Framework for Recycling of Wastes in Construction

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Abstract: Waste and recycled materials (WRM) that are used in structural systems are required to satisfy material strength, durability, and leachability requirements. These materials exhibit a wide variety of characteristics, owing to the diversity of industrial processes that produce them. Several laboratory-based investigations have been conducted to assess the pollution potential and load-bearing capacity of materials such as petroleum-contaminated soils, coal combustion ash, flue-gas desulphurization gypsum, and foundry sand. For full-scale systems that incorporate WRM, although environmental pollution potential and structural integrity are interrelated, comprehensive schemes have not been widely used for integrated assessment of the relevant field-scale performance factors. In this paper, a framework for such an assessment is proposed and presented in the form of a flowchart. The proposed framework enables economic, environmental, worker safety, and engineering factors to be addressed in a number of sequential steps. Quantitative methods and test protocols that have been developed can be incorporated into the proposed scheme for assessing the feasibility of using WRM as partial or full substitutes for traditional materials in construction.

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Introduction

In an effort to promote the sustainable use of natural materials, many countries, regions, and municipalities are increasingly formulating policies that promote the large-scale recycling of byproducts and used materials, herein referred to as waste and recycled materials (WRM) in a variety of applications. The development of physical infrastructure for civil and industrial operations provides significant opportunities for the use of WRM in large quantities. Their use has reduced waste storage costs and minimized the dereliction of land near urban areas where high volumes of construction often generate high demand for natural aggregate. Ironically, urban centers generate the highest volumes of recyclable wastes per unit area of land because of relatively high population densities.

In the United States, data (U.S. Environmental Protection Agency 2002) show that 210.4 million tonnes of municipal solid waste was generated in 2000, out of which 58.4 million tonnes was recycled. These data represent a substantial increase from those of 1960 that show that 79.9 million tonnes was generated with only 5.1 million tonnes recycled. Currently, about 3 million tonnes of refractory materials are produced in the United States (Bennett and Kwong 1997) but Nystrom et al. (2001) estimate that less than 10% of these materials are recycled. Large quantities of coal combustion products (CCP) are recycled each year in the United States. Of the 100 million tons of CCP produced in 1997 (American Coal Ash Association 1998), about 1.8 million tonnes were recycled in land reclamation projects. Based on the data presented by Ziemkiewicz and Skousen (2000), current production of flue gas desulphurization (FGD) solids exceeds 22.7 million tonnes annually out of which about 9% is recycled.

In addition to both the sentimental value and environmental sustainability considerations of using WRM in construction, two additional factors are determinants of the trend. First, the development of transportation facilities such as roadways, railways, and pipelines in countries that have very large land masses with population centers that are separated by long distances requires huge quantities of construction materials each year. The use of virgin materials alone is unsustainable in terms of costs because such materials would have to be transported from sources that are distant from locations of use as proximal sources become depleted. Material haulage costs are exceedingly high relative to the cost of residuals and wastes produced near construction sites. The United States and Australia are examples of large countries where recycling is favored. In the United States, vehicular and truck traffic are expected to increase by 45 and 90% respectively, within the next 20 years, according to recent projections by the Federal Highway Administration (Stidger 2002). This will require the use of huge quantities of materials of which a sizable fraction will need to be WRM. Spatial expanse as a driver of WRM use is best illustrated by the Australian scenario in which about 900,000 km of roads serve a small population. As observed by Gnanendran et al. (2002), Australia has a road length per capita of 450 m compared to 280 m for the United States and 90 m for Japan.

An additional driving factor for use of WRM in construction is high population density. This is most prevalent in countries and regions with large populations and scarce space for disposal of wastes. An example is India where about 65% of total electricity generation comes from combustion of high ash content coal (Battacharjee and Kandpal 2000), and coal reserves are expected to last for at least 100 years (Economic Intelligence Service 1996). Owing to the need to manage increasing quantities of coal ash, the Indian Ministry of Environments and Forests (MOEF) issued

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new regulations in 1999 that require that new power plants use 100% of the fly ash that they produce within 9 years of the beginning of their operations. Existing power plants are to comply within 15 years. It is estimated (Chen et al. 2002a) that landfill space in Hong Kong will be exhausted by 2015. This evolving scenario has highlighted the need to increase WRM use in Hong Kong, particularly, demolition wastes such as waste glass. The annual generation of waste glass in Hong Kong is 58,060 tonnes of which 45,360 tonnes is recoverable (Chen et al. 2002a). Singapore, an island city-state with a land area of only 647.5 km^2 and perhaps the highest population density in the world (more than 4,600 persons km^{-2}), produces huge quantities of wastes without adequate disposal space. Based on data quoted by Seik (1997), the daily solid waste output in Singapore increased from 2075 tonnes in 1973 to 7329 tonnes in 1995. The average annual rate of growth of solid waste generation in Singapore was 11.5% up until 1995. The annual quantities of solid wastes disposed rose from 0.74 million tonnes in 1972 to 2.80 million tonnes in 2000 (Bai and Sutanto 2002). Singapore's Environmental Pollution Control Act (EPCA) of 1999 contains provisions that support large-scale recycling of wastes.

A 1998 survey (Wei and Huang 2002) indicated that about 18 million tonnes of industrial waste is generated in Taiwan each year, out of which 8% (1.47 million tonnes) is classified as hazardous waste. This waste generation rate roughly translates to 500 tonnes per km^2. The 1996 Hazardous Industrial Waste Reuse Permitting Process established by the Taiwan Environmental Protection Administration (Taiwan EPA) promotes the reuse of WRM such as waste paper, coal ash, wood, glass, cast sand, plastics, and scrap metal. A recycling target of 54% by 2004 has been set. Material balances in Japan require larger scale utilization of WRM than in most other small-sized countries because of high industrialization rate and consumption patterns. In 1995, Japan discarded about 725.7 million tonnes of materials out of which 190.5 million tonnes was recycled (Ishimoto et al. 2000). Each year, the areal waste production of about 650 tonnes/km^2 (Wei and Huang 2002) creates the need for recycling large volumes of WRM in construction. The approach adopted in Japan is the "4RE concept" (reduction, recycling, reuse, and renewal). A large fraction of wastes is reprocessed into high-value construction and other products using novel techniques. As an example of the deviation from the approach that is primarily adopted in other countries, paper sludge in Japan is being reprocessed into concrete admixtures and cation exchangers. Ishimoto et al. (2000) report on the use of paper-making sludge containing 5–20 g/m^2 coating agents to produce zeolite through a process of incineration and reaction with an alkali.

Individually, European countries, particularly the Netherlands, took the early lead in the development of national policies and framing/implementation of research to support large-scale utilization of WRM in construction. In most of the Nordic countries, land disposal space for wastes is highly limited, and surface and groundwater hydrology does not favor widespread and long-term storage of wastes. Currently, although some individual countries in Europe still maintain jurisdictions over regional waste management programs, more general policies are now framed within the European Union (EU) Framework Directive on Waste (91/156/EEC). Therein, waste is defined as any substance or object which the holder discards or intends to discard. As relisted by Read (1999), this definition puts such a substance or object into one or more of the following categories: production or consumption residues; products whose dates of appropriate use have expired; materials that are contaminated or soiled; and substances and objects that no longer perform satisfactorily. The European Union Landfill Directive (OF 1999) calls for the following phased reductions in the amount of biodegradable municipal solid waste disposed in landfills relative to 1995 levels: to 75% by 2010; to 50% by 2013; and to 35% by 2020.

The Dutch Building Materials Decree (BMD) which came into full operation in 1999 following a 3-year trial period, combines soil and groundwater protection standards in establishing performance requirements for construction materials, regardless of whether they are primary (traditional) materials or WRM. Owing to many years of intense research on the leachability of chemical substances from the class of WRM materials produced in the Netherlands, the Dutch have been able to recycle about 90% (Eikelboom et al. 2001) of their waste stream in many applications, primary of which is construction. In Finland, almost all of the solid wastes produced are utilized because of lack of waste storage space and acceptable sources of primary construction materials. For example, in 2001, Helsinki Energy produced 86,183 tonnes of fly ash, 18,144 tonnes of bottom ash, and 22,680 tons of FED residues (Havukainen 2002). About 50% of the ash was used in concrete while the remainder was used in earthwork constructions.

In the foregoing discussion, it is evident that many countries have developed materials management policies that support waste recycling in large quantities. The driving factors which translate to economic considerations are material generation rates, lack of waste storage/disposal space, demand for construction materials, and available technical support systems to address engineering and environmental issues.

Material Substitution Factors

In general, waste and recycled materials (WRM) that are utilized in construction are generated from a diversity of sources and through a variety of industrial processes and materials reclamation operations. A preliminary identification of the categories of factors that determine the extent to which WRM can satisfy environmental, worker safety, occupational health, and structural performance requirements is made in Fig. 1. These factors largely define the economic and technical feasibility of utilizing WRM in large-scale construction. While some factors pertain to the physical characteristics of the waste material, other factors relate to the design, mix proportioning, regional hydrology, site stability, level of loading, and type of facility. In this paper, the focus is herein kept on highway and embankment construction because of the high quantities of WRM that are currently used in those applications.

Material Source and Characteristics

The physical and chemical characteristics of WRM stem from their sources, processing methods, and handling techniques. In turn, these factors partly determine their suitability for use in construction with respect to satisfaction of strength, durability (structural), and leachability (environmental) requirements. Slow cooling of slags and other materials that are generated through high temperature processes result in coarse grains whereas fast cooling in liquid or air results in glassy and fine textures that could result in lower values of substance diffusion coefficient when such materials are used in construction. Considering coal ash characteristics, for example, conventional coal-fired power plants burn coal at temperatures of about 1600°C. Element dis-

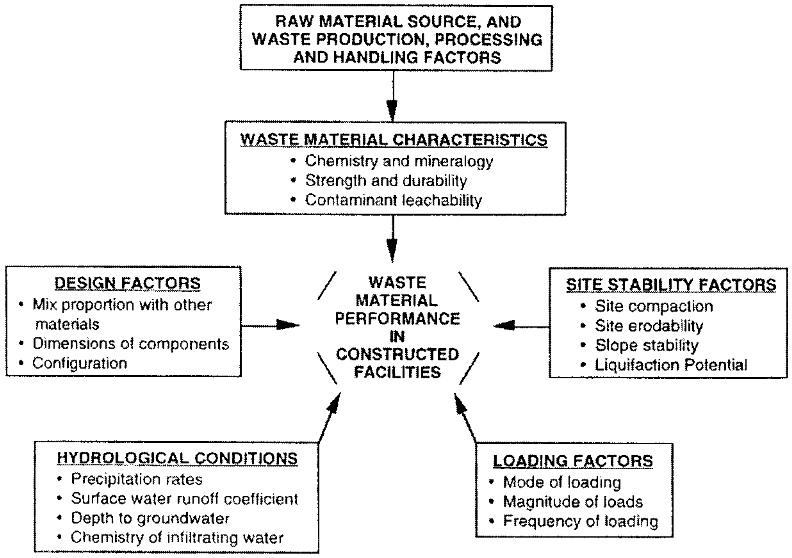

Fig. 1. Factors that control environmental and structural performance of waste and recycle materials (WRM) in constructed facilities

tribution and mineralogy of the ash produced from a plant depend on coal source, the design of the plant, and the system design for ash collection. Wet ash removal systems are richer in elements than dry systems (Chadwick et al. 1987). Additional information on the fate of trace elements during coal combustion has been provided by Sandeling and Backmand (2001).

Facility and Mix Design Factors

In addition to the intrinsic characteristics of the WRM, the design of the mixes of WRM with other materials (mix proportioning) and the structural design of the facility for which the material is used affect the performance of WRM in construction. The mix design determines the quantity of the WRM present per unit volume or weight of the composite mix and affects transport parameters such as substance diffusion rates, leachability, and durability of the composite mix. The design of a WRM facility in terms of the degree of binding and/or coverage of the WRM and the thicknesses of WRM-amended components is a significant factor in structural and environmental performance. When WRM is bound as in bituminous and portland cement concrete and stabilized bases, contaminant leachability is minimized and strength is highly influenced by the degree of cementation. Unbound systems provide higher opportunities for contaminant leaching. Inyang (1998) has discussed additional design factors that influence WRM performance.

Loading Factors

The mode, magnitude, and frequency of loading affect WRM performance in constructed facilities. Some facilities such as pavements and foundations are designed to carry external loads and place greater strength requirements on WRM than other facilities such as flowable fill for utility trenches or unloaded embankments. In addition, both categories of structures may be subjected to fatigue stresses that arise from environmental phenomena such as freeze-thaw cycling, wet-dry cycling, and desiccation. For one of the most common uses of WRM (i.e., pavements), the partition of released contaminants to runoff from the pavement surface and infiltrating moisture depends partly on the structural state of the pavement as illustrated in Fig. 2. At very large fracture spacings that are common within the first 5–10 years of service of bituminous pavements, surface drainage is more pronounced than infiltration because the pavement is still mostly intact. Thus the runoff coefficient is typically high (above 0.5) with the infiltration fraction being relatively low as depicted by I_m in Fig. 2. The concentrations of targeted contaminants in the runoff water are primarily dependent on their diffusion rates from the asphalt pavement across the pavement surface into the flowing water. Being that the diffusivity and permeability of intact asphalt concrete are in the orders of 10^{-12} and 10^{-9} cm/s, respectively (Testa et al. 1992; Hickle 1996), the emission rates of contaminants into runoff water on well-cemented WRM pavements should be expected to be very low initially. However, with service, pavements deteriorate such that their diffusivities and permeabilities increase. Increased infiltration which is depicted in Fig. 2 by I_c implies that the runoff coefficient could decrease to levels lower than 0.5, depending on fracture density, slope factors, and rainfall intensity. The net effect of increased infiltration is greater opportunity for contact between encapsulated WRM aggregates and moisture. A review of the factors that determine the generation, geometry, and flow through such fractures has been provided by Bai et al. (1996, 2000).

Site Stability and Hydrological Conditions

The rates of release and migration of substances from WRM used in facilities construction are affected by site hydrology and stability. Precipitation supplies the moisture that serves as the leachant and transport agent for displacement of the substances into the surrounding media. Site instabilities such as foundation and slope failures can exacerbate WRM deterioration by exposing them to agents of weathering and erosion. During facility construction or rehabilitation with WRM or WRM-amended materials, it is usually convenient to stockpile the material along the cleared tracks used for construction. A common concern with material washout under this scenario is illustrated in Fig. 3. In the cross-section of Fig. 3, a conically shaped waste pile is placed on a road side flanked by a slope that terminates at a pond or water-containing roadside ditch. Rainwater falls uniformly on the roadway pavement, waste pile, and slope. A fraction of the rainwater that falls on the pavement percolates downward while a fraction drains into

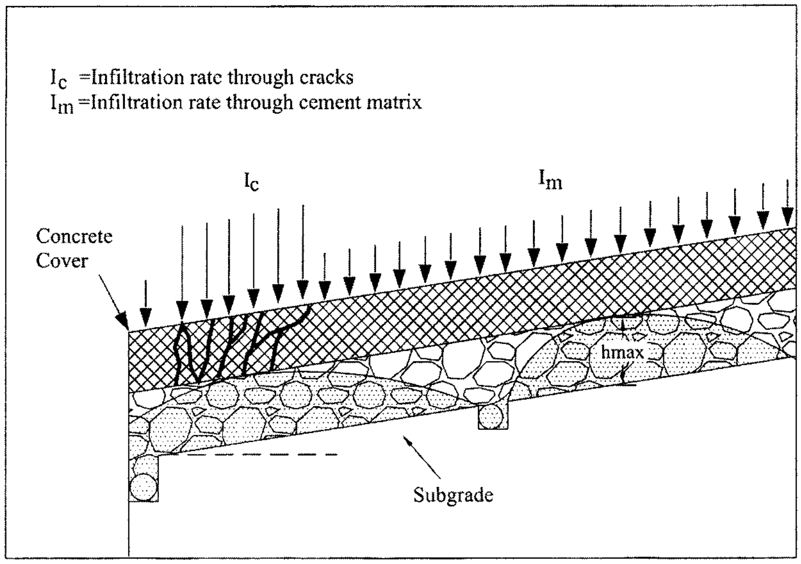

Fig. 2. Structural deterioration and its impacts on infiltration of pavements and covers

the granular waste pile. Rainwater also percolates downward through the waste pile, thereby raising the concern that the roadside pond may receive contaminated water that could threaten its ecology. In general, contaminant loadings from highways have been characterized by Wu et al. (1998).

Contaminant leachability from the waste pile is directly proportional to the specific surface of the WRM, within the limits imposed by the original content (or mix proportion) of the contaminant in the material. The contaminant concentration in the water that sweeps through the base of the waste pile builds up from C_n to the maximum level, C_{rp} where it exits the waste pile but may be reduced to C_e immediately before the entry of the draining water into the pond. Such a reduction is attributed to dilution by rain that falls on the slope and possible adsorption of contaminants by the ground over which the water flows. In the pond, dilution and mixing further reduce the concentration to C_p (a concentration that should be the focus of ecological risk assessments rather than C_{rp}). Several researchers (Tossavainen and Forssberg 1999; Yukselen and Alpaslan 2001; Huang 2003) have developed methodologies and experimental data on the assessment of contaminant leachability from materials. The challenge is to integrate such models and test data into a quantitative framework that describes fluid percolation as influenced by specific scenarios and system physical configurations exemplified by Fig. 3.

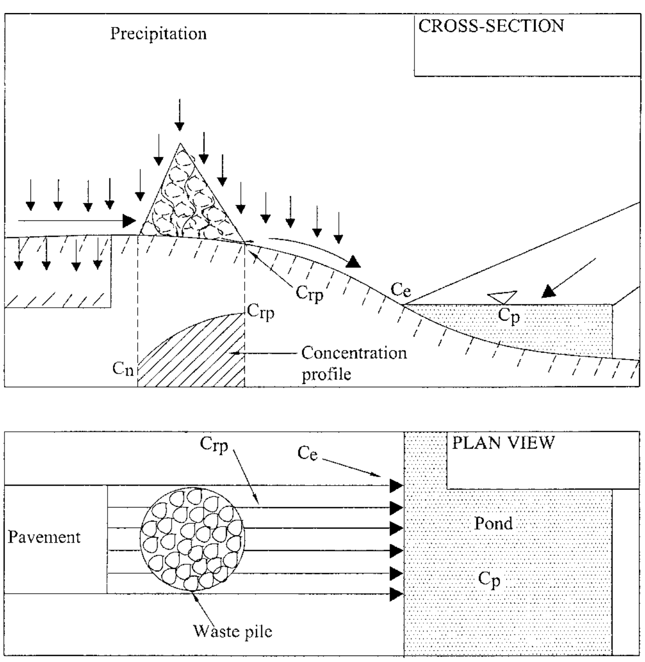

Fig. 3. Schematic illustration of contaminant washout from waste piles at construction sites

Recycling Potential Evaluation Methodology

The physical and chemical characteristics of WRM derive from their sources, processing operations, and handling techniques as discussed in the preceding section. These characteristics partly determine the extent to which each WRM can satisfy strength, environmental, worker occupational safety, and health requirements. It is desirable to develop a WRM evaluation scheme that allows a systematic analysis of candidate materials on each of the significant utilization parameters. Such a system which is proposed and discussed herein, integrates both environmental and structural performance aspects and allows the screening of pure materials and their mixes with other materials. Elements of the scheme have been adapted and expanded from the preliminary scheme proposed by Inyang (1992). The proposed assessment methodology which is illustrated in Fig. 4, can be used for a specific WRM material considered for a specific project. It requires information which is usually gathered to support material evaluations for use in construction projects. This 14-step methodology which also covers project implementation and monitoring aspects is briefly described in this paper with references to literature where necessary.

Step 1. Establishment of Facility Performance Standards

Each facility in which WRM is used as a construction material has specific design functions. Often, the WRM is used in con-

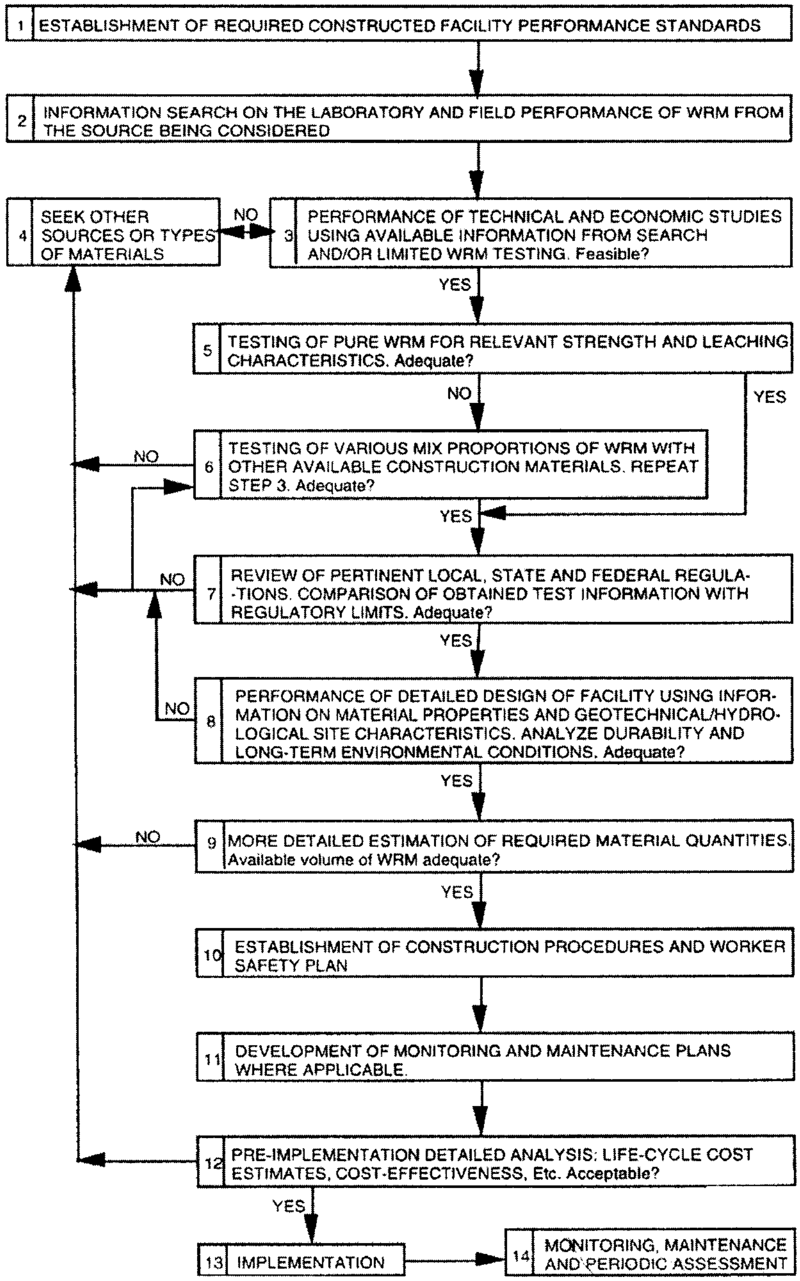

Fig. 4. Analytical steps of feasibility assessment and project implementation methodology for use of waste and recycled materials (WRM) in construction

structing a component of a structural system that may comprise several components that are built with different materials or material mixes. The performance requirements of a facility generally drives the selection of its design configuration, component dimensions, and material mix proportions. The first step of this methodology is the determination of the required performance standards. For a bituminous pavement in which bottom ash aggregate is used, performance standards can be specified in terms of specific strength and durability requirements that can be assessed through standard testing. As another example, if a WRM is admixed with other materials for use as a landfill cover material, the composite material needs to be amenable to the following design functions of landfill covers: minimization of water infiltration and fire hazard, containment of gases, inhibition of dusting, service as a medium for plant growth, and adequate stability if the use of the landfill surface for recreational facilities or construction of temporary structures is planned. The USEPA (1989) developed minimum requirements for the properties of materials that may be used as landfill cover materials. A critical requirement is a maximum hydraulic conductivity of 10^{-7} cm/s.

Step 2. Information Search on Source Characteristics and Continuity

When large-scale construction projects are targeted for WRM use, excessive fluctuations in construction material characteristics can negate the economic gains that would stem from WRM use. The cost of frequent redesign of mixes and perhaps changes in construction procedures to accommodate wide variabilities in WRM characteristics can be high. Such would also be the case if only a limited quantity of the WRM is available for a project that requires high volume application of materials. Thus, this evaluation step involves the search for general information on the ranges of characteristics of the WRM considered and the prospects for adequate quantities to be continuously available for use in the target project. Test data on a variety of WRM have been generated and published by investigators in many countries during the past 20 years. Some examples are those on flue gas desulphurization jgypsum (Taha et al. 1992), slags (Proctor et al. 2000; Nabeshima and Matsui 2002), paper mill sludges (Moo-Young and Zimmie 1996, 1998), municipal solid waste incineration ash (Collivignarelli and Sorlini 2002), and sewage sludge (Chen et al. 2002b).

Steps 3 and 4. Performance of Economic/Technical Studies Using Available Information

This step is the screening assessment of the economic feasibility of using the WRM for the target application using information collected from literature and related projects. At this stage, the material has not yet been subjected to the variety of elaborate tests needed for detailing of required quantities. The assessments cover rough estimation of the amount of materials to be used, haulage distances and costs, and associated construction costs of various ranges of ratios and proportions of WRM substitution for traditional materials. This is a "what if analysis." If the analysis shows that for a reasonable target level of WRM substitution for traditional materials, the cost is excessive relative to the cost regime of other options, then using this WRM for the specific project should be considered not to be feasible, and other types and sources of materials should then be sought for the project as directed in Fig. 4. If feasibility is indicated, then the next analytical step is taken.

Step 5. Testing of Unmixed WRM for Relevant Strength and Leaching Characteristics

Although many materials that are considered to be WRM are frequently used in combination with other materials in construction, material approval processes that target their strength and contaminant leachability often focus on the unmixed material rather than its mixes with other materials. It should be noted that although this approach suffices for regulatory and policy considerations, it does not provide the scientific basis for simulating the performance of WRM mixes with other materials during facility service in the field. Nevertheless, strength and durability tests of unmixed WRM during this stage of the evaluation provide data for comparison with specifications. The type of strength tests selected depends on the design functions and load-bearing requirements of the structural components in which the WRM is intended for use. For example, the unconfined compressive strength is applicable to pavement base and embankment materials. Leaching tests on unmixed materials provide data for screening of WRM with respect to satisfaction of regulatory standards. For example, in the United States, the toxicity characteristics leaching procedure (TCLP) test establishes concentration limits for substances beyond which a material is classified as hazardous waste. Although there is no direct quantitative correlation between those limits and prospective emission rates of the substances from WRM in the field, many jurisdictions use the TCLP test data as the basis for screening of unmixed WRM in construction. An unmixed material that exhibits acceptable characteristics at this stage is subjected to step 7 evaluation. If an unmixed material fails step 5 but there is still an economic advantage of using it, its mix proportion with other materials can be considered and evaluated in step 6.

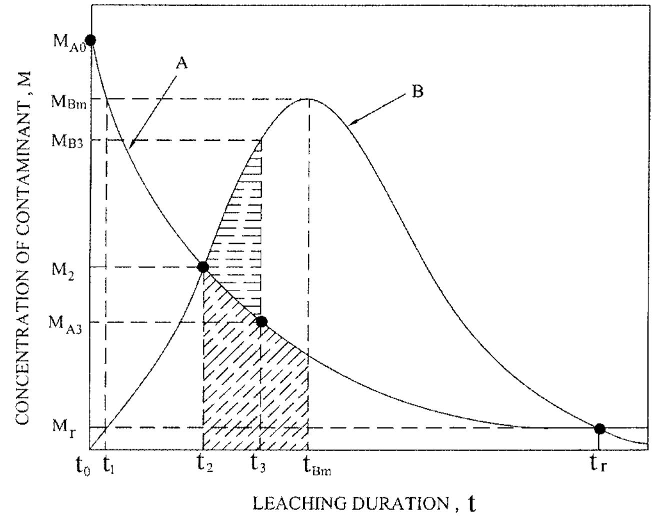

Fig. 5. Idealized patterns of contaminant leaching from substances in continuous leaching tests

Step 6. Testing for Selection of WRM Mix Proportions with Other Materials

In cases in which WRM is intended for use in mixes with other materials, testing of the mixes rather than unmixed samples provides more realistic information on the prospective performance characteristics of the composite material than testing of unmixed samples. Step 5 focuses on the determination of the acceptable mix proportion of the WRM with other materials and provides baseline information for use in facility design and models for performance assessment.

Several leaching test protocols have been developed to model the release of substances from materials. Unfortunately, laboratory-based leaching tests can never be completely simulative of natural processes in the field because of the time constraint element and involvement of a greater number of environmental factors in the field. The synergistic effects of the factors cannot be fully captured in the laboratory tests. Besides, some processes such as weathering that can generate a new assemblage of leachable minerals and elements occur in the field at rates that are too slow for coverage in standard laboratory tests. Fig. 5 shows some idealized patterns of contaminant leaching from materials that are often observed in laboratory-based tests. For curve A, the contaminant concentration in the leachate exhibits negative exponential decay with time. Examples of this pattern are data recorded by Griffin et al. (1980); Hayward et al. (1986); Miner et al. (1986); Electric Power Research Institute (1987); and Garcez et al. (1987). Curve A is of the form

$$M_t = M_0 \exp(-kt) \tag{1}$$

where M_t = contaminant concentration at leaching duration t_i [M/L^3]; M_0 = contaminant concentration at reference time t_0 [M/L^3]; k = empirical leaching constant; and t = time. In terms of the cumulative fraction leached,

$$C_t = a + bt^{0.5} + ct \tag{2}$$

where C_t = cumulative fraction leached. The terms a, b, and c = empirical constants. It is generally believed that the term "a"

depends on the initial wash-off of pollutants that may be attached to particle surfaces. Conceivably, the term "b" should depend on the contaminant species diffusion from granular interiors to surfaces. Thus the internal pore structure and shapes of the WRM particles play a role. This is influenced by the WRM source, processing (discussed earlier in conjunction with Fig. 1), and test conditions. The term "c" is believed to depend on kinetically controlled chemical reactions which may be significant at the solid–leachant interface. One would expect that the decay of contaminant concentration with time may be attributable to transport constraints (or longer travel times) as the diffusing contaminant gets depleted in the outer regions of each particle.

The empirical leaching constant "k" in Eq. (1) is dependent on parameters such as leachant pH (Fallman and Aurell 1996; Fleming et al. 1996; Webster and Loehr 1996; Fallman 1997; Poon and Lio 1997; Wang et al. 1999; Herck and Vandecasteele 2001; and van der Sloot et al. 2001a,b), material particle size (Wahlstrom et al. 2000; and Feda 2002), agitation conditions, and leachant/solid (L/S) ratio. In some cases, the leaching pattern follows curve B of Fig. 5. The concentration of the target substance builds up with time, to a maximum designated as M_{Bm} at time t_{Bm}, and then decays thereafter. This pattern usually results from either the delayed access of the aggressive components of the leachant (e.g., H^+ in acid leachants) to interior sites of the leached substances in the material or slow rates of occurrence of the reactions needed to release the leached substance from the solid.

Step 7. Assessment of Pertinent Regulations and Regulatory Limits

The strength and contaminant leachability data obtained in steps 5 and 6 should be compared with regulatory limits and specifications that may exist on the use of specific materials or classes of materials for constructing facilities in general, or in specific applications. While the construction industry in many countries has established material specifications (such as the ASTM standards) for strength and durability, environmental specifications for WRM mixtures are less standardized. Where regulatory limits exist, then data from steps 5 and 6 need to be used to examine compliance with reference to Fig. 5. It is not realistic for any regulatory authority to use concentration levels represented by M_{A0} and M_{Bm} to establish maximum emission limits for WRM because these concentration levels are not sustained over a long segment of the leaching test duration. More realistically, a regulatory leaching test duration, t_r, should be specified over which the concentration can be integrated (from t_0 to t_r) to obtain the total quantity of the target contaminant that leaches from the sample. If the specific mix proportion of the WRM with other materials does not satisfy the specifications at this stage, the mix needs to be changed or other materials considered as indicated by the loop in Fig. 4.

Step 8. Performance of Facility Design with the Satisfactory Mix

The satisfactory mix proportion selected in step 7 is then used in step 8 to determine the component dimensions and configuration of the facility that is planned. This iterative process involves the optimization of facility design factors with satisfaction of structural and environmental performance requirements as the objective function. At this stage, the focus is no longer on the WRM sample but the structure in which it is to be used. For example, if the WRM mix is a landfill cover material with the measured per-

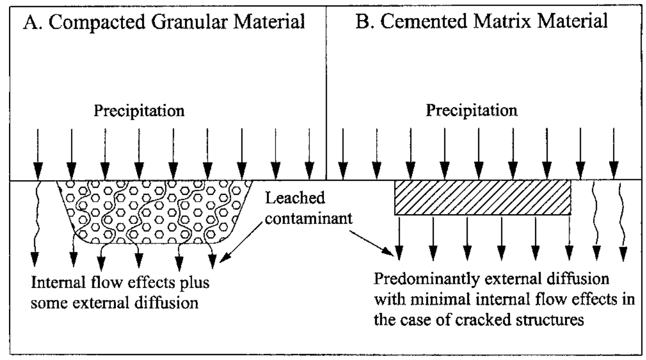

Fig. 6. Categorization of contaminant emission models for structural systems in the field (a) compacted granular material and (b) cemented matrix material

meabilities such as those described by Bowders et al. (1987) and Fleming and Inyang (1995), a design thickness of the cover can be selected to attain a specified maximum infiltration capacity. For applications in which the WRM is intended to carry load, strength related performance requirements (e.g., resilient modulus for pavement bases) can be used to establish the required thickness.

Some investigators have developed quantitative methods for relating material leachability characteristics, facility design, hydrological factors, and site conditions to contaminant emission rates from structural systems. With respect to contaminant emission mechanisms vis-à-vis the structural state of the WRM-amended facility component, there are two categories as illustrated in Fig. 6: a compacted granular material with significant permeability in which the permeating leachant leaches contaminants cumulatively, from the top to the bottom of the emplaced structure; and a cemented mass with very low permeability such that low hydraulic flow rates constrain the removal of contaminant to mostly external diffusion from the soil/cemented matrix interfaces. Schreurs et al. (2000) developed the following equation for treating the case of a compacted granular material:

$$E_P=\frac{d_b h(E_r-E_g)(1-e^{-kr_p})}{(1-e^{k\cdot 10})} \tag{3}$$

In Eq. (3), E_p = quantity of the target contaminant released from the waste heap or buried mass [M/L^2]; d_b = density of the soil [M/L^3]; E_r = quantity of the contaminant leached in a standardized column test (NEN 7343) at a liquid/solid (L/S) ratio of 10 [M/M]; E_g = average quantity leached from clean soil at L/S = 10 [M/M]; k = constant that relates to the degree of interaction of the contaminant with the matrix (dimensionless). Essentially, the magnitude of k is a partial determinant of the geometry of the leached concentration versus time curve; r = liquid/solid ratio [M/M]; and r_p = liquid/solid ratio reached in practice [M/M] and can be estimated through the use of

$$r_p=\frac{Nj}{d_b h} \tag{4}$$

N = net infiltration rate [L/T]; j = period of exposure in the field [T]; and d_b and h are as defined previously. It should be noted that the parameter E_r can also be estimated using models such as those summarized by Poon et al. (1999) and Bishop (1986). Furthermore, for this percolation controlled scenario, Kosson et al. (2002) have established that:

$$M_t=r_p\cdot s_{fp} \tag{5}$$

In Eq. (5), M_t = cumulative mass of the contaminant released at time t after the beginning of leaching in the field [M/M]; and s_{fp} = solubility of the contaminant [M/L] at a pH value that is prevalent at the field site. For the case of the cemented matrix illustrated in Fig. 6, Inyang et al. (2003) have developed the following equation to estimate the maximum concentration (C_d) of the target contaminant in the emplaced concrete that will not result in the exceedance of the limiting concentration (C_w) specified for a well placed at some distance away from the emplaced concrete:

$$C_d \leqslant 0.9 C_w V_t \left[[F_d S K_p (D_e t)^{0.5}] \left(\sum_{i=1}^{n} A_i \right) \right]^{-1} \tag{6}$$

In Eq. (6), C_d = maximum concentration of the contaminant in the structural component [M/L^3]; C_w = specified (regulatory) maximum concentration of the contaminant in a well at some distance from the structural component [M/L^3]; V_t = volume of leachant in which the leached mass is transported before exiting the base of the component [L^3]; F_d = dimensionless dilution factor that describes the extent to which moisture from external sources dilutes the released contaminant in its travel pathway to the well (fraction); K_p = mass transfer coefficient of the target contaminant from the solid matter in the concrete to its internal pore (dimensionless fraction); S = external surface area of the concrete through which leaching occurs [L^2]; D_e = diffusion coefficient of the contaminant from the matrix [L^2/T]; t = time since the beginning of leaching [T]; and A_i = contaminant attenuation factor for travel pathway segment i in the direction of the well. This accounts for the reduction in contaminant concentration through sorption, decay, and other attenuation mechanisms.

Leachability test protocols and data analysis methods have been described in detail in various reports and articles (Jackson 1986; Van Der Sloot 1988; USEPA 1989a,b; and Fallman 1997). Others are protocols described by Cote and Isabel (1984), Wahlstrom (1996), Chang et al. (2001), and Moudilou et al. (2002). Quantitative models and frameworks for analyzing the results of various types of contaminant leachability tests have been proposed by Barna et al. (1997), Batchelor (1998), Poon et al. (1999), and Baur et al. (2001). The reader is referred to the documents cited above for details on test protocols and data interpretation. With respect to the use of data from various leaching tests for regulatory purposes, the central question is "how well do leaching test data represent the scenario in the field?" This is a simulation issue. Various tests capture some elements of one or more field scenarios. The scenarios are diverse and depend on many factors that are characterized by uncertainties.

Nevertheless, it is clear that certain types of leaching tests are more suitable than others for use in analyzing contaminant emission potential for identified or formulated scenarios. Although there are variations within each category, leaching tests can be grouped into the three categories described below.

1. Batch leaching: The discrete particulate material is agitated with a leachant in a container for specific time durations. The concentrations of contaminants in the leachate are measured. The toxicity characteristics leaching procedure (TCLP) belongs to this category. Batch leaching tests provide the maximum exposure of waste materials to leachant.
2. Column leaching: In this category of tests, discrete particles of the material are packed into a column with or without compaction. The leachant is introduced, typically, into the bottom of the column such that the hydraulic gradient moves it upward through the material. Thus the leachant enters at the bottom and the leachate flows out of the top of the column. The concentration of the leachate and its flow rate or volume are monitored continuously or at specific time intervals.
3. Monolith leaching: A cemented mass (monolith of the material containing the targeted contaminants) is submerged in the leachant. The concentration of contaminants in the liquid is monitored at time intervals. In the most common variation of this test, the leachate (which is also the leachant) is purged at time intervals ranging from 2 to 90 days. The results are usually analyzed to determine the diffusion coefficients of specific contaminants from the monolith to the leachate. The monolith may be fabricated as a cylindrical or rectangular block. Monolith leaching provides the least surface area for contaminant release per unit mass. The contaminants must diffuse to the surface of the monoliths through much longer travel pathways than in the cases of batch and column leaching.

The thermodynamics and kinetics of leaching phenomena are much more variable in the field than in short-term laboratory tests. Due to relatively long service periods and variable pH, Eh, and temperature conditions within and around WRM in the field, the mineralogy and chemistry of substances available in the pore fluid for leaching can be significantly different from those that are obtained in short-term laboratory tests. In cognizance of this factor, some researchers (Meima and Comans 1997, 1998; Hansen et al. 2001) have developed speciation models that account for phenomena such as surface complexation/precipitation and weathering processes of WRM. Others (Baverman et al. 1997; Baranger et al. 2002; Park and Batchelor 2002; and Dijkstra et al. 2002) have coupled geochemical models with transport equations for use in estimating contaminant emission rates in situations in which time and environmental conditions favor geochemical compositional changes in emplaced WRM.

Step 9. Estimation of Required Material Quantities

In this step, the objective is to determine whether for the mix proportion established, the WRM will be available in sufficient quantities to support the project. Usually, for large-scale construction of transportation facilities and fabrication of concrete, the typical production rates of coal combustion ash and municipal waste incineration are adequate. Other materials such as foundry sand, carpet wastes, and demolition wastes may not be produced continuously in quantities that are large enough to support sizeable construction projects.

Step 10. Establishment of Construction Procedures and Worker Safety Plan

The use of WRM to construct a facility may require the development of special construction and materials handling procedures. This may be necessary due to differences in mix structural characteristics such as workability and/or emission characteristics. An example is the use of petroleum contaminated soils (PCS) in bituminous paving mixtures in which there is the concern about worker safety and the potential for release of vapors during various stages of construction and rehabilitation of pavement. Inyang (1998) has discussed various contaminant release scenarios for PCS in bituminous concrete applications. Volatiles are typically released from PCS into the atmosphere during hot-mix concrete batching operations. In order to address this concern, some states in the United States have placed some restrictions on the proportion of PCS used in concrete. As mentioned by Kostecki et al.

(1989), Massachusetts has restricted the PCS feed quantity to less than 5% (by weight) in at least one plant.

In the United States, some aspects of worker safety regulations and guidelines developed by the Occupational Safety and Health Administration (OSHA), U.S. Environmental Protection Agency (USEPA), and the U.S. Department of Transportation (USDOT) are adaptable to the processing of WRM and construction of facilities with WRM. OSHA's and USDOT's guidelines are more directly relevant to WRM utilization because they focus on occupational exposures and material transportation hazards, respectively. USEPA recommendations tend to be excessively conservative because they were developed for operations on hazardous and radioactive waste sites. Most WRM that are utilized are solids. Potential occupational risks are associated with the following construction activities: sampling of stockpiles; loading of trucks at the plant; mix proportioning of WRM with other materials; transportation of WRM to placement sites; and placement of WRM as designed.

As stated in the U.S. Code of Federal Regulations 29 CFR1910.1200, OSHA defines substances with health hazards as chemicals "for which there is statistically significant evidence based on at least a study conducted in accordance with established scientific principles, that acute or chronic health effects may occur in exposed employees." OSHA has set exposure limits for about 600 chemicals. The substances in WRM which may pose health hazards to workers need to be assessed in terms of both reactivity and toxicity. Using OSHA's language and categorization, the following analyses should be made and protocols developed to assess worker safety for each type of WRM.

1. Classification of each WRM into the following OSHA reactivity categories: unstable, water, reactive, explosive, and pyrophoric;
2. Listing of the oral and dermal median lethal dose (LD_{50}) values and inhalation median lethal dose (LC_{50}) values of common substances in each type of WRM; and
3. Identification of WRM types that may have hazard characteristics in excess of those specified under HM-181, the USDOT guidelines for the transportation of hazardous materials which became effective October 1, 1993.

Step 11. Development of Monitoring and Maintenance Plans

The facilities in which WRM is used have various design configurations, overlie different hydrogeological and pedological zones, and traverse various climatic regions. In this analytical step, monitoring systems are developed to track the performance of the designed facility. At this stage, the monitoring systems are not actually built but configured on paper to produce critical data on both the structural and environmental performance of the facility when it is built. The maintenance plan is also developed using an estimated pattern of deterioration of the facility.

Steps 12 and 13. Preimplementation Life-cycle Cost Estimation

At this stage a comprehensive plan has been developed for the facility, including information on its structural design, required material quantities, logistics, construction techniques, and monitoring and maintenance plans. This step involves the estimation of the life-cycle cost of constructing and operating the facility in which the WRM is used. If the total cost of partial or complete replacement of traditional materials with WRM is excessively high, then the owner or operator can seek other materials consistent with the return loop from step 12 to step 4 in Fig. 4. Villalba et al. (2002) and Mrouch et al. (2001) have proposed quantitative methodologies for performing life-cycle cost assessments of WRM in construction and for other purposes. Ideally, the decision on WRM use should be based on benefit/cost ratio that takes into consideration the general benefits to society that result from recycling of wastes and the opportunity costs of using traditional construction materials. However, the benefits and costs linkages are difficult to completely quantify. Project owners generally ignore societal benefits and costs and focus on direct cost savings. When this is reasonable and acceptable, then the project would then be implemented in step 13.

Step 14. Monitoring, Maintenance and Periodic Assessment

Postconstruction monitoring is necessary for all constructed facilities. Performance monitoring provides data for confirmation or revision of operational costs and benefits as well as maintenance planning for the remainder of the service life of a facility. For WRM applications on which there is limited field experience, monitoring of field performance can provide data that can be used for tracking the correlation between preconstruction estimates of contaminant releases and material durability and actual performance of the WRM facility in the field. Monitoring of some field projects has provided useful data on the performance of facilities vis-à-vis laboratory test data. For the case of stabilized air pollution control (APC) residues (41% APC, 21% Portland cement, 3% Na_2CO_3, and 32% water, all weight percentages), Baur et al. (2001) used laboratory monolith leaching data to model field concentration of heavy metals from in situ blocks of 16 m^2 in surface area, at a site in Teuftal, Switzerland. Although modeled results showed reasonable agreement with field data, the concentrations of the heavy metals concerned (Cd, Co, Cu, Mn, Ni, Pb, Zn) were generally lower in the field leachate that would be expected on the basis of laboratory test data. This difference was attributed to solubility control in the laboratory and transport (diffusion) control on the leachability of some of the metals in the field.

The displacement of the target contaminant from the emplaced WRM component into the surrounding media can also be determined by sampling across the interface. A displacement (by leaching) is indicated by depletion in the concentration of the target contaminant in the region of the structural component proximal to the interface, and an increase in the concentration of the same contaminant in the bounding region of the surrounding media. Schreurs et al. (2000) used this approach to monitor stabilized coal fly ash under asphalt and the same material under sand cover at a site in Coloradoweg, Holland after 11 years of emplacement. For the asphalt covered ash (where leaching can be assumed to be diffusion dominated), the releases of Cr, Mo, S, V, and Zn were 31, 60, 15, 400, 183, and 130 mg/m^2, respectively. For the sand-covered ash (where leaching is predominantly induced by percolating water), the releases of the same elements were determined to be 300, 1,580, 1,100, 300, and 135 mg/m^2, respectively.

With reference to the scenario depicted in Fig. 3 and other field situations in which leaching occurs through internal intergranular flow of the leachant (presumably rain and snowmelt water) through an emplaced material, the differences in contaminant concentration between the internal portion and surrounding media of the structure can be highly affected by chemical buffering in ways that are absent in laboratory tests. For a large heap

(375 tonnes) of municipal solid waste (MSW) bottom ash near Paris, France (Freyssinet et al. 2002), monitoring of internal pore water for Cu, Pb, and Zn yielded concentrations of 42.7, 9.6, and 0.8 g/l, respectively. Similar measurements for the same metals in the leachate at the exit of the waste pile produced 10.2 g/l for Cu but nondetectable concentrations for Pb and Zn. At the outlet, exposure of the leachate to CO_2 resulted in carbonate precipitation which buffered the leachate and reduced Pb and Zn concentrations to very low (nondetectable) levels. Such factors need to be considered in environmental impact assessments of plans to use WRM in construction.

In the field, the leachant is supplied intermittently. It has been shown (Hertwich 2001) that steady-state models that address emission of substances from media that are exposed to the atmosphere (often categorized as level III fugacity models) can result in underestimation of emitted concentration for chemicals that have low values of Henry's law constant ($<0.01\ \mathrm{Pa\,m^{-3}\,mol^{-1}}$) because of the assumption that rainwater is supplied continuously. For WRM application, this realization applies mainly to uncapped and uncemented materials that often have significant internal fluid permeability. It is then advisable to use projected rainfall patterns to estimate contaminant release quantities for specific time segments within the overall service period of concern. The total quantity of the target contaminant can be integrated over the analytical period as a summation of the data for time segments as performed by Mudd and Kodikara (2000) for a coal ash storage pond within the Loy Yang Dump site in the Latrobe Valley of Victoria, Australia. This approach has been used for determination of contaminant concentration source terms for use in risk assessments to support waste containment projects and programs.

Approaches to Specifications and Regulations

The utilization of waste recycled materials (WRM) in construction is an international objective that is promoted by several state and federal agencies, the private sector, and nonprofit institutions. Most of these organizations view waste utilization as a means of minimizing waste disposal problems while developing physical infrastructure. To allay concerns about the prospective release of contaminants from WRM in service, while ensuring that excessive constraints are not placed to diminish the benefits that result from such large-scale recycling of materials, approaches that fall into one or more of the following options may be considered.

1. Specification of a maximum substitution rate of WRM for traditional materials in specific applications;
2. Specification of maximum allowable emission rates of specific substances from laboratory tests of specified protocols. In this case, any percent substitution of WRM can be used in construction provided that laboratory test-based emission limits are not exceeded; and
3. Specification of maximum allowable contaminant concentrations at sensitive or other selected points outside the WRM structure. This is the field performance specification that requires the implementation of a monitoring program. This approach is more amenable to integration into environmental (ecological) risk assessment frameworks.

Directions of Innovation and Conclusions

Apart from municipal waste sources and overburden/tailings excavated in mining operations, most WRM are produced in industrial processes that can be optimized to produce by-products that are suitable for other uses, including construction. Such an optimization can also be targeted at reducing the rate of generation of wastes. In a sense, the current reprocessing of wastes into construction products represents a move in this direction. Examples are the processing of fly ash into zeolites (Beretka et al. 1993; Querol et al. 1997; and Choi et al. 2001) and the conversion of solid wastes into a variety of products (Kocasoy et al. 1999). Essentially, by-products from one industry or plant can serve as raw materials for different production processes in another industry such that the rate of accumulation of wastes is reduced. The concept of by-product resource exchanges has been discussed by Lowe (1997). A possibility is the design of in-plant processes for waste minimization or elimination. This implies that a plant would deliberately produce its primary product as well as the secondary product which could be used in construction. The optimization of the processes to yield both categories of products would be driven by economic considerations. The environmental considerations would be covered by the fact that the price of the secondary product would be partly dependent on whether it can meet environmental requirements such as contaminant leachability specifications. An example (Comans et al. 2000; Crannell et al. 2000) of a process that could stabilize contaminants on a material is the use of soluble phosphate to stabilize heavy metals on municipal solid waste combustion ash particles.

In the larger picture, large-scale waste recycling in construction needs to be considered as a part of overall materials/energy management systems that can be operated using the principles and tools of industrial ecology. Some economic models (Nakamura 1999; Di Vita 2001), conceptual systems depicting material flows (Van Berkel and Lafleur 1997), and regional approaches (Wakeman and Themelis 2001; Korhonen 2001) have been proposed for materials management. The use of comprehensive approaches is necessary to account for the complex web of factors that apply to materials use for sustainable development of society.

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