materials is obvious, the process has to be strictly controlled and monitored to eliminate risks, particularly those to human health.

Within the European Union, Regulation Number 305/2011:

> . . . lays down harmonised conditions for the marketing of construction products
>
> (EU, 2011)

The basic requirements for construction also encompass those for building materials (Annex I of the Regulation). Importance is given to the health, hygiene and environmental impacts of construction projects and to the sustainable

> use of natural resources, including the use of environmentally compatible raw and secondary materials in the construction works.

The Technical Committee CEN/TC 351, 'Construction products: Assessment of release of dangerous substances' of the European Standardization Committee works to standardise the assessment of building materials for re-use. One of the problems under consideration is the release of pollutants from the construction materials into water, soil and air by different mechanisms. Currently a global methodology to solve this issue is being proposed. Its further application to typical environmental scenarios will lead to changes in the processes of manufacturing and management of building materials and similar practices.

The first part of the chapter presents widely used and recommended methodologies for environmental assessment, with a specific focus on their application to construction materials. The concept of pollutant emission by leaching and its dependence on the utilisation scenario is introduced. An explanation is given of the ways in which water contact with materials causes transfer, transport and dispersion of the (previously contained) contaminants and of the range of parameters influencing the leaching process. Then the most relevant experimental tools designed to assess hazard identification and environmental impacts are presented. In particular laboratory leaching tests for future standardisation in the field of construction materials and the potential for adapting ecotoxicology tests for standard testing are considered. The second part discusses recent discoveries regarding the leaching properties (and, by extrapolation, the potential hazards) of materials of concern. The chemical behaviour of several pollutants in the cement matrix is explained and examples of results obtained by leaching studies on concrete materials are discussed. Examples of studies on recycled aggregates (RA) from demolition and new concrete materials containing these wastes are presented.

24.2 Methods for assessing the potential hazard of construction materials and wastes

24.2.1 Scenarios of pollutant emissions

During the service life of construction materials, the main hazard potential is posed by contact with water, the major pollution vector in the environment. When waste comes into contact with water or aqueous solutions, that contact results in the phase transfer, transport and dispersion of the contained contaminants. Leaching mobilises various substances which, in turn, have an impact on the quality of the soil, the water, the biosphere and human health via direct contact and through ingestion of water and food.

Construction materials are exposed to natural waters (meteoric, surface and underground water) in different ways. These depend on the structure of the material itself and on the water circulation at the surface and inside the material. Several typical scenarios have been identified (Schiopu *et al.*, 2007):

1. **Sloping plane**: corresponds to a slope steeper than 15% for which the main water contact is due to runoff of water.
2. **Horizontal plane**: e.g. materials for terraces or public works. Meteoric water in contact with horizontal surfaces can:
   - (i) form stagnant layers;
   - (ii) run off;
   - (iii) infiltrate the porous material; and
   - (iv) percolate, if the material is macroporous.
3. **Vertical plane**: corresponds to a façade exposed to rain water. Meteoric waters run off and, to a certain degree, infiltrate if the material is porous.
4. **Contact with soil**: corresponds to buried foundations for civil works and earth works. All water contact types are possible: runoff; infiltration; percolation; stagnation.
5. **Completely immersed in water**: e.g. dykes, piles and basins. These are usually made of concrete based materials.

24.2.2 Leaching mechanisms and parameters of influence

Leaching is a complex process. The main mechanisms involved are chemical reactions coupled with transport/transfer phenomena. Chemical reactions and transport of dissolved species are dynamic processes, which produce structural changes such as modification of the porous structure, modification of the chemical composition of the material and the release of undesirable chemical species.

At the liquid–solid interface (the apparent material surface, pores or anywhere a liquid phase is present) different interactions occur. These include dissolution–precipitation, adsorption–desorption, complexation and bio-chemical reactions. The interactions that occur depend on the chemical composition of the system components and on the presence of exogenous chemical species, such as gases (e.g. CO_2, O_2) and dissolved compounds in the natural waters.

The dispersion of the dissolved chemical species (pollutants) is a combination of physical and chemical processes. It takes place by different transport mechanisms:

- **Diffusion**: dissolved species inside a porous material diffuse toward the apparent surface of the block when a concentration gradient exists in the liquid phase. Diffusion is the main transport mechanism for stagnant liquids in pores, in the case of concrete blocks;
- **Convection**: the movement of a liquid volume containing the dissolved species. Convection takes place at the surface of a piece of material (runoff) and/or in the pores (percolation) if the porous structure is permeable. The latter is a typical leaching mechanism for granular materials.

The type of leaching behaviour and the extent of the pollutant release depend on various parameters affecting the chemistry and transport mechanisms. Examples of these include:

- **Water/solid ratio and contact surface**: this determines the intensity of the chemical interactions and the mass-transfer of pollutants. For example, release from a granular material is more important than from a block of the same mass (until liquid saturation). When the transport phenomena are not limiting, the only variable that can limit leaching is the solubility of the target species in the respective chemical context. In this case, various liquid to solid ratios (e.g. L of leachate/kg of material) release a certain mass of pollutant, dependent on a limiting concentration. This concentration is thermodynamically determined by the equilibrium constants, temperature and solution ionic strength and/or by the reaction kinetics.
- **Time**: water contact is usually not continuous but intermittent, depending on the climate and scenario. Generally similar events take place periodically, for example, the rainy periods in a season. The effects of the water contact duration and sequencing on the pollutant release are difficult to foresee and have yet to be researched.
- **Temperature and humidity (climate)**: temperature influences the chemical reactions that occur in the water as well as its physical state. The material humidification and saturation favour dissolution and diffusion of substances. Material drying determines the migration of water from deep pores toward the material surface and precipitation of dissolved species. Freezing affects the structure of the porous material by increasing the effect of temperature and humidity cycles. Temperature and humidity variation can cause mechanical and chemical changes, including material swelling or cracking.
- **Material ageing**: the modification of the chemical, physical and mechanical structure of the material due to functioning stresses and natural factors (leaching time, UV, temperature, humidity, etc.). During leaching, the pollutant release dynamics vary with time. Because the porous structure and the chemical composition of the material itself change.

24.2.3 Methodologies for health and environmental impact, and risk assessment

Specific methods and tools have to be used to assess the harmful effects of pollutants on humans and the environment. All the existing methods are based on two general methodologies:

1. **Health Risk Assessment** (HRA) for humans, described in a series of documents available from national organisations for health surveys (EPA, 2011); and
2. **Ecological Risk Assessment** (e.g. ERA guide published in EPA, 1998).

Both methodologies comprise several steps, some of which they have in common. The main steps of HRA are:

1. Hazard identification: type of problem the pollutant can cause;
2. Exposure assessment: the length of time, concentration or dose of the pollutant to which the person is exposed;
3. Dose-response assessment: how the pollutant quantitatively affects the person;
4. Risk level: calculation of the risk that the exposed persons incurs.

The ERA methodology is built on:

1. Exposure assessment.
2. Assessment of the pollutant effect on living organisms.
3. Risk level.

Two major aspects must be investigated, whatever the case study:

1. the characterisation of the effect a given substance can induce on a given organism (HRA steps 1 and 3 and ERA step 2);
2. the exposure assessment (HRA step 2, ERA step 1).

The latter concerns the way the pollutant is emitted by the source (here, the concern is construction materials in service life scenarios) and transported through different environmental compartments to the living targets. Knowledge of the pollution source and its dynamic behaviour over the period of concern is of key importance for a reliable assessment.

In practice, simplified methods are used, adapted from the HRA and the ERA for the sake of cost and time savings. Previously, several simplified methods have been proposed for assessing the eco-compatibility of re-using waste as a secondary material in earthworks and road construction (Petkovic *et al.*, 2004). The principle of these simplified methods is to determine, by taking measurements, experiments and modelling, the exposure model, that is the time dependent pollutant concentration in a given compartment (soil, sediments, groundwater, surface waters, etc.). Then the contaminant concentration is compared with acceptable limit values for a given substance and its harmful effects. The limit values are taken for each hazardous pollutant from toxicology and ecotoxicology databases or are already specified in related regulations (i.e. the water directive relating to water intended for human consumption).

The aforementioned methodologies require experimental and modelling tools in order to determine:

- the intrinsic properties of the material: chemical and mineralogical composition, porosity and pore size distribution;
- the variables affecting each leaching scenario;
- the evolution in time of the leaching characteristics.

A complex iterative validation process then has to be implemented.

24.2.4 Leaching and ecotoxicity tests

The information on material–water contact conditions and their influence on pollutant release may be obtained from appropriate leaching tests at laboratory scale and experiences and monitoring at pilot or field scales. In order to obtain reliable and reproducible information on the leaching process and pollutant emission, the leaching tests must be performed in the standard defined and controlled conditions.

Standardised leaching tests for wastes have existed for decades in different countries and more recently at EU level. Generally, only the issue of mineral contaminants are addressed. These tests have different objectives, for example:

- **To compare leachate composition to regulatory thresholds**. The compliance tests (e.g. EN 12457.1 to 4) concern four different but close and simple leaching protocols on materials of different granulometries.
- **To characterize the leaching behaviour of waste**. These tests have been developed in order to understand the release mechanisms. The main leaching tests developed at the European level are presented hereafter; these tests are expected to be adapted for construction materials. Progress is still necessary in the assessment of the leaching behaviour of organic pollutants, alone or in inorganic contaminants, when different effects, such as synergy or attenuation can occur.

The chemical behaviour of the material and pollutant solubility is investigated in steady state conditions, such as at liquid/solid chemical equilibrium. Two tests have been developed to satisfy this objective by similar principles. Each of them aims to reach steady state in solid–liquid processes at a controlled pH, for different pH values (CEN TS 14429 'Influence of pH on leaching with initial acid/base addition', CEN TS 14997 'Influence of pH on leaching with continuous pH-control'). In order to facilitate the equilibrium state, specific procedures are put in place: fine crushing of the material; efficient stirring; negligible atmospheric

CO_2 uptake; long contact time. The system pH is controlled by adding different exogenous reactants such as nitric acid. The liquid composition obtained at different controlled pH values is specific for a given solid matrix and provides useful information about the pollutant speciation and solubility.

CEN TS 14405, 'Up-flow percolation test (under specified conditions)', is specifically designed for the investigation of the pollutant released by percolation of water in a column filled with granular material, by means of leachate time-dependent composition analysis. The leaching of 'monolithic' blocks at laboratory scale allows the study of leaching behaviour in the context of the solid material, at a fixed leachate volume–surface of the sample. The leachates are collected after a certain time and analysed. Two variants of this test exist, CEN TS 15863 and CEN TS 15864. These are differentiated by the protocol for the leachate renewal (periodic flow and continuous flow of leachate respectively for the two variants). These tests facilitate calculation of the fluxes of released pollutants and the assessment of the leaching dynamics.

The Toxicity Characteristic Leaching Procedure (TCLP), EPA Method 1311 (EPA, 1992) is an 'arbitrary' leaching test designed to investigate the mobility of organic and inorganic analytes. The extraction fluid is set between a buffered solution of $CH_3COOH/NaOH$, pH 4.93 and a CH_3COOH solution, pH 2.88, in order to simulate landfill leaching. The process is run under rotary agitation for 18 ± 2 h, the eluate filtered and the concentration of 40 possible contaminants analysed. If the concentration of any of the 40 target contaminants exceeds the established regulatory limit, the waste possesses the characteristics of toxicity. If that is the case, it has to be treated adequately prior to landfill disposal.

Although standard experimental tools exist for the leaching assessment of waste and waste containing materials issued from waste stabilization-solidification processes, there are currently no standard experimental tools for the evaluation of construction materials. Establishing such experimental tools could be difficult due to very low supposed pollutant concentrations, long testing times and a combination of bio-physico-chemical processes. Research has been performed in some cases, but it is too early to form general conclusions or standard testing methods. Several regulations are in the process of being drafted, concerning the experimental tools for assessing the release of dangerous substances from construction materials. The Technical Committee (TC) 351 of the European Committee of Normalization (CEN) introduced the concept of a horizontal testing procedure (CEN/TR 16098, 2010). This might form a common methodology for testing any substances and construction products covered by the Construction Products Directive. At the European level, the working group CEN/TC 351/WG1 is mandated to study the 'release from construction products into soil, ground water and surface water' and to propose the appropriate experimental (horizontal) leaching tests. The emerging leaching tests (and corresponding EU standards) could be a monolith leaching test and an up-flow percolation test, similar to those already used for waste characterization.

The toxic or ecotoxic properties of certain materials are usually determined by bioassays composed of experimental tools for measuring the pollutant effects on living organisms. The bioassays can be applied at different steps as part of a risk assessment. In very upstream steps, the bioassays could be applied to materials to estimate an intrinsic toxic or ecotoxic potential. However, the results are only relevant for the laboratory conditions used in the tests. For real toxic or ecotoxic effect to be determined, the actual conditions that influence the chemical and leaching behaviour of the exposed material and the target leaving species would have to be used. The experimental conditions of the bioassays must therefore be as effective as possible in simulating the real scenario.

Currently, there is neither regulation nor guidance for how the bioassays must be performed on construction materials. Generally, the ecotoxicity assays can be performed on different media (solids, liquids) using the appropriate leaving species sensitive to the studied contaminant. The experimental conditions (dilution, temperature, light, culture medium, etc.) are controlled in order for the test to be specific to the leaving species and the monitored effect (mortality, growth, reproduction, luminescence, etc.). In the case of a chemically complex matrix (solid or liquid), effects such as synergy, antagonism and bioavailability can make the test results difficult to interpret. In construction materials, many chemical parameters (heavy metals, metalloids, extreme pH values, high salinity of leachates) contribute to the bioassay response. It is therefore difficult to correlate the chemical composition and structure of the material with the ecotoxicity test result.

There are numerous standard ecotoxicity tests at international or national levels. These were developed initially for chemical substances, and then for different matrices such as industrial waste waters and polluted soils. A lot of documentation on toxicity and ecotoxicity tests is available on the web sites of regulating organisations such as the OECD (Organisation for Economic Cooperation and Development (OCDE, 2011)), ISO (International Organization for Standardization (ISO, 2011)) and US EPA (United States Environmental Protection Agency (EPA, 1994)). The direct application of these tests on wastes and materials containing contaminants (on solid samples or on leachates) raises the question of the significance of the experimental conditions chosen, especially the dilution and the leaching duration, and consequently of the relevance of the ecotoxic result and its interpretation.

24.3 Pollutant emissions from concrete materials

The potential hazards of concrete material containing RA will be determined by the chemical composition and properties of different components, such as cement or conventional concrete and the used RA. RA are mainly composed of concrete and other masonry residues. They are supposed to have similar or close chemical properties to the original materials. In this section, the leaching properties of conventional concrete and the behaviour of pollutants in a cement matrix are considered.

The chemistry of pollutants incorporated in cement or concrete materials is mainly affected by the very alkaline medium generated by the hydraulic binder, which is comprised of alkaline species, portlandite and hydrated calcium silicates. The most studied chemical species with potential hazardous effects are the heavy metals, such as Cr, Cd, Pb, Cu, Mo, V, etc. These exist in the raw materials used for cement fabrication (e.g. residues with high calorific potential, i.e. oil, tyres, plastics or composite materials), but also in the natural aggregates (NA) used in concretes. Another source of pollutants in concrete is the use of contaminated RA or various residues such as slags, ashes from various thermal processes, sludge, etc.

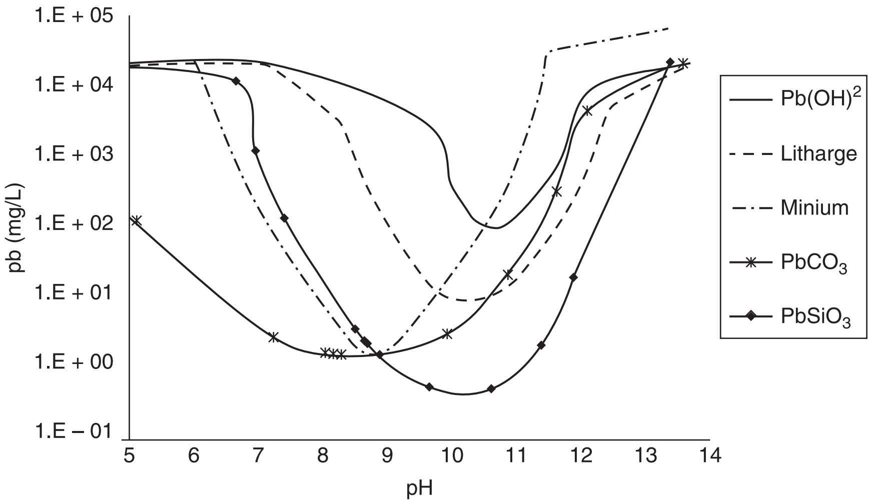
The stabilization of wastes is one standard method to reduce the mobility of certain pollutants. The resulting materials will either enter controlled landfills or their harmful character will be put to a suitable use, depending on their harmful characteristics. The issue of pollutant release arises from the use of these cement-based materials under real conditions. Studies performed over the last few decades have shown that two main chemical processes contribute to pollutant fixation in a cement matrix:

1. fixation of metals as hydroxides or oxides during the material hydration under very alkaline conditions (if the waste is added in the hydration process);
2. incorporation of metals into the structure of silicates and alumino-silicates by substitution of generic ions.

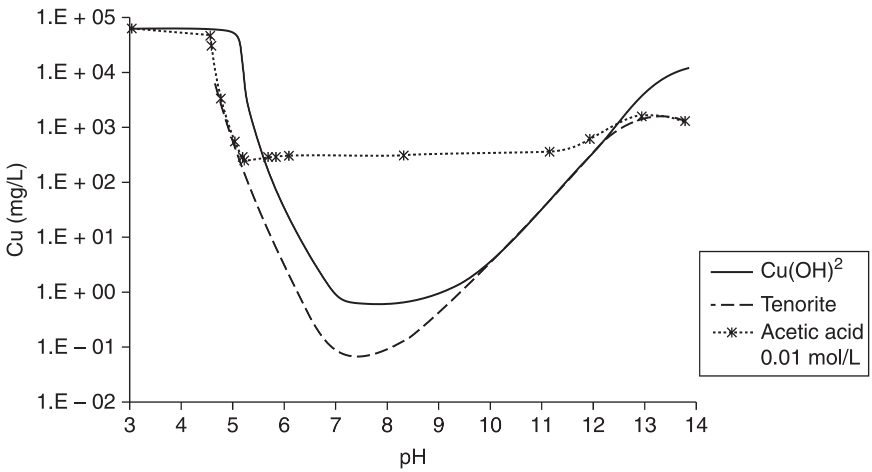
For the most toxic elements, several types of chemical behaviour are distinguished.

1. The amphoteric property of some heavy metals (e.g. Pb, Zn) should increase the metal solubility, because of the high alkalinity of the pore water. Fixation of these elements is then strongly dependent on the material pH. For example, lead is an amphoteric element whose solubility is minimal at neutral pH and increases by many orders of magnitude in acid and alkaline conditions. Figure 24.1 shows the solubility of different solid phases as a function of pH. These phases are all likely to control Pb release in cement matrices, but the actual speciation (controlling phase) is not fully elucidated. Furthermore, in contact with a solution, the less soluble phase will control the metal solubility. It has been observed that for the alkaline pH of the pore water of cement matrix (over pH12), the solubility is not at a minimum and therefore that a cement matrix is not the best stabilization method for this metal.
2. The presence of organic matter (e.g. from additives or unburned organic matter) enables the mobilization of some elements by complexation. This is especially the case with Cu but also other elements such as Ni and Zn. Figure 24.2 shows the level of solubility of Cu as a function of pH as imposed by mineral phases such as oxides and hydroxides, but also in the presence of organic matter. The nature and concentration of soluble organic matter in a concrete matrix and/or in a leachant controls the release level.

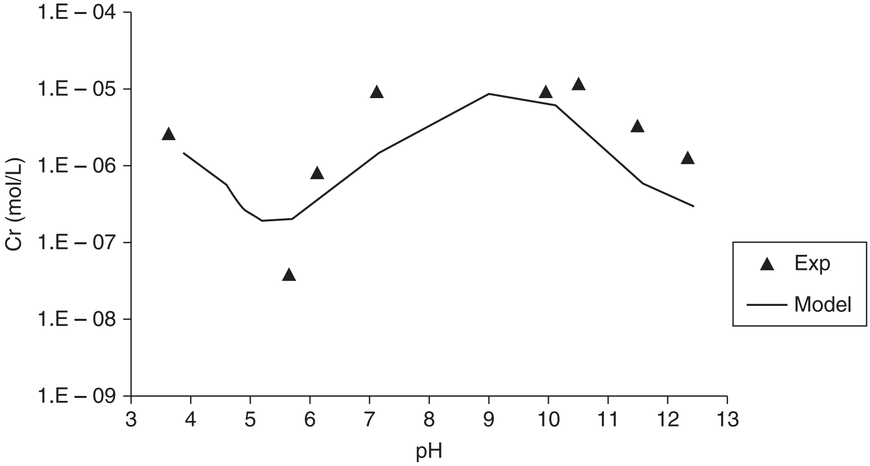
3. Anionic forms of elements such as Mo, Cr(+6), As, B, V and SO_4^{-2} have a tendency to be leached at mild alkaline pH (about 7–9) and to be fixed at pH >12. Sulphates are fixed by ettringite. The other anions could also be fixed by sulpho-aluminate phases. Besides pH, the concentration of accompanying cations (Ca, Ba, Sr) is also a determinant in anion fixation.



24.1 Solubility curves of different lead-based solid phases (in case of $PbSiO_3$, a saturated liquid in silica was considered). Results obtained with PHREEQC® and MINTEQ data base. Available from: *http://www.brr.cr.usgs.gov/projects/GWC_coupled/phreeqc/*



24.2 Solubility curves of different copper-based phases alone and in presence of organic matter (tenorite in presence of acetic acid 0.01 mol/L). Simulations with PHREEQC® and MINTEQ.



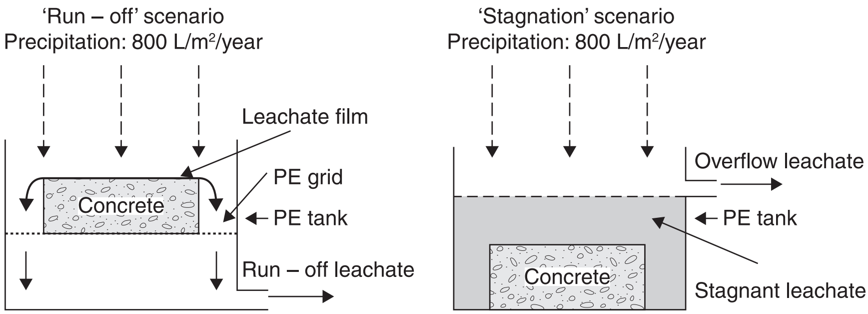
24.3 Typical pH-dependent behaviour of chromium in a cement matrix (experimental and simulation data from Schiopu *et al*, 2009).

Figure 24.3 shows Cr release in a pH-dependent leaching test conducted at different pH values. This experimental result can be interpreted by two main fixation mechanisms: Cr-ettringite in alkaline conditions (pH>11) and Cr fixation on iron hydroxides at pH 5 to 7.

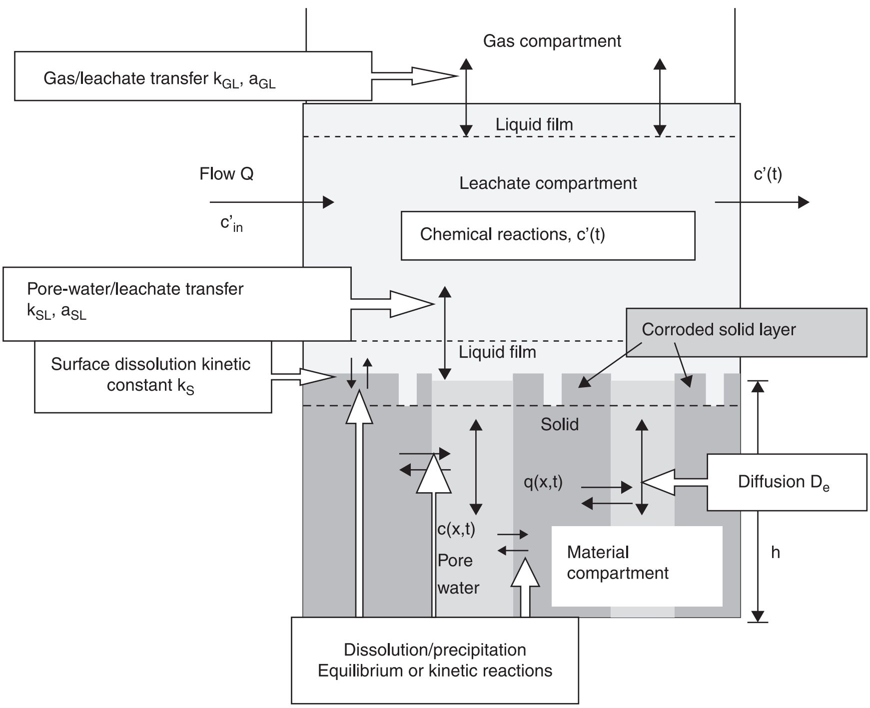
24.3.1 Conventional concrete materials

Environmental assessment of concrete materials fabricated with conventional raw materials (without RA or wastes) is a recent issue and few studies have been reported as yet. The leaching behaviour and potential hazard of conventional concrete materials is determined by the type and origin of cement used. Laboratory tests along with long-term, scaled-up experiments on concrete leaching are necessary for a relevant evaluation of potential hazards in real scenarios.

The study presented in the following was conducted on a commercial construction product (Schiopu *et al.*, 2009). Laboratory leaching tests and field pilot-scale tests were investigated. The concrete paving slabs manufactured with CEM I-concrete were exposed for one year in two types of outdoor leaching scenarios, 'runoff' and 'stagnation' (Fig. 24.4), and the released elements were monitored. The leachate analysis results were then used for the validation of a leaching model composed of the geochemical model of the material coupled with appropriate transport models for each scenario. Figure 24.5 presents the conceptual model. Three compartments are considered, each of them defined by using certain chemical reactions and transport phenomena, and linked to each other by mass transfer processes. This model is able to simulate the leaching behaviour of major elements, as well as of trace pollutants such as Cr and Cu.

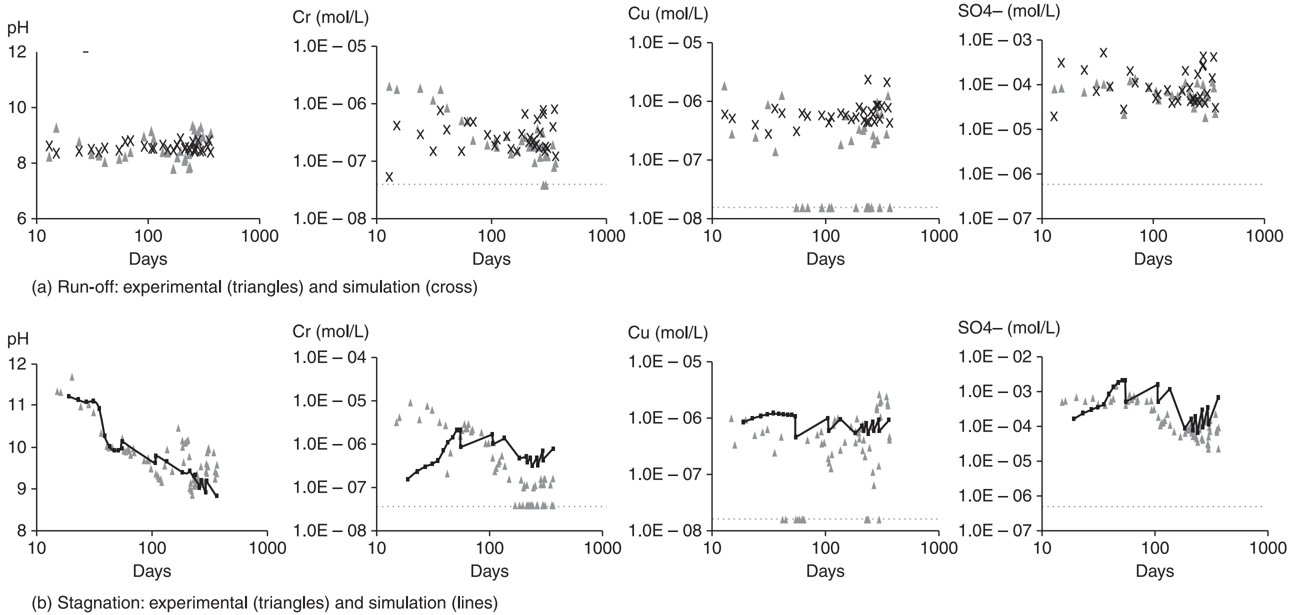


24.4 Leaching scenarios for cement based materials.



24.5 Conceptual model of the leaching process.

Figure 24.6 shows the concentration of the leachate in contact with the concrete slab in real conditions. The pH is different, depending on the water contact type. The concentration of heavy metals is very low and the concentration-to-time curve shows different shapes, depending on the element's chemical properties. For example, Cr concentration drops over 100 days and corresponds to threshold values for good-quality ground water (according to French regulations for water quality assessment).



(a) Run-off: experimental (triangles) and simulation (cross)

(b) Stagnation: experimental (triangles) and simulation (lines)

24.6 Pilot results for concrete slabs leaching in outdoor conditions.

24.3.2 Examples of pollutant behaviour in the cement matrix

Many research studies have investigated the behaviour of concrete materials containing heavy metals. In these studies, concrete model materials were prepared with a controlled composition (type of cement, sand, water and their proportion) by adding spiking substances containing the target metal (e.g. PbO, $Pb(NO_3)_2$, CdO, $ZnCl_2$) or real wastes. More complex concrete materials containing waste as aggregates or additives (municipal solid waste incineration, foundry slags, different sludge, coal ashes, etc.) have been investigated and the results written about extensively. In the following example, three different hydraulic binders were employed to obtain three model concrete materials, spiked with lead at a total content of 10 g per kg of material (Imyim, 2000; Tiruta-Barna *et al.*, 2004). Their composition is given in Table 24.1.

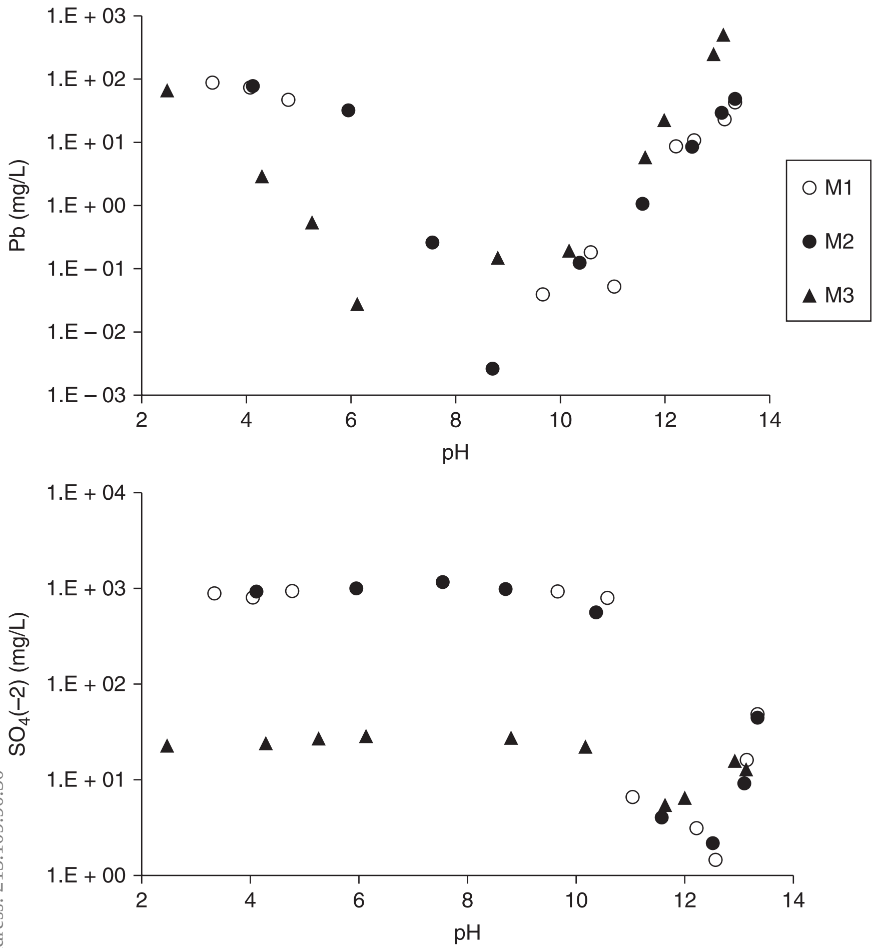
The equilibrium pH dependency test shows similar trends for lead in the studied materials (Fig. 24.7), characteristic of its amphoteric behaviour. Lead silicate seems to control the solubilisation level. Within the chemical context of the concrete materials, such as native pH of 12.5 for M1 and M2, and 12 for M3, lead is not optimally fixed. In the case of sulphate, M1 and M2 show identical behaviour over the entire pH range. At pH>11, all materials show similar trends corresponding to ettringite solubility. Ettringite no longer exists at pH<10 and consequently sulphate solubility increases. Its concentration reaches a plateau that gives an estimation of the total available sulphate content. M3 releases less sulphate, due to its lesser total content, since no sulphate was added in its preparation (Table 24.1).

Other studies reported similar observations. Four types of materials were compared through the results of the pH dependent leaching test (van der Sloot, 2002). These materials were: a mortar prepared with regular Portland II B cement;

Table 24.1 Composition of model materials

		% Dried mass of material		
		M1	M2	M3
Binders	OPC CEM I	25		
	OPC CEM II		25	
	Coal fly ash			15
	CaO			3.00
Additives	NaCl	0.25	0.25	
	$CaSO_4$	0.50	0.50	
	PbO	1.07	1.07	1.07
	Standard sand	73.17	73.18	80.93
Water	Water/cement ratio	0.45	0.50	0.90

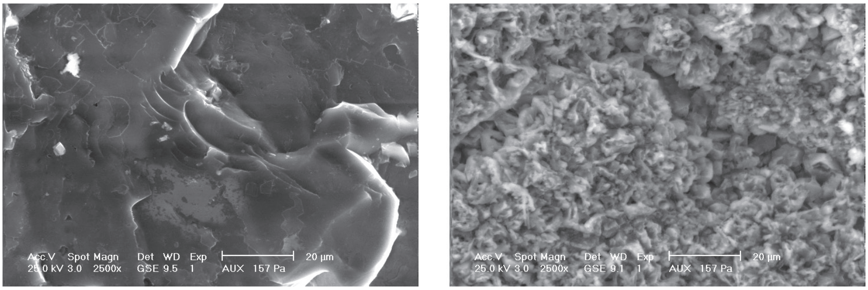
Source: Tiruta-Barna *et al.*, 2004.



24.7 Lead and sulphate behaviour in pH dependent test, for three model materials.

a mortar prepared with cement obtained with alternative fuels and raw materials; a mortar containing MSWI-fly ash in low proportion (<10%); and a mortar containing two stabilized MSWI-fly ashes (>80%). Observation of solubility as a function of pH and the use of geochemical modelling confirmed the cited fixation mechanisms and resulted in the conclusion that the same solid phases control the solubility of a given pollutant in different materials, with very little exception in the case of high waste content (i.e. a low cement to paste proportion).

Besides the material composition, the leaching conditions (particularly, the water contact mode) represent an important variable parameter with a significant influence on the pollutant behaviour.



(a) (b)

24.8 Environmental SEM view of the sample surface before leaching (a) and after leaching (b) with demineralised water.

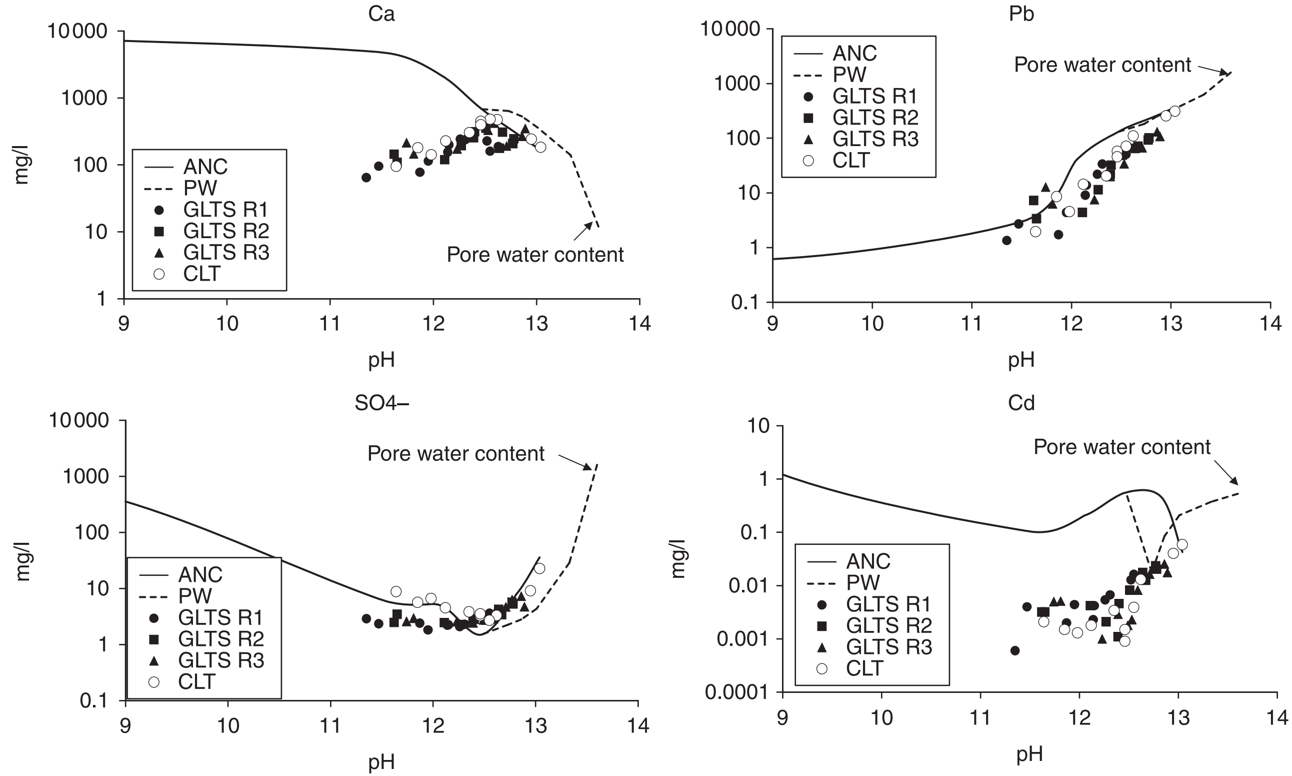
As an example, several results are presented, obtained by different leaching tests on the same cement-based model material, composed of 30% Portland cement (PC) (CEM I), 60% siliceous sand, 5%wt PbO, 5%wt CdO, at a water to cement ratio of 0.4 (Barna *et al.*, 2005). Lead and cadmium behaviour has been studied in laboratory leaching tests. As observed in Fig. 24.8, leaching degrades the material surface. The progression of the 'degradation front' depends on the intrinsic properties of the material and of the exposure conditions.

Distinct assays were conducted on the same 'model' material, in order to observe the mechanisms that control species release. These included a pH dependent solubilisation test (noted ANC in Fig. 24.9 – Acid Neutralization Capacity), assessment of the pore composition (noted PW in Fig. 24.9), monolithic leaching test, and granular material leaching with demineralised water under continuous flow and/or sequential renewal. The following graphs (Fig. 24.9) show the concentration of the eluate represented as a function of pH. Each graph gives:

- the pH dependent concentration (continuous line – ANC);
- the simulated pore water content of the species (dotted line);
- leaching tests on granular material in continuous stirred reactors with sequential renewal of the eluates (GLTS, analogous to the Dutch tank leach test, in which there were 3 renewals, denoted by R1, R2 and R3);
- leaching tests of granular materials with continuous flow using a column test (CLT).

As observed in Fig. 24.9, the pH of the eluate decreases during leaching from about 13 to 11.3, due to the depletion of the alkalinity of the material itself and to the high mobility of the alkaline species. The release of Ca is therefore controlled by the pH of the eluate, until values become lower than 12.5.

Lead and sulphate releases are controlled during the test by their pH dependent solubility with respect to the leachate, while Cd release seems not to be controlled



24.9 Eluates concentration versus pH for four leaching tests applied on a 'model' concrete material.

by the eluate pH. This behaviour can be explained by the existence of a transport limitation through the porous system and by the conditions of the solubility test, namely, the presence of nitric acid in an ANC test rather than only deionised water in dynamic tests.

These results illustrate the necessity of finely adapting the experimental protocols for the characteristics of the materials and the contained contaminants in order to assess leaching behaviour as it would actually occur in the environment. In some conditions, for example when saturation occurs, the opposite processes (release and uptake from the eluate of the same species) take place with the same kinetics. In these cases, the rate of mass accumulation in the eluate tends to be annulled. Therefore, the direct extrapolation from a laboratory experiment to a different up-scaled scenario would be erroneous.

Studies performed by different authors on different cement-based materials containing real wastes have shown similarities in the chemistry of the pollutants. Many leaching studies (outside the subject matter of this chapter) have been performed on solidified/stabilized wastes, aiming to assess their environmental effects on disposal or re-use scenarios.

24.4 Recycled aggregates (RAs): properties and intrinsic potential hazards

Waste aggregates resulting from construction and demolition activities have the chemical composition and properties of the original materials. In certain cases, contamination with toxic materials or substances can exist, for example in the case of painted masonry (e.g. lead-based paint), contact with asbestos, presence of biocides (especially from wood products) or when other compounds have been used as additives, such as glues, solvents and resins. Concrete is a major constituent of the aggregate category. The most used waste aggregates incorporated in concrete fabrication (the so-called 'recycled aggregate concrete') come from masonry (bricks), concrete works and public works (pavements, roadbeds).

Limbachiya *et al.* (2007) studied the chemical composition and mineralogy of coarse recycled concrete aggregates collected on three construction sites and conditioned (crushed and sorted) by a recycling plant. The aggregates (size 4–16 mm) were intended for concrete production. Spectroscopic methods were used for determining mineralogy and chemical composition. Several potentially hazardous species were analysed and compared with the PC composition. For example, Pb 30 to 50 ppm (19 ppm in PC), Zn 31 to 37 ppm (53 ppm in PC), Cr 21 to 52 ppm (42 ppm in PC) and V 42 to 62 ppm (134 ppm in PC). It was concluded that these elements are absent or very scarce in the employed NA (rocks). The study shows that PC and recycled concrete aggregates contain comparable orders of magnitude of such elements. Concerning the major elements and phases, quartz, feldspar and calcite were identified. A water extraction test was performed using a liquid-to-solid ratio of 50 ml/g and 1 h contact time. In these conditions, the pH of leachates ranged from 10.4 to 11.2, and the concentration of the elements mentioned above ranged from 0.01 to 0.08 ppm. Notably, the concentration was 0.14 ppm for Pb. Cr was totally solubilized, but its content in aggregates was judged as relatively high.

A similar study was performed (Bianchini *et al.*, 2005) on mixed masonry (bricks and terracotta) and concrete aggregates with close mass fractions (some debris of wood, paper, asphalt, plastics and metal did not exceed a total of 10 wt%). The most abundant solid phases identified in aggregates were typical for cement materials such as quartz, calcite, dolomite and, to a lesser extent, feldspars, muscovite/illite chlorite and hydrated calcium silicates (CSH). Typical phases for bricks and terracotta, such as gehlenite and wollastonite, were identified. Concerning the heavy metals, their content in the analysed solids showed a variable composition, depending on the size fraction of the aggregates, with a small particle fraction (<0.125 mm) more contaminated than the coarser metals. The concentrations of several metals were of the following order of magnitude: Pb: 25 to 171 ppm, Zn: 34 to 220 ppm, Ni: 21 to 107 ppm, Cr: 25 to 166 ppm and V: 29 to 86 ppm.

These studies demonstrate that the aggregates have a mineralogical composition close to the original materials. This could suggest that their leaching behaviour, due to their chemical properties, and the potential hazards caused by leaching are close to those of cement-based materials. Engelsen *et al.* (2009, 2010) studied the chemical structure and leaching behaviour of recycled concrete aggregates using the pH dependent leaching test CEN/TS 14429 (an equilibrium contact solid–liquid test conducted at different pH values). The waste samples collected from a recycling factory contained more than 90% concrete, masonry and NA and less than 5% asphalt debris. The samples were originally from building and road demolition sites. These were then compared with laboratory-prepared aggregates obtained from common concrete, including PC CEM I, NA and common additives for concrete.

Investigations on solid composition show that the wastes were mainly constituted of quartz, feldspars (albite, microcline), phyllosilicates (chlorite from NA), portlandite and calcite at different levels of carbonation degree. The cement paste (soluble in acid, e.g. portlandite, C-S-H, calcite) content in all samples was estimated at 12 to 18 wt%, compared with 18% for lab-prepared concrete. The concentration of heavy metals in the waste samples ranged from 31 to 61 ppm in Pb (12 ppm for concrete), 45 to 553 ppm in Zn (53 ppm in concrete), 69 to 116 ppm in Cr (49 ppm in concrete), 53 to 84 ppm in V (35 ppm in concrete), 12 to 43 ppm in Cu (24 ppm in concrete), 14 to 34 ppm in Ni (15 ppm in concrete), and 4.4 to 8.7 ppm in Mo (<3.7 ppm in concrete). It was observed that waste samples contained more heavy metals than the lab-prepared concrete used for comparison in this study, but the ranges were of the same order of magnitude as those reported in other studies (cited and discussed above).

The resulting pH, when samples are immersed in pure water, varied between 11.6 and 12.6, according to the carbonation degree. At these pH values, the leached quantities of metals were 0.002 to 1 ppm Cu, ~0.5 ppm Mo, 0.1 to 1 ppm Cr and ~0.5 ppm V, while only vestiges were detected in Pb, Cd, Zn and Ni (i.e. <0.01 ppm, which is lower than the detection limit of the analytical method used). The leachate concentrations (as obtained in the leaching test) ranged from 10^{-7} to 10^{-4} mg/L for Cu, $5 \times 10^{-5} - 10^{-4}$ mg/L for Ni, $5 \times 10^{-5} - 5 \times 10^{-4}$ mg/L for Cr and $5 \times 10^{-5} - 10^{-4}$ mg/L for Mo.

These leached quantities were similar to those released by the concrete material produced in a parallel experiment. As observed, the leached quantities are not directly correlated to the metal content in the respective recycled materials. The most important factor for their release seems to be their speciation and fixation mode in the solid matrix.

Following the experimental results and geochemical modelling assays performed by the authors, the metals released in significant quantities behave in the following ways:

- as oxyanions entrapped in ettringite phases (Mo, Cr, V);
- as oxides and hydroxides for Cu and Ni. Their leaching is enhanced by complexation with traces of organic matter.

These conclusions corroborated previous studies performed on cement based materials. A compliance batch leaching test was used (Galvín *et al.*, 2012) in two variants (the Dutch and the European versions) to assess the leaching behaviour of RA from building demolition, collected from a treatment plant. The applied tests consisted of extractions in water in different conditions. In the Dutch test (NEN7341), two consecutive extractions are conducted at controlled pH values 7 and 4, at a liquid-to-solid ratio (L/S) of 50 L/kg with 3 h contact time for each step. In the European test (UNE-EN 12457-3), two successive extractions are conducted with deionised water at L/S of 2 and 10 L/kg, at the natural pH of the material and with a contact time of 6 and 18 h, respectively.

The leaching results for emission of heavy metals were analysed with respect to some operation conditions, pH and L/S. It was concluded that: the pH is the most influencing factor, therefore acidic aggressive conditions are suitable to evaluate the leaching potential of an element; the quantity of metal released (mg element/kg material) increases with L/S.

The waste samples were then classified according to standards specified by the landfill admission conditions (Council decision 2003/33/EC). This regulation classifies the wastes as inert, non-hazardous or hazardous, depending on their leaching results. Although landfill is not the main focus of this chapter, this example is relevant because of the assessment methodology used. The results obtained in aggressive acidic conditions classified the majority of wastes as non-hazardous, while results obtained at natural pH (European test) suggested they are inert ('inert' is more benign than 'non-hazardous').

This study confirms the former remarks (Section 24.3.2) concerning the importance of the experimental leaching methods used to determine the hazardous character of the material. Contradictory classification, within the parameters of a given regulation, may be assigned for a certain material, depending on the leaching test used, since the concentration of leached contaminants can differ.

24.5 Concrete materials containing RAs: properties and potential hazards

Very few studies have been published so far reporting leaching or hazardous properties of concrete made with RA. A concrete prepared with masonry and/or concrete residues is likely to behave as a 'common' concrete material (produced with primary raw materials), if the contents of the materials used comply with standards. Consequently, the numerous studies performed so far on release of pollutants from concrete and 'model' materials could be used as a basis for understanding the leaching mechanisms.

In their study, Sani *et al.* (2005) compared the leaching properties of a concrete material prepared with NA with a concrete incorporating demolition aggregates collected at an industrial site in Italy. For test material preparation, a commercial CEMII/A-L42.5R was used. The NA, such as sand and gravel, were entirely replaced by demolition aggregates with appropriate granulometry. X-ray diffraction analysis has shown that both materials have similar solid composition, namely, major crystalline phases.

A dynamic leaching test was performed on samples of 10 × 10 × 10 cm, with sequential renewal of leachate (deionized water) for 26 days. The leachates were analysed for conductivity, alkalis (Ca, Na, K) and chloride. This study showed that the leaching behaviour of the material containing RA is close to that of a conventional concrete. However, conventional concrete has a higher density and lower porosity, which affects the diffusion of soluble ions. The estimated effective diffusion coefficient was 1.5 to 2 times higher for the concrete with RA. The authors also observed that Ca release is slightly lower in the material containing waste. This lower availability for leaching could be the result of the carbonation of the demolition aggregates and a certain reactivity with respect to portlandite contained in cement. The overall observed interval of pH (a crucial parameter for the leaching process) is reported as being 10.3 to 11.4. This information is not sufficient to fully understand the chemical mechanisms occurring in the process. The heavy metals or other pollutants were not tracked, so environmental concerns were not addressed.

Lead-contaminated masonry contaminated by lead-based paints is commonly used as an aggregate. Lead and other heavy metals (Cr, Co, Cd, Hg, Sb, etc.) are toxic for living organisms. Their main toxicology issues, when used in construction materials, were discussed by Pacheco-Torgal *et al.* (2012). The general behaviour of these pollutants was also presented in Section 24.3 of this chapter.

Very little has been published on the type and use of contaminated aggregates and concretes in actual situations. A study was recently performed on several concrete materials made with OPC and RA contaminated with lead (Hu *et al.*, 2010). The RA came from two types of clay bricks and concrete blocks, all initially painted with lead-based paint, and then crushed and incorporated in concretes with different ratios of aggregate to cement and water to cement. A TCLP leaching test was used (see its brief description in Section 24.2.4). The lead total content was found to be between 4.2 and 21.4 g/kg material. The leachate Pb concentration was between 0.05 and 10 mg/L for the different materials, while the leachate pH varied between 6.4 and 11.5, with the higher values observed in formulations prepared with aggregates from painted concrete blocks. There is no effective correlation between the material composition, leachate pH and the lead concentrations. By comparing TCLP results with national standards in terms of limit concentration, the authors concluded that almost all samples are classed as non-hazardous, according to standards for acceptance in municipal landfills. It must be noted that these values correspond to a particular leaching protocol and do not directly inform about the toxicity level of the materials.

Although interesting, this example is not sufficient to conclude on the environmental compatibility of such materials. More research into the effect of the composition of recycled demolition wastes on their leaching properties is needed. Detailed analysis and appropriate tools, such as distinct leaching scales and modelling, are necessary to assess accurately the pollutant emission when recycled waste is used.

24.6 Conclusions

More efficient use of material and energetic resources is becoming a key issue for the sustainable development of the construction industry. This area is responsible for the highest contribution to the emission of greenhouse gases, and also demands high energy and material flows. The implementation of a responsible, safe and efficient circular economy that re-uses materials where it can, will contribute to the significant reduction of the negative environmental impact of this area.

Limiting the extraction and processing of mineral resources and the associated energy consumption (extraction, transport, manufacturing, etc.) by the appropriate re-use of residual materials from deconstruction or renovation of buildings is an obvious way. However, the process has to be strictly controlled and traced to eliminate all risks, particularly to human health. Within the EU, this complex process has begun. The EU Regulation, No. 305/2011, fixed basic requirements for construction works and building materials, while the European Standardization Committee (CEN) is working on the elaboration of the necessary standards and methodologies.

One of the problems under consideration is the release of pollutants from the construction materials caused by leaching and their dispersion in the environment, according to typical exposure scenarios implying specific reactive mass transfer and transport mechanisms. If general and specific methods and tools have been developed for assessing the harmful effects of pollutants on humans and the environment (ecotoxicological tests, etc.), research has to be done for their multi-scale validation and regulatory use. No sufficient toxicological studies on solid materials or on their leachates have been published.

A large amount of experience exists in the field of the leaching behaviour of inorganic contaminants from wastes. Numerous regulatory and research tests and a rich modelling experience into the development of a global methodology and its multi-scale validation have been conducted. The more limited knowledge concerning the leaching behaviour of organic contaminants, as well the complex interactions between organic and inorganic pollutants (synergetic or blocking effects, etc.) means that more research is required in this area, especially when recycling involves multi-component and heterogeneous residues generated by distinct activities.

RA and concrete produced with those materials show similar chemical composition, structure and leaching properties to the NA and concrete produced with primary raw materials. If no particular contaminants are present, these materials release chemical species (essentially heavy metals: Cu, Cr, Mo, Pb, Ni, Zn, V, etc.), at a comparable release level.

If a particular contaminant exists in a significant quantity, as lead does in lead-based paints, and when certain organic compounds are present, the leaching behaviour of the hazardous species depends on speciation in the cement paste and solubility in alkaline leachates (water in contact with cement). The pH of leachate plays an important role, as observed for amphoteric species.

A few studies report the leaching behaviour of concrete produced with RA, discovered using laboratory leaching tests. However, the real conditions upon the service life are more complex and involve multiple external parameters that might influence the leaching process. Those aspects have not yet been researched. Currently, it is thus not possible to conclude about the health safety and eco-compatibility of construction scenarios using RA. Moreover, the lack of standard procedures for environmental assessment of construction materials in their conditions of use has delayed the acquisition of relevant data and definition of conclusions, and therefore hampered the recycling of construction and demolition wastes.

The development of a European trade and competitive market for concrete containing RA requires that the new products comply with the construction product regulation (allowing the CE marking). One of the requirements specifically refers to hygiene, health and the environment. So, two major actions are required. The first is a rapid development of the European regulation in the field of environmental assessment of construction materials. Regulation must consider the following points:

- set-up of standard procedures, concerning leaching and construction materials;
- proposition of simplified methodologies for extrapolating laboratory data to field conditions, in order to evaluate pollutant emission in the actual scenarios in which it occurs.

The development of such regulations should enable a uniform investigation of a material's eco-compatibility in different countries and should assist the concerned industry with the necessary tools for their products' characterisation and their processes' optimisation. The second requires the development of a thorough knowledge of the potential hazards of concretes containing RA, by using the various existing and forthcoming tools discussed in this chapter.

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