Available online at www.sciencedirect.com

SCIENCE DIRECT®

Ecological Modelling 174 (2004) 175–189

ECOLOGICAL MODELLING

www.elsevier.com/locate/ecolmodel

Analysis of energy footprints associated with recycling of glass and plastic—case studies for industrial ecology

V. Krivtsov [a,*], P.A. Wäger [b], P. Dacombe [a], P.W. Gilgen [b], S. Heaven [a], L.M. Hilty [b], C.J. Banks [a]

[a] *Department of Civil & Environmental Engineering, University of Southampton, Highfield, Southampton SO17 1BJ, UK*
[b] *Swiss Federal Laboratories for Materials Testing and Research (EMPA), Lerchenfeldstrasse 5, CH-9014 St. Gallen, Switzerland*

Abstract

Roundput [Int. J. Sustainable Dev. World Ecol. 8 (2001) 29] is one of the most important principles of the development of both natural and industrial ecosystems, and is especially important for analysis of an ecosystem's dynamics and overall functioning, as it is related to an extent to which energy and matter are recycled and used in a cascade-type operation. Here we argue, using two modelling case studies from the UK and Switzerland, that increasing recycling rates for plastic and glass would improve the energy budget of waste management programmes, and, therefore, benefit the corresponding industrial ecosystems. In the first case study we show that the major source of energy savings from glass recycling is through increased use of cullet in glass manufacture (5.4% reduction in total energy consumption with 100% glass recycling when compared to the present-day situation). In terms of energy consumption, recycling is the preferred waste management option, even if a large proportion of the recycled glass is diverted for use as aggregates. Further energy savings could be achieved by introduction of a city-wide kerbside collection scheme, which would result in an estimated maximum reduction (100% recycling rate) of 7.6% in energy consumption for processing of the Southampton household glass wastes. In the second case study we compare the situation in which all wastes are burnt at a MSWI plant with two scenarios assuming that 8.1% of the plastic is diverted into a cement kiln (mixed plastics; scenario 1) or a mechanical recycling plant (polyethylene, polypropylene, polystyrene; scenario 2). The resulting net primary energy consumption values for both scenario 1 (5.85E8 MJ or 60% relative to the reference scenario) and 2 (7.46E8 MJ or 76.6% relative to the reference scenario) use less primary energy than the reference scenario (9.74E8 MJ). This means that, from the point of view of resource consumption, the diversion of plastics waste away from the MSWI plant has a beneficial effect. Therefore, the increased recycling of glass and plastic would benefit the industrial ecosystems in terms of energy savings. This is similar to the patterns observed in most natural ecosystems, and a careful consideration of this similarity within a framework of industrial ecology should help to reduce the conflict between the two systems.

Keywords: Industrial ecosystems; Natural ecosystems; Glass; Plastic; Recycling; Roundput

1. Introduction

It has previously been argued (Korhonen, 2001) that natural and successful industrial ecosystems share four basic principle conditions of their develop-

---

[*] Corresponding author. Tel.: +44-2380-593013; fax: +44-2380-677519.
E-mail addresses: vk2@soton.ac.uk, e96kri69@netscape.net (V. Krivtsov), patrick.waeger@empa.ch (P.A. Wäger).

0304-3800/$ – see front matter © 2004 Elsevier B.V. All rights reserved.
doi:10.1016/j.ecolmodel.2004.01.007

ment, namely roundput, diversity, locality and gradual change. The first one (i.e. roundput) is especially important for analysis of an ecosystem's dynamics and overall functioning, as it is related to the extent to which energy and matter are recycled and used in a cascade-type operation. In most natural ecosystems the main (and often the only considerable) input to the system is the solar energy. Wastes from one trophic level normally serve as a food resource for another trophic level, and, although much energy is lost through dissipation, ecosystems tend to evolve to optimise recycling and energy cascading in the food chain (Fath and Patten, 1998; Patten, 1992).

Traditionally, human society has developed without the necessary due respect to the rules and processes governing the stability of its environment. In particular, much of the recent (i.e. last few centuries) technological progress has been based on non-renewable energy sources, and the surrounding environment has commonly been used (and abused) to dispose of unwanted materials, i.e. wastes (Odum and Odum, 1976). In the last century, however, rapid economic and industrial growth has increasingly begun to be restricted by energy supplies and abatements for pollution control (Meadows et al., 1972, 1992). Consequently, growing environmental problems and associated public concern, together with the development of science (in particular natural and social sciences) and technology, have led to the emergence and elaboration of the concepts of sustainable development (Pearce et al., 1989; Meadows et al., 1992) and industrial ecology (Duchin, 1991; Frosch, 1994, 1995, 1998; Graedel and Allenby, 1995). The latter is particularly relevant to this paper, and is based on the analogy with natural ecosystems.

Industrial ecology is based on the analogy between natural and industrial ecosystems (i.e. as regards recycling and cascading networks), and aims to facilitate the development of industrial recycling and cascading cooperative systems by minimising the energy consumption, generation of wastes, emissions, and input of raw materials (Korhonen, 2001). Here we present two modelling case studies from the UK and Switzerland, to show that increasing recycling rates for plastic and glass would improve energy budgets of waste management programmes, and, therefore, benefit the corresponding industrial ecosystems.

2. Southampton glass case study

Prudent use of energy and raw materials is fundamental to sustainable development and will require a step-change in resource productivity. Working towards the goals of sustainable development, the EU Landfill Directive and national strategy documents have set ambitious targets and deadlines for diversion and recycling. Progress in waste management is hampered, however, by the lack of methods to identify and promote sustainable practices. One of the serious concerns is that benefits from recycling may be offset by the excessive energy consumption associated with waste management, and there is therefore a need to develop tools for rational evaluation and comparison of alternatives for the collection, separation and processing of waste fractions.

The objectives of this case study are:

- to understand, quantify and model energy usage associated with the collection, separation, processing and disposal of household glass waste;
- to produce an energy and materials balance that can be used for evaluation and comparison of different alternatives and combinations of options for waste management.

The mathematical model presented here was constructed to estimate the energy footprint of the current waste management practice, and to allow analysis of alternative choices and combination of waste management options. The work is based on Southampton in the UK, but the methods and findings can be applied to other areas by modifying the input data.

2.1. Model description

The model (Fig. 1) adopts a linear modelling approach routinely used in waste management and LCA models (Abou Najm et al., 2002a,b; Bjorklund et al., 1999; Cosmi et al., 2000; Eriksson et al., 2002; Everett and Modak, 1996; Gielen and Moriguchi, 2002; Solano et al., 2002). The processes considered by the present model start from the point where glass becomes waste and follow it through until disposal and/or reprocessing. Crucially, the model not only takes into account the energy consumed during processing/disposal, but also transport energy consumption. It consists of a number of interlinked submodels,

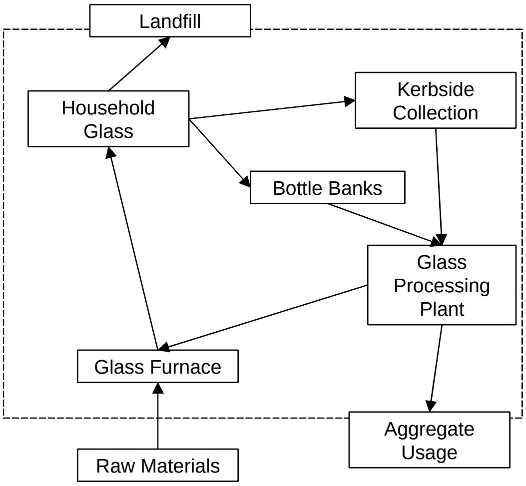


Fig. 1. Summary diagram of the processes considered in the glass model. Dashed line indicates system boundaries.

and each submodel simulates a separate stage in the overall process. The most important submodels are refuse collection, landfill transfer, kerbside collection, stage 1 transport (household to bottle bank), stage 2 transport (bottle bank to processing plant), glass processing plant, cullet transfer, and glass manufacture.

2.1.1. *Refuse collection and landfill transfer*

The amount of glass in the refuse for the present situation has been estimated from compositional data (M.E.L. Research Ltd., 1999) and actual monthly refuse tonnages (figures supplied by Southampton City Council). We defined the system boundaries for this analysis in such a way that the energy consumption is determined not just for the glass fraction, but for the total amount of refuse collected. This has been done because the glass waste is not separated from the other commingled wastes in the refuse (all the wastes are transported to the landfill site and disposed of there), and also because in future we intend to use the glass model as a module in a larger model considering the energy footprint of all types of wastes in Southampton.

2.1.2. *Kerbside collection*

Although currently there is no kerbside collection scheme in Southampton, one might be introduced in the future. In view of the relatively low amounts of glass in domestic refuse, in the analysis presented here it is assumed that the scheme would operate on a fortnightly basis. It is also assumed that bottle bank usage will be negligible in the areas covered by kerbside collection, and that the area covered by bottle banks will decrease proportionally to the increase in kerbside coverage. It is also assumed that the recycling rate via kerbside collection and via bottle banks is the same.

2.1.3. *Stage 1 transport*

This submodel accounts for energy consumption associated with the transfer of used glass bottles to bottle banks. It should be noted that in our model only the car journeys made specifically for the purpose of glass recycling are considered. The incidental trips (e.g. visiting a bring site at a supermarket whilst doing the weekly shopping), and the metabolic energy spent by those householders who travel on foot, are outside the system boundaries of the model presented here.

The percentage of trips made specifically for recycling was estimated as described by Edwards and Schelling (1999). For the current bottle bank density in Southampton, approximately 28% of trips are made by car specifically for recycling, with an average return-journey trip of 0.267 miles. The average fuel consumption was estimated as recommended by the European Commission (Commission of the European Communities, 1993) as 141.4 g/mile, and takes into account the ratio of petrol and diesel vehicles. From these data, the annual energy consumption for transporting the glass from the household to the bring-site can be calculated for the given site density and glass recycling rate.

It should be noted that a considerable proportion of glass wastes is recycled in the bottle banks positioned at household waste recycling centres (HWRC). To account for the differences in parameter values (e.g. distances travelled), the subprogrammes incorporating the code for the stage 1 transport and stage 2 transport (described below) submodels are called twice from the main programme, firstly with the parameter values characteristic of ordinary bottle banks, and secondly with the parameter values characteristic of HWRC. The values of energy consumption returned by the consumer transport submodel are then divided by three, as the model assumes that glass wastes constitute approximately 1/3 of the wastes brought to HWRC during any car trip to the site.

2.1.4. Stage 2 transport

This submodel describes transfer of the collected glass to the glass processing plant. In Southampton, a new processing plant at the city's Docks began operating at the beginning of 2003, and the majority of the cullet processed here will be transferred out by ship. Transfer of the bottle banks to the processing plant takes place by a skip lorry, and the banks are emptied at the processing plant.

The average distance to the processing plant was determined using a route planner to measure the distance from the plant to several points at the outer boundary of Southampton. It was then assumed that the average distance to the plant is half of the average maximum distance, giving a return-journey trip of 7.34 miles. Overall handling time necessary to process a bottle bank was estimated as 20 min. Using specific fuel consumption figures (Edwards and Schelling, 1999) and handling time of 20 min, an energy consumption per bottle bank collection was estimated as 267.2 MJ. Then assuming 348 collections per year (an average figure for the actual number of collections made in Southampton since 2000), the annual energy consumption associated with this stage of transport can be calculated as 93 GJ.

2.1.5. Glass processing plant

Despite what it name implies, the Midland glass processing plant is conveniently positioned in the Southampton docks within the city boundary, thus rendering the energy expenditure for local transport (see above) relatively low. At the plant, the glass is crushed to produce cullet, and sorted using laser-separation equipment capable of sorting 40 t/h. Plant energy consumption for these processes has been estimated from data on electrical usage, allowing for an efficiency of electrical production of 30.2% calculated as described by McDougall and coauthors (McDougall et al., 2001) using the data available on the internet (Department of Trade and Industry Website, 2002). It should be noted that energy expenditure due to fuel consumption of site vehicles (used for miscellaneous operations) is relatively low compared to crushing/sorting, and has therefore been left outside the boundaries of this research.

2.1.6. Cullet transfer

The output from the glass processing plant is transferred first by ship and then by truck. The calculations were based on the assumption that the destination port is Goole, and the distance of a truck travel was estimated as an average from Goole to the different glass manufacturing plants located in the north of England (Enviros, 2002). For the analysis presented here we assumed a ship load of 1200 t (Associated British Ports website, 2003), whilst the ship fuel consumption has been estimated from real data supplied by Stephenson Clarke Shipping Ltd. (Hindmarch, 2003).

2.1.7. Glass manufacture

Use of glass cullet not only helps to save raw materials, but also considerably reduces the energy required to produce the melted glass (BUWAL 250/II, 1998). Currently, the model calculates the energy required to produce the city's average annual requirement for glass, and assumes that the cullet level will depend solely upon the amount of glass recycled in Southampton. It has also been assumed that the use of cullet will only influence the melting energy, but will not affect the other stages of glass production, or peripheral electrical usage. It is assumed that internal cullet (e.g. breakages during production) always constitutes 17%, and that the specific energy consumption (SEC) for melting with 39.5% level of cullet is 4.97 GJ/t (ETSU, 2000). Given that energy consumption for melting represents only 71% of the total energy consumed at this stage (ETSU, 2000), the overall energy usage associated with glass manufacturing at this level of cullet can be estimated as 7 GJ/t, while for other levels of cullet energy consumption is estimated taken into account a 2.5% decrease in the required melting energy for every additional 10% of cullet.

2.1.8. Model implementation and validation

The current version (vb 12) of the model was compiled in Visual Basic. The model consists of the main programme, and a number of subprogrammes incorporating the code for specific submodels described above. These subprogrammes are called from the main programme, and the total energy consumption is subsequently calculated by summation of the specific components returned by the submodels.

The model presented here has been validated as follows. The simulations in separate submodels were verified by independent calculations using Microsoft Excel spreadsheets. Credibility of the assumptions made, the input data used, and the results obtained have been discussed with the relevant representatives of Hampshire County Council, Southampton City Council, and a waste management company ONYX Environmental.

2.2. Results

The model was run in interactive mode to simulate the current situation and a number of 'What if' scenarios, and in automatic mode to perform sensitivity analysis on selected parameters. It was found that the energy consumption associated with handling and processing the Southampton glass amounts to between 60,000 and 70,000 GJ per year. With the current recycling rate of 25.16% the approximate estimate of the overall energy expenditure was 68,600 GJ per year. Among the processes considered, the glass furnace proved to be by far the most energy-demanding stage (Fig. 2). The main part of the overall energy consumption is in the manufacturing process (cf. Figs. 2 and 3), and the savings made here through increased use of cullet offset any increases in transport and processing energy consumed elsewhere. Therefore, any energy savings due to the increased cullet level (caused by increased recycling) would easily outweigh the increase in energy consumption related to glass collection, transport, and processing at the processing plant. For example, increasing the recycling rate from 25.16 to 100% would lead to a total energy consumption of approximately 64,900 GJ, thus resulting in 5.4% energy savings (see Figs. 2 and 3).

There are a number of options that may help to increase the recycling rate. One factor that affects recycling rates is the number of bottle banks available: in theory, increasing bottle bank site density should increase glass recycling rates, since it makes recycling easier. However, an increased site density would also lead to a number of logistical problems, including increased expenditure on bottle bank maintenance, difficulties as regards allocation of space for new bottle banks, etc. Introduction of a kerbside collection scheme would also lead to increased recycling. Switching to 100% coverage of a kerbside scheme would result in a considerable decrease in energy consumption (Fig. 4). For a hypothetical value of 100% both for glass recycling rate and for kerbside coverage, the total energy consumption would be less than 63,400 GJ, resulting in a 7.6% decrease compared with the present situation. Furthermore, with the current model assumptions (see, however, discussion related to model limitations) both an increase in kerbside coverage and in recycling rate would invariably result in a decrease of total energy consumption per tonne of glass recovered (Fig. 5).

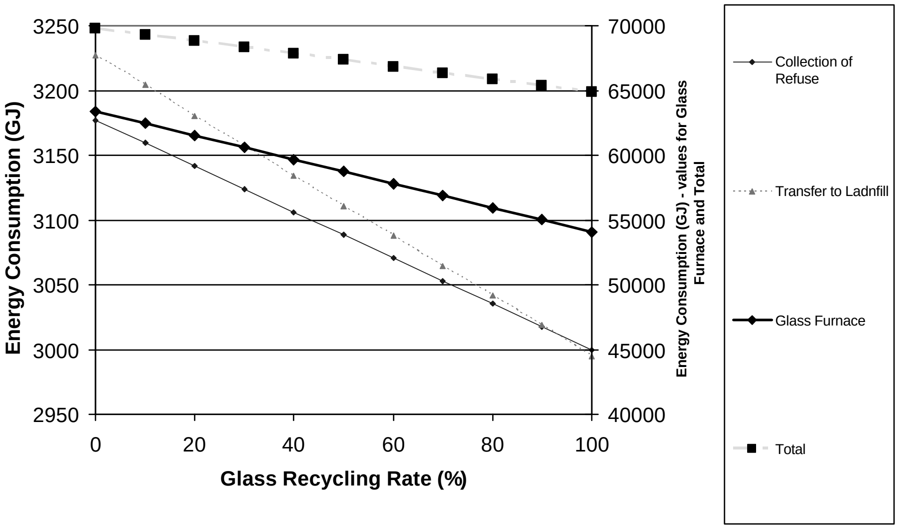


Fig. 2. The effect of glass recycling on energy consumption (major components).

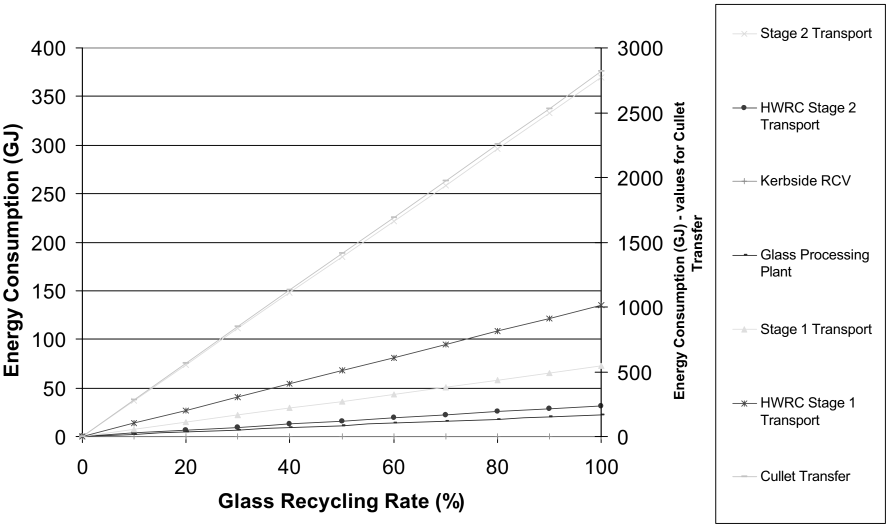


Fig. 3. The effect of glass recycling on energy consumption (other components).

It should be noted that if the distance to the glass processing plant were considerably increased, the energy savings in glass manufacture might be offset by the increased transportation requirements (Fig. 6). For excessive distances, however, use would be made of a waste transfer station (WTS) on the outskirts of the collection area. Here the skips carrying bottle banks would be emptied, and subsequently returned to their designated locations. Between the WTS and the processing plant glass wastes would, therefore, be transported without skips, thus dramatically increasing the load and overall efficiency of the process. This effect

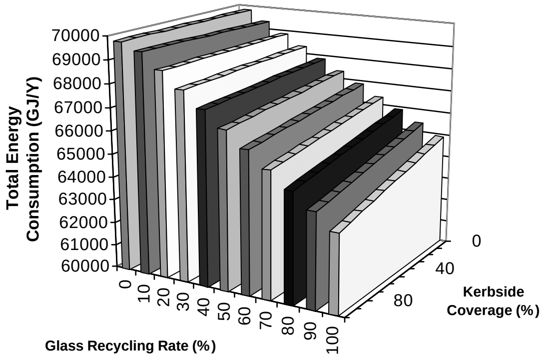


Fig. 4. A model response surface simulating the combined effect of recycling rate and kerbside collection scheme coverage on energy consumption associated with processing of glass wastes.

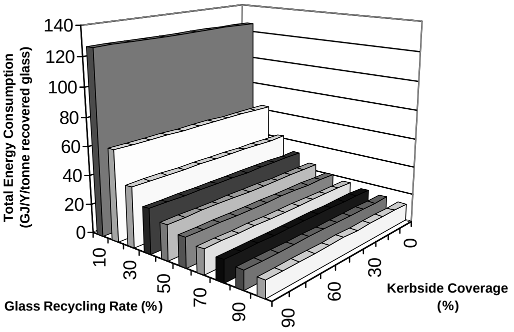


Fig. 5. Total energy consumption expressed per tonne of recovered material.

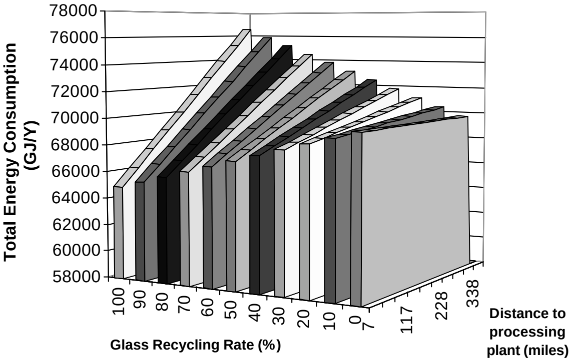


Fig. 6. Variation in energy consumption with distance to processing plant.

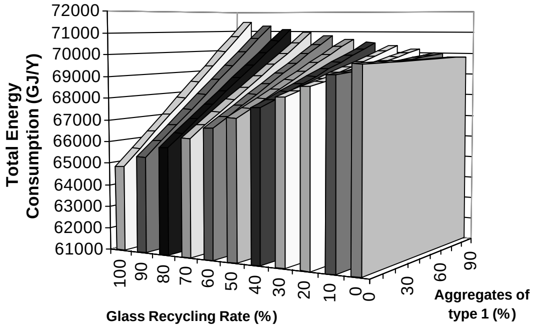


Fig. 7. Variation in energy consumption in relation to the amount of glass diverted for use as aggregates (transportation distance = 50 miles).

is primarily due to the considerable savings in energy consumption through the increased use of cullet in glass manufacture, compared to the relatively small amounts of energy consumed through collection, processing and transportation of the recycled glass.

Another scenario examined (Fig. 7) was the effect of diverting the cullet away from glass manufacturing to use as aggregates in, for example, road construction. Again, the results show that with a realistic assumption for the transportation distance it is not until high levels of diversion that the energy consumption becomes excessive.

2.3. Implications and limitations

In terms of energy consumption, recycling is the preferred waste management option, even if a large proportion (but not all!) of this recycled glass is diverted for use as aggregates. Hence, it is evident that increased glass recycling would benefit the industrial ecosystem in terms of energy savings. This is similar to most natural ecosystems, and this similarity will be explored in our further studies.

As any model is, by definition, a simplification of reality, a number of processes have been left outside the scope of the analysis presented here. One considerable limitation of the model is that it only takes into account household glass, whilst commercial and industrial glass wastes have been left outside the system boundary. We expect, however, that the patterns observed should hold for these types of wastes as well, as most of the processes accounted for by the present model are also applicable to the management of commercial and industrial wastes. Likewise, incineration was not included in the version used for the analysis reported here, because at the time of writing there is no incineration of Southampton wastes. If the model is to be applied for the scenarios that include the incineration option, this limitation would have to be addressed. However, we expect that the patterns observed (i.e. a decrease of the overall energy consumption with an increase in recycling rate) should hold in this case as well, as the calorific value of incinerated wastes is inversely related to their glass content (Papworth and Poll, 1991).

It should also be noted that in the analysis presented here we only consider as fully credible the range of recycling rates between 10 and 90%. Britain is currently far behind some other European countries, where recycling rates are often in the range of 60–80%, and there is a legislative pressure from the EU urgently to increase glass recycling in the UK. Hence, recycling rates lower than 10% would not be tolerated due to legislative constraints. On the other hand, very high recycling rates are not realistic, and may lead to a sharp increase in energy required to collect the glass recycled (Edwards and Schelling, 1999). This potential increase has not been incorporated into the model. Instead, it is hoped that relatively high recycling rates could be achieved by changing public awareness and attitudes towards recycling through education and media. Until then, however, the estimates at the top extreme of the recycling range considered should be treated with caution.

3. Plastic case study

As a consequence of the increasing amounts of municipal solid waste expected and observed since 1999, installation of additional municipal solid waste incineration (MSWI) capacity is planned for Switzerland. This intention has led to controversial discussions, in which plastics waste plays a major role.

In Switzerland, most plastics waste (more than 80% by weight) is incinerated in MSWI plants. Due to the amount of this waste which was estimated at 570,000 t in 1999 (SAEFL, 1999), its heating value (which exceeds the heating value of typical MSWI waste by a factor of approximately 3), and its potential for recycling or thermal recovery, plastics waste could be diverted from MSWI plants into cement kilns and mechanical recycling facilities. Hence, the necessity to install additional MSWI-capacity is questionable.

To answer the question of whether an additional diversion of plastics waste could be a suitable alternative to extending MSWI-capacity, EMPA was commissioned to develop a simulation system in cooperation with Rytec Inc. Münsingen and relevant Swiss waste management representatives. The modelling system developed allows us to simulate the diversion of plastics waste from an MSWI plant into thermal recovery and recycling facilities. Evaluation of the simulation results is carried out by comparison of the output of a given scenario with the output from a reference scenario.

3.1. Model description

The system—called EcoSolver IP-SSK—has been implemented using the system dynamics simulation software Powersim® Constructor (Powersim Corporation, 1996). Based on the results of the project ‘Dynamics of Waste Treatment’ (Widmer et al., 1998), it has been conceived as a system which allows simulation of the environmental and economic effects of possible future developments (scenarios) in regional plastics waste management for time periods up to 15 years. EcoSolver IP-SSK consists of an input module, a simulation module and an output module (see Fig. 8), which are briefly described below.

3.1.1. Input module

The input module allows the user to set input parameters—e.g. the expected development of the waste streams in the disposal system under study, the amounts of thermally recovered or recycled plastics waste as well as the transportation distances—according to a defined scenario (see Fig. 9).

3.1.2. Simulation module

The simulation module includes the core model and additional sub-models for an ecological and economic assessment. In the core model, the transportation, collection, sorting and treatment processes related to the disposal routes considered (incineration in MSWI plants, thermal recovery in cement kilns and mechanical recycling) are represented. As a database, indicators for processes and systems typically found in Switzerland have been used (specific energy consumption, specific emissions of CO_2, NO_x, Cd, Hg, COD, etc.). The central element of the core model is the incineration process in MSWI plants, which has been modelled in detail.

The ecological assessment of the disposal system is based on the problem-oriented CML method employed in Life-Cycle Impact Assessment (LCIA),

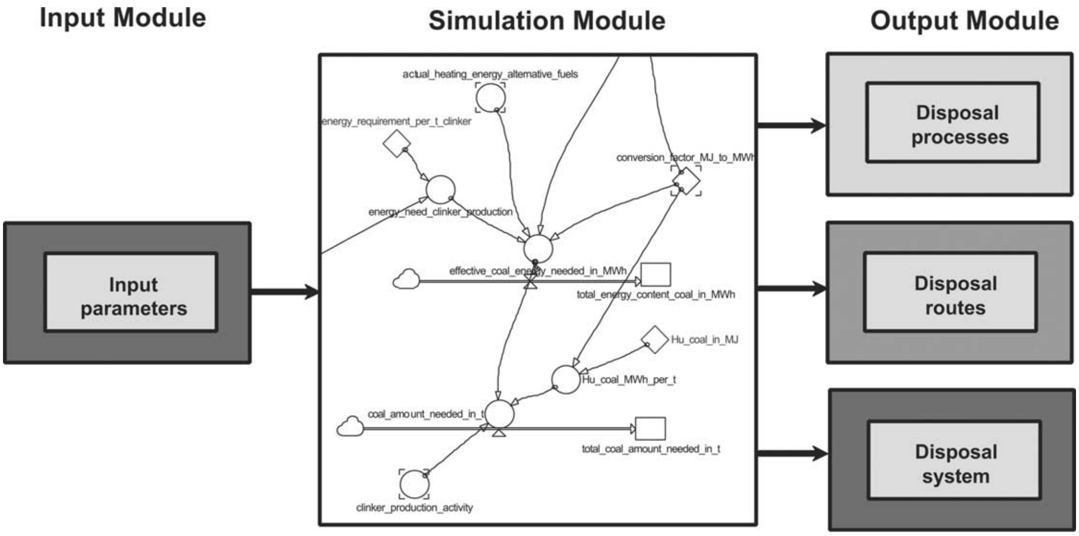


Fig. 8. General structure of EcoSolver IP-SSK.

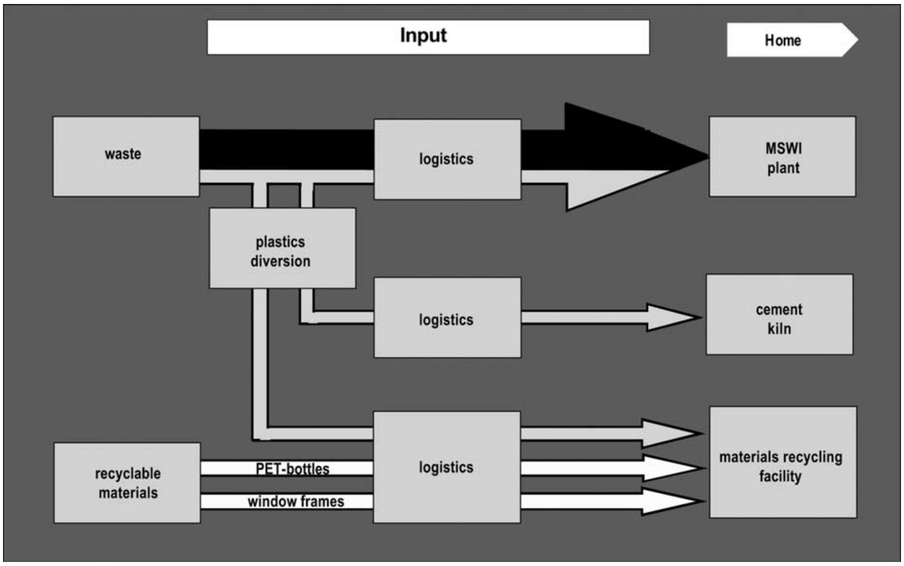


Fig. 9. Structure of the input module of EcoSolver IP-SSK.

and the 'basket of products' principle, which allows a fair comparison of scenarios with different outputs (Fleischer, 1994; Förster and Ishikawa, 1999; SAEFL, 1998a). In addition to the impact assessment categories (abiotic resource depletion, global warming, ozone layer depletion, etc.), environmental indicators are calculated in order to consider important environmental aspects that are not addressed by the CML method (amount of waste materials, heavy metal distribution into different compartments, etc.). The economic assessment is based on process-specific economic indicators. Currently, it is limited to single processes and disposal routes.

3.1.3. *Output module*

The output module shows the results of a simulation run on three different levels:

- processes (e.g. the incineration process in an MSWI plant, see Fig. 10);
- disposal routes (i.e. the sum of the disposal processes for each disposal option);
- entire disposal system under study.

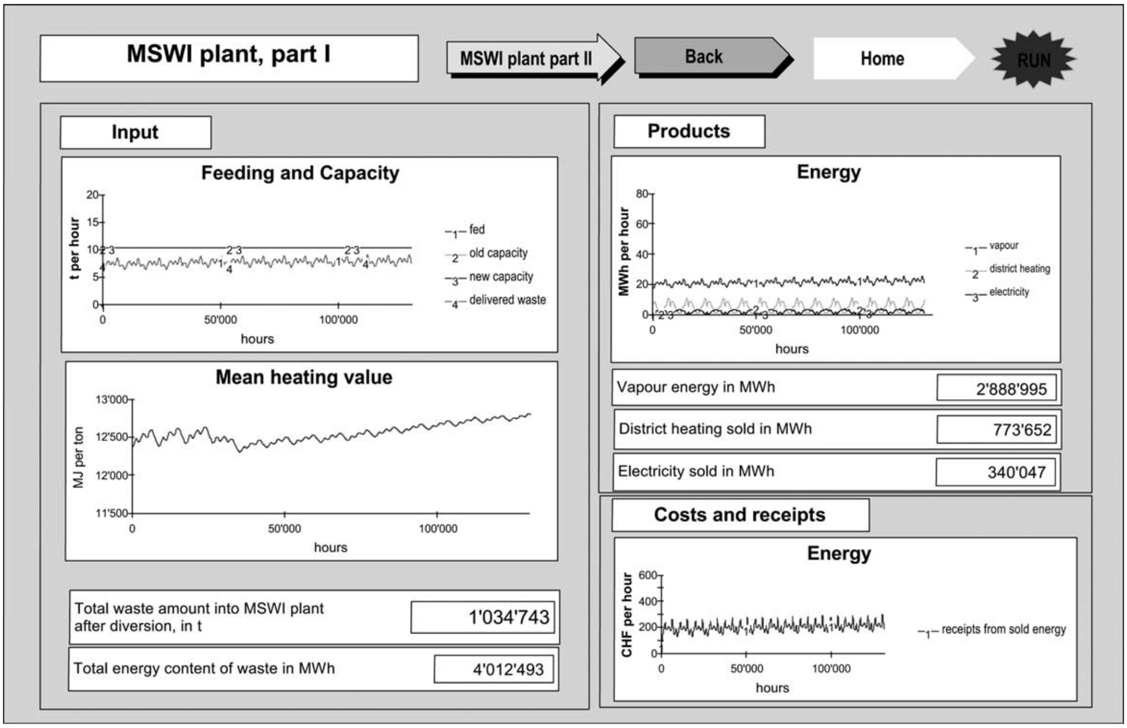
3.2. *Results*

The starting point for our analysis was the key question "What will happen if up to 100,000 t of plastics waste per year are taken out of the waste stream that is incinerated in Swiss MSWI plants, and fed into thermal recovery or mechanical recycling instead?". As MSWI plants differ from each other in technology and operating conditions, EcoSolver IP-SSK was applied to simulate thermal recovery and recycling options of plastics waste in a geographical area around a specific MSWI plant, the MSWI plant of the city of St. Gall in Switzerland (Wäger and Gilgen, 2000).

Two different scenarios were defined, each covering a time period of 15 years. Both scenarios assume that 8.1% of the plastics waste input into the MSWI plant is diverted either into a cement kiln (mixed plastics; scenario 1) or a mechanical recycling plant (polyethylene, polypropylene, polystyrene; scenario 2), beginning with year 4 of the simulation period (Table 1). When scaled up from the model region to Switzerland, the amounts of plastics waste diverted correspond to a

Table 1
Relative distribution of plastics waste for reference scenario, scenarios 1 and 2 at the end of the simulations

	MSWI plant (%)	Cement kiln	Mechanical recycling
Reference scenario	100	0	0
Scenario 1	91.9	8.1	0
Scenario 2	91.9	0	8.1



Total waste amount into MSWI plant after diversion, in t 1'034'743

Total energy content of waste in MWh 4'012'493

Vapour energy in MWh 2'888'995

District heating sold in MWh 773'652

Electricity sold in MWh 340'047

Fig. 10. Simulation output of EcoSolver IP-SSK on the process-level (MSWI-plant), scenarios 1 and 2.

total amount of about 100,000 t per year—the amount which was to be investigated according to the key question.

The major assumptions of the scenarios considered are:

- in year 10 of the simulation period both furnaces of the MSWI plant (with a total capacity of 66,000 t of waste per year) are replaced;
- plastics waste thermally recovered in the cement kiln replaces hard coal and complies with the standards for alternative fuels of the Swiss Agency for the Environment, Forests and Landscape (SAEFL, 1998b);
- due to its lower quality, the amount of recycled plastics waste needed to generate a new plastic product exceeds the amount of primary plastics material needed for the same amount of product by a factor of 1.1; nevertheless, there is a market for products from recycled plastics.

Scenarios 1 and 2 (described above) are each compared to a reference scenario, in which all plastics waste in the model region is incinerated in a MSWI together with other municipal solid waste. According to the basket-of-product principle implemented into EcoSolver IP-SSK, the product output has to be the same for the scenario and the reference scenario. For example, a reduced electricity generation in the MSWI plant, which results from plastics waste diversion into another disposal route, has to be compensated by electricity generation from conventional processes.

Validation of the results was carried out by sensitivity analysis of selected parameters and by discussion of the plausibility of simulation results with the stakeholders involved. In this paper we present the simulation results related to energy consumption.

3.2.1. Processes

EcoSolver displays the output of the simulation runs for each of the relevant processes considered in the three disposal routes, i.e. for transportation, pre-treatment and treatment. As an example, one of the two output frames for the process of treatment in the disposal route MSWI plant—the incineration in the MSWI plant—is shown in Fig. 10. The diversion of plastics waste, beginning at $t = 4$ years (26,281 h in Fig. 10), affects the mean heating value of the total municipal solid waste input into the MSWI plant: due to the high heating value of plastics waste (29,750 MJ/t), it leads to a significant reduction of the mean heating value of the municipal solid waste incinerated in the MSWI plant. Without plastics waste diversion, the heating value would continuously increase starting from the beginning of the simulation period, as it has been assumed that the use of plastic products, e.g. for food packaging, and hence the relative amount of household plastics waste in the municipal solid waste input into the MSWI plant will continuously increase.

The mean heating value of municipal solid waste incinerated affects the amount of steam vapour produced in the MSWI plant, which in turn has an influence on electricity and district heating sold to the municipality and hence the receipts of the MSWI plant. As a consequence of the reduction of the mean heating value through diversion of plastics waste, steam production as well as electricity and district heat generation decrease, resulting in lower receipts for the MSWI plant.

3.2.2. *Disposal system*

Based on the inventory data implemented, EcoSolver IP-SSK calculates energy consumption and emissions for the processes relevant to a scenario. Due to their known environmental relevance in connection with plastics waste recycling and recovery, the following inventory data related to the resource consumption were included in the calculations: feedstock energy, aggregated renewable primary energy sources, aggregated non-renewable primary energy sources, crude gas, brown coal, hard coal, crude oil, and uranium from ore.

The data were taken from the study 'Life Cycle Inventories for Packagings' published by the Swiss Agency for Environment, Forests and Landscape (SAEFL, 1998c). They do not include infrastructure. Table 2 shows which inventory data have been used for the processes considered, allocated to the three disposal routes.

Table 3 presents the simulation results related to primary energy consumption, as the sum of renewable and non-renewable primary energy, each for the reference scenario (in the reference scenario, plastics waste is incinerated in an MSWI plant together with the municipal solid waste of the model region), for scenario 1 and for scenario 2.

In Table 3, primary energy use was first calculated without consideration of the basket-of-products principle (these uncompensated values are given in the second column of Table 3). According to this principle, a decrease in electricity and district heating generation caused by the diversion of 8.1% of plastics waste has to be compensated by standard electricity generation and standard heat generation. At the same time, the difference between the output for hard coal as a standard fuel and plastics as an alternative fuel has to be considered. Hence, for scenarios 1 and 2, several

Table 2
Inventory data used for the calculation of the outputs of EcoSolver IP-SSK in scenarios 1 and 2

	MSWI plant	Cement kiln	Mechanical recycling
Treatment	• Direct emissions from municipal solid waste incineration (transfer coefficients) • Electricity generation • Ammonia (NH_3), sodium hydroxide (NaOH) and quicklime (CaO) production	• Direct emissions for fuel combustion	• Emission factors for plastics waste recycling
Pre-treatment		• Diesel extraction and combustion • Electricity generation	• Diesel production extraction and combustion • Electricity generation
Transportation	• Transportation by car, delivery van, garbage truck, lorry	• Transportation by lorry	• Transportation by lorry
Substituted processes according to basket-of-products principle	• Standard electricity generation (Swiss model) • Standard heat generation (from 50% crude gas and 50% crude oil)	• Hard coal extraction and combustion	• Standard production of HDPE-pellets, LDPE-pellets, PP-pellets, HIPS and GPPS

Table 3
Primary energy use for the scenarios considered in the plastic case study

	Primary energy without compensation (MJ)	Primary energy compensation for substitution (MJ)			Primary energy with compensation (MJ)	Primary energy relative to reference scenario (%)
		MSWI	Cement kiln	Mechanical recycling		
Reference scenarios	9.74E8	0	0	0	9.74E8	100
Scenario 1	9.77E8	+2.48E8	−6.40E8	0	5.85E8	60.0
Scenario 2	1.06E9	+2.48E8	0	−5.58E8	7.46E8	76.6

compensations had to be made as detailed below. The last two columns of Table 3 show the values that take these compensations into account.

For scenario 1, the difference between the higher primary energy use for electricity and district heat generation from standard processes and the lower primary energy use for electricity and district heat generation in the MSWI plant by plastics waste has been added. Likewise, the difference between the higher primary energy use resulting from hard coal as a fuel and the lower primary energy use from using plastics waste as an alternative fuel in the cement kiln has been subtracted as a bonus.

For scenario 2, again, the difference between primary energy use for electricity and district heat generation from standard processes and the lower primary energy use for electricity and district heat generation in the MSWI plant by the diverted amount of plastics waste has been added. Likewise, the difference between primary energy use resulting from standard production of HDPE-pellets, LDPE-pellets, PP-pellets, HIPS and GPPS and plastics recycling has been subtracted as a bonus.

The resulting net primary energy values for the reference scenario, scenarios 1 and 2 show that both scenario 1 (5.85E8 MJ energy consumed, or 60.0% relative to the reference scenario) and scenario 2 (7.46E8 MJ energy consumed, or 76.6% relative to the reference scenario) use less primary energy than the reference scenario (9.74E8 MJ). This means that, from the point of view of resource consumption, the diversion of plastics waste away from the MSWI plant has a beneficial effect.

A comparison between scenarios 1 and 2 shows that there is a higher reduction of the primary energy amount for a diversion into cement kiln (5.85E8 MJ or 60.0% relative to the reference scenario) than for a diversion into mechanical recycling (7.46E8 MJ or 76.6% relative to the reference scenario).

It should be noted, however, that scenario 2 additionally ‘generates’ a feedstock energy amount of 7.44E8 MJ, which can still be used, e.g. for thermal recovery in a cement kiln after re-use of the recycled plastics. From the perspective of primary energy use, this makes mechanical recycling the most beneficial of the three disposal/recycling options considered.

3.3. Implications and limitations

Simulation results with EcoSolver IP-SSK, which in addition to the issues considered above also included calculations of cost and environmental impacts, have led to the conclusion that specific strategies for diversion of plastics waste from industry should be developed and investigated. Subsequently, an expert group was initiated and a parliamentary initiative addressed to the Swiss Government (UREK-N, 2001). The expert group, which is chaired by the Swiss Agency of Environment, Forests and Landscape (SAEFL), has, among other actions, commissioned a detailed investigation of the additional diversion potential for industry plastics waste in Switzerland and invited relevant stakeholders to participate in the process of developing diversion strategies.

It is necessary to point out that the software used and the analysis carried out were both subject to a number of limitations. For example, it should be noted that recycled plastics usually have lower quality than plastics from primary raw materials and that the relatively low price of plastics from primary raw material does not encourage recycling.

The major limitation, however, stems from the assumptions incorporated in the analysis and from uncertainties associated with the input data, in particular as regards their future values. Hence, our future research will focus on the assessment of parametric uncertainty and sensitivity as well as on validation and robustness tests. In addition, we are also planning to improve the layout of EcoSolver IP-SSK by increasing transparency through clarification of model structure and user-interface, e.g. by a more strict modularisation.

4. Discussion

Analysis of multicomponent systems is never straightforward, and is greatly aided by application of simulation modelling techniques (Krivtsov et al., 2000; Krivtsov, 2001). Complex interplay among system components has previously been taken into account in a number of waste management and industrial ecology studies (Abou Najm et al., 2002b; Adamov et al., 1999; Bjorklund et al., 1999; Clift, 1998; Cosmi et al., 2000; Duchin, 1992; Edwards and Schelling, 1999; Eriksson et al., 2002; Everett and Modak, 1996; Gielen and Moriguchi, 2002; Hansen et al., 2002; Hui et al., 2003; Korhonen, 2001; Korhonen et al., 2001; Linton et al., 2002; Snakin and Korhonen, 2002). The results presented in this paper appear to be in good agreement with the previous research.

Higher recycling rates correspond to higher system roundput. It is evident that increased recycling (and, therefore, increased roundput) would benefit the industrial ecosystem in terms of energy savings. This is similar to the patterns observed in most natural ecosystems (Snakin and Korhonen, 2002), and industrial development is therefore likely to benefit from a careful consideration of these similarities, e.g. by optimising energy consumption as opposed to cost. It should be noted that in nature system development relies on real non-monetary values, e.g. on material and energy fluxes as opposed to abstract exchange rates (Korhonen, 2001). Unlike natural ecosystems, the development of industrial ecosystems is usually driven by economic costs and benefits, which hardly ever account properly for environmental issues. However, if humanity is to achieve a real long-term sustainability of its development, then material and energy values should routinely be used as primary optimisation criteria for the development of industrial ecosystems. Other optimisation criteria may include measures of diversity, locality, and the rate of change (Korhonen, 2001).

As already mentioned, the results for the high recycling rates are rather overoptimistic, and may require considerable changes not only in waste management systems, but also in people's private and social habits. The latter are outside the scope of industrial, but are within the scope of human ecology (Duchin, 1992), and the analysis presented here may be refined by introducing further fine details of human behaviour and decision making. It should also be noted that the modelling analyses presented here were carried out for the specific conditions of the investigated industrial ecosystems. Hence, although the results obtained in this study are likely to be applicable to a wide range of other geographical locations and materials, in each new application they should be treated with caution. It is worth pointing out that recycling projects may result in environmentally undesirable side effects, e.g. increased energy consumption by consumer transport and due to road maintenance at high recovery rates of recycled materials; undesirable by-products such as de-inking sludge in paper recycling; etc. Hence, in each case the potential energetic benefits of recycling should be assessed with the aid of mathematical tools taking into account the specific conditions of an industrial ecosystem under consideration.

Acknowledgements

B. Page and H.J. Staines are kindly thanked for their insightful comments on style and content of the manuscript. This work has also benefited from discussions with B. Lisney, K. Riley, A. Richardson, B. Beecroft. The authors wish to thank Rytec Inc. Münsingen and the stakeholders of the Swiss waste management sector involved in the development of the Ecosolver simulation system. Southampton City Council (SCC) and Hampshire County Council (HCC) are thanked for providing data for the glass case study and valuable discussion. The research presented here was financially supported by Biffaward, Onyx, Environmental, HCC, SCC, BOC foundation, the Swiss National Science Foundation, the Swiss Agency for the Environment, Forests and Landscape (SAEFL), the Association of Plastics Manufacturers in Europe

(APME), the Swiss Plastics Association (KVS), the Swiss PVC Association (PVCH), PET-Recycling Switzerland (PRS), the working group 'New Fuels in Cement Industry' of the Swiss Cement Kilns, Poly Recycling Inc. and the Swiss Foundation for the Reintegration of Plastic Materials (SSK).

References

Abou Najm, M., El-Fadel, M., Ayoub, G., El-Taha, M., Al-Awar, F., 2002a. An optimisation model for regional integrated solid waste management I. Model formulation. Waste Manag. Res. 20, 37–45.

Abou Najm, M., El-Fadel, M., Ayoub, G., El-Taha, M., Al-Awar, F., 2002b. An optimisation model for regional integrated solid waste management II. Model application and sensitivity analyses. Waste Manag. Res. 20, 46–54.

Adamov, E.O., Ganev, I.K., Lopatkin, A.V., Orlov, V.V., Smirnov, V.S., 1999. Self-consistent model of nuclear power development and fuel cycle. Atomic Energy 86, 337–344.

Associated British Ports website, 2003. News Release 21st March.

Bjorklund, A., Dalemo, M., Sonesson, U., 1999. Evaluating a municipal waste management plan using ORWARE. J. Cleaner Prod. 7, 271–280.

BUWAL 250/II, 1998. Life Cycle Inventories for Packagings, vol. II. Swiss Agency for the Environment, Forests and Lansdscape (SAEFL), Berne.

Clift, R., 1998. Engineering for the environment: the new model engineer and her role. Process Saf. Environ. Prot. 76, 151–160.

Commission of the European Communities, 1993. CORINAIR Working Group on Emissions Factors for Calculating 1990 Emissions from Road Traffic, 1. Office for Official Publications, Luxembourg.

Cosmi, C., Cuomo, V., Macchiato, M., Mangiamele, L., Masi, S., Salvia, M., 2000. Waste management modeling by MARKAL model: a case study for Basilicata Region. Environ. Model. Assess. 5, 19–27.

Department of Trade and Industry Website, 2002. Digest of United Kingdom Energy Statistics.

Duchin, F., 1991. Prospects for environmentally sound economic-development in the north, in the south, and in north-south economic-relations—the role for action-oriented analysis. J. Clean Technol. Environ. Sci. 1, 225–238.

Duchin, F., 1992. Industrial input–output-analysis—implications for industrial ecology. Proc. Natl. Acad. Sci. U.S.A. 89, 851–855.

Edwards, D.W., Schelling, J., 1999. Municipal waste life cycle assessment part 2: transport analysis and glass case study. Trans. IChemE 77 (Part B), 259–274.

Enviros, 2002. Recycled Glass Market Study & Standards Review. WRAP, Banbury.

Eriksson, O., Frostell, B., Bjorklund, A., Assefa, G., Sundqvist, J.O., Granath, J., Carlsson, M., Baky, A., Thyselius, L., 2002. ORWARE—a simulation tool for waste management. Resour. Conserv. Recycling 36, 287–307.

ETSU, 2000. Energy Efficient Environmental Control in the Glass Industry (revised). Good Practice Guide 127. ETSU, Didcot.

Everett, J.W., Modak, A.R., 1996. Optimal regional scheduling of solid waste systems. 1. Model development. J. Environ. Eng.-Asce. 122, 785–792.

Fath, B.D., Patten, B.C., 1998. Network synergism: emergence of positive relations in ecological systems. Ecol. Model. 107, 127–143.

Fleischer, G., 1994. Methodik des Vergleichs von Verwertungs-/ Entsorgungswegen im Rahmen der Ökobilanz. Abfallwirtschafts J. 6, 697–701.

Förster, R., Ishikawa, M., 1999. In: Barrage, A., Edelmann, X. (Eds.), The Methodologies for Impact Assessment of Plastics Waste Management Options—How to Handle Economic and Ecological Impacts? R'99 Geneva, Switzerland, pp. 220–225.

Frosch, R.A., 1994. Industrial ecology—minimizing the impact of industrial-waste. Phys. Today 47, 63–68.

Frosch, R.A., 1995. The industrial ecology of the 21st-century. Sci. Am. 273, 178–181.

Frosch, R.A., 1998. Industrial ecology and sustainability. Am. Sci. 86, 2–2.

Gielen, D.J., Moriguchi, Y., 2002. Waste benefits of CO_2 policies in Japan. Waste Manag. Res. 20, 2–15.

Graedel, T.E., Allenby, B.R., 1995. Industrial Ecology. Prentice Hall, New Jersey.

Hansen, Y., Notten, P.J., Petrie, G., 2002. A life cycle impact assessment indicator for ash management in coal-based power generation. J. S. Afr. Inst. Mining Metall. 102, 299–306.

Hindmarch, G., 2003.

Hui, I.K., Li, C.P., Lau, H.C.W., 2003. Hierarchical environmental impact evaluation of a process in printed circuit board manufacturing. Int. J. Prod. Res. 41, 1149–1165.

Korhonen, J., 2001. Industrial ecosystems—some conditions for success. Int. J. Sustainable Dev. World Ecol. 8, 29–39.

Korhonen, J., Wihersaari, M., Savolainen, I., 2001. Industrial ecosystem in the Finnish forest industry: using the material and energy flow model of a forest ecosystem in a forest industry system. Ecol. Econ. 39, 145–161.

Krivtsov, V., 2001. Study of cause-and-effect relationships in the formation of biocenoses. Russian J. Ecol. 32, 230–234.

Krivtsov, V., Corliss, J., Bellinger, E., Sigee, D., 2000. Indirect regulation rule for consecutive stages of ecological succession. Ecol. Model. 133, 73–82.

Linton, J.D., Yeomans, J.S., Yoogalingam, R., 2002. Supply planning for industrial ecology and remanufacturing under uncertainty: a numerical study of leaded-waste recovery from television disposal. J. Operational Res. Soc. 53, 1185–1196.

Meadows, D.H., et al., 1972. The Limits to Growth.

Meadows, D.H., et al., 1992. Beyond the Limits.

M.E.L. Research Ltd., 1999. Project Integra—Kerbside and Household Waste Recycling Centre: Waste Analysis and Questionnaire Survey Results. MEL Research Ltd., Birmingham.

McDougall, F.R., White, P.R., Franke, M., Hindle, P., 2001. Integrated Solid Waste Management—A Life Cycle Inventory. Blackwell Science, Oxford.

Odum, H.T., Odum, E.C., 1976. Energy Basis for Man and Nature. McGraw-Hill.

Papworth, R., Poll, A.J., 1991. Analysis of Four Samples of Domestic Refuse Arising from the County of Hampshire. Warren Spring Laboratory, Stevenage, p. 20.

Patten, B.C., 1992. Energy, emergy and environs. Ecol. Model. 62, 29–69.

Pearce, D., et al., 1989. Blueprint for a Green Economy.

Powersim Corporation, 1996. Powersim 2.5. Reference Manual. Powersim Press, Isdalstø, Norway.

SAEFL, 1998a. Assessment of Ecoinventories. Swiss Agency for the Environment, Forests and Landscape (SAEFL), Bern.

SAEFL, 1998b. Entsorgung von Abfällen in Zementwerken. Richtlinie. Swiss Agency for the Environment, Forests and Landscape (SAEFL), Bern.

SAEFL, 1998c. Life Cycle Inventories for Packagings. Environmental Series No. 250, vols. I and II. Swiss Agency for the Environment, Forests and Landscape (SAEFL), Bern.

SAEFL, 1999. Kunststoffabfall im Haushalt Separat Sammeln? Umweltschutz (1) Swiss Agency for the Environment, Forests and Landscape (SAEFL), Bern. http://www.umwelt-schweiz.ch/buwal/de/medien/umwelt/1999_1/unterseite19/index.html.

Snakin, J.P.A., Korhonen, J., 2002. Industrial ecology in the North Karelia Region in Finland—scenarios for heating energy supply. Int. J. Sustainable Dev. World Ecol. 9, 9–21.

Solano, E., Ranjithan, S.R., Barlaz, M.A., Brill, E.D., 2002. Life-cycle-based solid waste management. I: Model development. J. Environ. Eng.-Asce. 128, 981–992.

UREK-N, 2001. Bessere Verwertung von Kunststoffabfällen. Parliamentary Initiative 01.3642 of the Parliamentary Commission ‘Umwelt, Raumplanung, Energie’ (UREK-N) of the Swiss Parliament, 22 October.

Wäger, P., Gilgen, P.W., 2000. Auswirkungen der thermischen und stofflichen Verwertung von Kunststoffabfällen auf die schweizerische Abfallwirtschaft. Bericht Nr. 249, EMPA.

Widmer, H., Steiner, P., Textor, S., 1998. Betriebsoptimierung und Investitionsplanung in der Abfallbehandlung. Müll. Abfall. 12, 753–761.