Material and Energy Recovery in Integrated Waste Management Systems. Economic Analysis

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ABSTRACT

A critical assumption of studies assessing comparatively waste management options concerns the constant average cost for selective collection regardless the Source Separation Level (SSL) reached, and the neglect of the mass constraint. The present study compares alternative waste management scenarios through the development of a desktop model that tries to remove the above assumption. Several alternative scenarios based on different combinations of energy and materials recovery are applied to two imaginary areas modelled in order to represent a typical Northern Italian setting. External costs and benefits implied by scenarios are also considered. Scenarios are compared on the base of the full cost for treating the total waste generated in the area. The model investigates the factors that influence the relative convenience of alternative scenarios.

JEL: Q42, Q53

1. INTRODUCTION

The European policy framework for management of municipal solid waste (MSW) is inspired by a precise ranking of solutions, with materials recovery to be preferred to energy recovery, and landfill to be considered as a last-resort option once all valorisation opportunities have been exploited (Kraemer and Onida, 1999). Directive 2008/98 requires demanding targets for material recovery of priority material flows (such as glass, paper, plastics, metals, wood).

However, the evidence supporting this policy is far from conclusive. While a considerable agreement exists about the necessity to prioritize recovery of materials and energy over landfilling of MSW, at least in congested areas such as Europe, the question about the optimal ranking of alternative valorisation opportunities – with particular reference to energy vs. material recovery - is still debated.

The present study aims at improving the state of the art of the discussion through an economic assessment of alternative scenarios that entail different combinations of energy and materials recovery from MSW. The chosen approach is that of economic life-cycle analysis, often referred to as life-cycle costing (LCC) (Reich, 2005).

The study is conducted through a desktop simulation of scenarios applied to hypothetical regional settings which have been characterized according to patterns of MSW generation and urban structure that are representative of a typical area of central-northern Italy.

With the aid of a support tool, we have simulated alternative MSW management scenarios, whose main feature is represented by the mass balance constraint.

All scenarios are referred to the total amount of MSW generated; the management system is therefore considered as an integrated set of phases and techniques, in which the residuals of upstream phases are the inputs of downstream ones, until materials are finally marketed (with a positive economic value) or landfilled. An innovative aspect of the study concerns the explicit consideration of diminishing returns in the capacity of the system to improve the separation level of materials waste flows: this assumption is reflected by increasing marginal costs for separate collection and increasing share of non recyclable residuals when the source separation level (SSL) increases.

Assumptions on technical performance and economic cost have been made through a detailed meta-analysis of the literature and verified through a set of interviews with operators from different locations. For this reason, while the assumptions concerning urban structure and MSW composition may be specific to the context analyzed, the cost assessment model is largely independent from that so as to allow some generalization of results.

The study adopts a welfare economic (i.e. social) perspective. For this reason, we consider among the costs either the financial cost of each phase or the externalities generated; among the benefits, revenues from the sale of recovered materials and energy, evaluated at the market price, and positive externalities deriving from the emissions saved in the production of energy and materials that are displaced by waste-derived ones.

The aim of our study is not to individuate "the best" option, but rather to enlighten what are the assumptions that most influence the feasibility of each scenario and determine its strength and weakness. For this purpose, we have conducted a thorough sensitivity analysis on the crucial variables.

The study shows that once the materials balance constraint and the diminishing returns of materials recovery are duly accounted for, the preferred management solution in social cost-benefit terms entails a mix of energy and materials recovery, and therefore maximising the latter is not optimal. Energy and materials recovery should be considered as complementary options rather than alternatives.

The present article first summarizes the basic assumptions and the structure of the model (par. 3); then provides an outline of the results (par. 4), and finally reports the policy implications (par. 5). The full study is available in Massarutto et al.(2010).

2. STATE OF THE ART OF THE LITERATURE

The literature concerning the decision support to the choice of waste management solutions is overwhelmingly vast and has provided a consistent set of arguments in favour of a policy targeted at reducing waste flows addressed to landfill and recovering materials and energy (Pires et al., 2011).

The economic dimension has obviously been a substantial element of the evaluation; while initially constrained to the analysis of financial costs, an important innovation of the last two decades concerns the incorporation of environmental costs in the evaluation, either through environmental cost-benefit assessment or multicriteria approaches (Pearce and Turner; Powell, 1996; European Commission, 2000).

We can find in the economic literature dedicated to waste management two alternative approaches.

The first approach includes studies that provide detailed and comprehensive analysis of single technologies, such as incineration, mechanical sorting etc. (Economopoulos, 2010; Tsilemou and Panagiotakopoulos, 2006). Often in this category of studies we find comparative assessment of technologies (eg incineration vs. composting; anaerobic digestion vs. landfilling) that discuss them as alternative options.

This kind of comparison is useful for evaluating the range of application of each technical solution, but unsatisfactory for assessing the management system as a whole, since no single technology can address the 100% of waste flows, and all of them generate residues that should be treated in some other points of the system.

The second category is that of integrated management scenarios. In this case, models do explicitly take care of the materials balance, considering the bulk of waste flows generated and following them through the management system, with the aim of optimizing the choice of decisionmakers (Pires et al., 2011; Dornburg et al.; Zotos et al., 2009; Tanskanen, 2000).

These models are most often forced to over-simplify the analysis of costs of different techniques. A common hypothesis concerns average costs remaining constant: these are normally expressed in a certain value per quantity collected, eventually distinguishing between drop-off and kerbside schemes, regardless the source separation level (SSL) actually achieved (Eunomia and Ecotec, 2001; Craighill and Powell, 1996; Daskalopoulos et al.; 1999).

However, collection costs are ultimately fixed, depending on the method of collection, the frequency of emptying etc (Anex et al., 1996). One cannot imagine that the cost per ton of collecting materials is the same if the target is 10% or 90% of the total amount of each material that is contained in the MSW. Moreover, one should also account for the quality of materials collected: a higher SSL is normally associated with a poorer quality, resulting in higher sorting costs and a greater share of residuals from sorting, that will have later to be disposed of.

The importance of diminishing returns in recycling has been shown for example by Kinnaman (2006) in his detailed survey of US recycling programmes: while the net benefit (in terms of saved social cost of landfilling) do not necessarily compare positively with the extra cost implied by recycling schemes.

Both approaches in recent times have found a significant improvement through the methodology of LCA (Cleary, 2009; WRAP, 2006; Shmelev and Powell, 2006). This aims at understanding the economic and environmental implications of a certain good “from cradle to grave”. Its application to waste management, however, is not obvious (Reich, 2005; Finnveden, 1999). As a “good having negative value” (Porter, 1996), waste needs to be looked at with an approach to LCA that is opposite to that of normal goods: the “cradle” should hence be the moment when a certain good acquires negative value (i.e. when one decides to dispose of it into the municipal system), while its “grave” could either be the moment when some materials return back to the productive system with a positive value, or are permanently landfilled. The market value of materials (positive or negative) could hence be considered as a practical indicator of when something begins and ends to be a waste.

While this general rule is conceptually clear, applying it in concrete is not straightforward; in order to decide whether to incorporate a certain cost in the analysis or not, we previously need to clarify the perspective of the analysis: whether private or social, and in the latter case whether referred to the optimization of a specific local management system or having an holistic vision.

For example, materials collected for recycling have often a positive market value only because the set of incentives provided by extended producer responsibility enable industry-funded compliance schemes to pay a price to collectors; yet these systems have to bear further costs after collection, even if these will not be paid in the waste bill, but rather through the market price of goods incorporating charges levied by compliance schemes. In other cases, benefits may be overrated because waste-derived outputs (such as energy) receive an extra above the market price (for example, incentives to non-fossil energy sources paid by many EU governments, revenues from so-called "green certificates"). Other prices may be distorted by taxation (e.g. landfill) or market power of operators.

If the purpose of the analysis is to optimize the choices of a waste management authority, relying on market values is appropriate (see for example Emery et al., 2007; Bovea and Powell, 2006). This is not the case if the analysis aims at providing a more general evaluation of the appropriateness of a certain management scenario for the society as a whole (since in the latter case these costs will be ultimately be borne by the system and paid by consumers, e.g. through higher prices of packaged products). In order to ensure that recycled materials have reached their "grave" in the production system, therefore, costs and benefits should be considered net of direct and indirect incentives.

Many LCA studies have been designed for assessing alternatives referring to *specific material flows* such as glass, paper, plastics, packaging, electronic waste (WRAP, 2006). While this is useful for understanding the potential of establishing separate management systems and identifying priority flows, it is not always clear what are the implications for the remaining waste. Again, if we introduce the constraint that the evaluation concerns *all waste generated* (directly from households or indirectly as residuals of other management phases) it is not obvious whether the solution that is optimal for a certain waste flow is also optimal for the system as a whole.

As Morrissey and Browne (2004) put it, "despite the important achievements, models used in the literature … have limitations and none have considered the complete waste management cycle, from the prevention of waste through to final disposal. Most are only concerned with refining the actual multicriteria technique itself or of comparing the environmental aspects of waste management options (recycling, incineration, and disposal). In addition, while many models recognize that for a waste management model or strategy to be sustainable, it must consider environmental, economic and social aspects, no model examined considered all three aspects together in the application of the model and none considered the intergenerational effects of the strategies proposed".

3. MODEL AND BASIC ASSUMPTIONS

2.1 The model background

The present study aims at making some steps forward in the direction advocated by Morrissey and Browne, trying to integrate the above approaches by providing an economic evaluation of integrated waste management scenarios that at the same time (i) considers the materials balance constraint and (ii) explicitly consider the cost variability associated to different yields in terms of recovery of specific waste flows.

In this perspective we follow a line that is similar to the one adopted in the GIGO model by Anex et al., 1996; however, our perspective differs from theirs for the fact of adopting a social cost-benefit approach. We include externalities in the evaluation, and consider all costs and benefits net of subsidies and environmental taxation. The focus is on the total amount of MSW generated, and not on specific technologies or materials; according to the LCA methodology, we consider MSW from the "cradle" represented by citizens addressing materials to the municipal service to the "grave" represented by their return to the productive systems as goods having a positive market value; however, as pointed out in par. 2, we also consider treatment phases that are required downstream of separate collection in the industrial system.

Instead of relying on data collected directly from case studies, we have adopted a standard cost approach. Costs of all management phases have been estimated according to a complex methodology that involves desktop simulations, meta-regression on literature and validation through confrontation with real operators. While the analyzed setting is a typical Northern Italian one, the potential for generalization of results is therefore higher.

Following Reich, we develop a life-cycle costing (LCC) model that treats separately the financial and the external cost dimension. Although these are finally summed together in order to get the final result, partial results of each are also shown in order to make comparison easier.

The management system is modelled on two hypothetical regions. Assumptions concerning the size of management areas, social and urban structure, composition of MSW are made considering the typical setting of a Northern Italian region and have been validated through interviews with operators.

From this starting point, we model a set of alternatives (scenarios) characterized by a different combination of technological phases that imply different (growing) targets for SSL and materials recycling; a materials balance constraint has been introduced both for the system as a whole and for single phases, i.e. an equivalence between the weight of materials entering and exiting, net of losses due to evaporation and emissions to air. The model considers all phases until material flows contained in the MSW either (i) are transformed into materials that can be marketed again, i.e. acquire a positive value; or (ii) are landfilled.

The analysis has been conducted on the same scenarios already outlined in the other papers in this issue (see the paper of Consonni et al. for a general outline). Two hypothetical regions are simulated: a large metropolitan area producing 750 kt of Gross Waste (GW) per year and a district with a small city and a rural surrounding, producing 150 ktGW/year. Composition of waste, urban structure and density is also the same; on this base collection systems are optimized and assumptions are made about the share of population that can use home composting facilities.

Both areas are imaginary, but based on features that are typical of a Northern Italian setting. We have concentrated the analysis on the ordinary waste flow; we have assumed therefore that special waste contained in the GW flow are collected separately through dedicated systems (eg electronic waste, bulky waste, waste from street cleaning).

The scenarios consider different levels of commitment in the separate collection, with direct incineration of residual waste. On top of the 4 scenarios analysed in the other papers (see again Consonni et al. for details)., 2 further scenarios are presented, in which we assume to stress separate collection in order to achieve the highest SSL; while for managing residual waste we consider, besides mass incineration, alternative options represented by different combinations of composting, RDF and indirect materials recycling in the construction industry.

2.2 Scenarios

Six scenarios have been analyzed and discussed, with different assumptions on the targeted SSL and the technologies used for treating the materials sorted by Separate Collection (SC) and the Unsorted Residual Waste (URW). Each scenario consists in a combination of phases, starting from the generation of waste and ending with its return to the productive system or to the environment. We assume that neither import nor export of waste takes place.

The common assumption behind all scenarios is the objective to minimize landfill requirement. Once landfill is minimized, in turn, we do not make any further constraint about the hierarchy to be respected, assuming instead that all forms of recovery (direct and indirect, materials recycling and energy) are potentially equal; the relative ranking of alternatives is therefore considered as a result of the model, and derives from the objective of minimizing the total social cost.

Scenarios are labelled according to the prevailing collection technique (D = dropoff; K = kerbside) while the figure in each label corresponds to the targeted SSL (e.g. in D35 the target is 35% of total waste collected separately). Since the target differs, collection strategy also differs: for example, in order to obtain a SSL of 50% with drop-off systems (D50) we need a different organization with respect to D35, which leads to different average and total costs.

The first four scenarios correspond to the ones defined in the previous papers in this series (Consonni et al., 2011). D35 and D50 are based on drop-off collection systems with separate collection of all priority flows except organic waste. D35 corresponds to a standard system with ordinary diversion targets, while D50 can be considered as an upper limit that can be achieved by drop-off systems only in specific cases, where all external circumstances cooperate at their best.

Scenarios K50 and K65 are based on kerbside collection extended to organic waste. A crucial assumption concerns the fact that, in order to achieve higher SSL rates, the quality of materials collected worsens, and therefore a higher amount of residues from sorting and treatment is produced. Despite this assumption can be criticized on the basis that quality can be improved through education, commitment and control, it has been actually verified on field and seem to capture the experience of operators who practice kerbside collection in the Northern Italian context.

The last two scenarios, K75 and K85, assume kerbside collection with higher separation rates, better quality of materials (lower downstream residues) and domestic composting of organic waste. Both scenarios reproduce the best practices that have been reported in the Italian experience so far. It is important to note that all these experiences regard medium-small towns (10.000 – 50.000 inhabitants maximum) and have never been tested so far at a higher and more complex scale.

In all scenarios but K85, the unsorted residual waste (URW), arising from undifferentiated collection and from the residues downstream of selection and recycling, is destined to mass-burn incineration with recovery of heat and electricity (the option of electricity alone is also considered for the larger region only). In K85, instead, we assume that URW is recycled in facilities producing inert materials for the construction industry and for road foundations, in combination with other industrial waste flows.

The assumptions concerning mass-burn incinerators and energy recovery efficiency are the same as in Part C of this paper series; for the recycling of inerts we have made specific assumptions derived from the experience of an already operating facility in the North East of Italy (Vedelago, Treviso), accessed through direct interviews, which is regarded as the leading experience adopting the “zero-waste” philosophy in Italy (see also Viale, 2008). The most critical assumption for the materials balance concerns in this case the need to mix flows originated from MSW with special waste in given proportions in order to achieve the composition required by Italian Standard Organisation (UNI) norms for reuse in construction industry.

Finally, for all scenarios, we have considered three alternative options to mass-burning, corresponding to as much sub-scenarios. The alternatives considered are (i) separation and bio-stabilisation of organics, mass-burning of combustibles; (ii) the same as (i), with production of low-quality RDF, burned in dedicated facilities and landfilling of residues; (iii) the same as (ii), with production of high quality RDF destined to cement kilns and landfilling of residues.

2.3 The basic technologies: financial cost

Management phases (collection, treatment and disposal) have been modeled according to an engineering cost approach (Pratten, 1971; Kaulard and Massarutto, 1997). This methodology implies a careful desktop simulation of each phase, based on assumptions concerning technologies, while cost inputs are derived from a mixture of market prices when available, meta-analysis of technical literature, expert assessments derived from interviews and sometimes verified on-field through interviews with operators. All the detailed references and basic data are provided in Massarutto et al.(2010).

Financial costs are evaluated net of taxation and any kind of fiscal incentives. They all include both operational and capital cost. For all phases, capital costs have been estimated on an annual base making assumptions concerning their economic life, with an interest rate of 5%.

Annual investment costs and operational costs are transformed into unit costs (per ton treated in each specific phase) by making assumptions concerning the actual yearly functioning time. Excess capacity has also been accounted for (this hypothesis will be later object of sensitivity analysis).

For each scenario, collection systems have been modelled by solving an equation that minimizes the total cost, subject to constraints represented by the number, size and density of collection points, the time needed for emptying each bin, the minimum frequency that allows to prevent overflows. Assumptions have been made with respect to the equipment used, based on the experience of operators collected with a detailed survey on site. Cost ranges are presented in table 1; unit costs vary from one scenario to the other, depending on the specific assumptions that characterize each of them.

<Table 1>

For incinerators, we have used the reference data provided in the detailed meta-study conducted by ENEA (Iaboni and De Stefanis, 2007), integrated with other literature (Ragazzi and Del Duro, 2006; Iefe and Eco&Eco, 1999; Regione Emilia Romagna, 2005). Assumptions have been verified on the field through interviews with operators, with particular reference to the facility operating in Brescia, that is commonly regarded as a reference benchmark in Italy. We have considered facilities equipped with both electric energy and heat recovery; for the large management area only, we have considered separately the option of producing electricity alone. We assume that heat recovery is feasible without extra cost (i.e. we do not account for the cost of installing a district heating network, which is assumed to be already in place).

Sorting facilities have been modeled with a desktop work supported by interviews with operators; the same is done for other simple facilities, such as composting plants.

For the production of RDF, we have again started from a desktop study based on literature data; these have later on been carefully verified with data obtained from existing facilities, in particular the Idea Granda plant in Cuneo (reference for an up-to-date technology) and other sources (Ragazzi and Rada, 2008; Iefe-Eco&Eco, 1999).

For landfill, we have considered a financial cost estimated through direct interviews, net of taxation and royalties. We have later estimated a scarcity cost, as suggested in Massarutto (2007), by considering the difference between the financial cost and the average market price. The scarcity rent is assumed to be actually shared between local authorities (royalties and compensative payments), regional authorities (landfill tax) and site owners.

<Table 2>

2.4 Financial benefits: market value of materials and energy

All benefits deriving from the recovery of materials and energy from waste have been evaluated at the market price (when a market price exists), net of all subsidies. This choice is motivated by the fact that in a social cost-benefit analysis both subsidies and taxation should be considered as clearing entries. In particular, we have omitted the subsidies paid in support of renewable energy (subsidized price, green certificates) as well as the contributions paid by producers for sustaining separate collection and recycling.

For electricity, we have assumed a market price corresponding to the weighted average of prices registered in 2008 at the Italian electricity stock exchange (GME), equal to 75 €/MWh.

The market value of most commodities (glass, paper, metals, plastics) has been estimated on the base on the average market price registered in official transactions, as recorded by Chambers of Commerce. For quality compost (obtained from separate collection) we rely on a detailed market study conducted by Ricci et al. (2003).

For materials whose market price was not available or unreliable, we have adopted a net-back approach. According to this method, the economic value of a certain material can be approximated by the market price of the closest substitute (e.g. fertilizers, coal) less the cost needed for adapting industrial processes to the use of the waste-derived material. This approach has been used in particular for heat from combustion; compost from mechanical sorting; RDF; inerts for building industry. In the case of bio-stabilised material (from mechanical sorting) and inerts, we have assumed that the cost of production equals the cost of upgrading the material until it acquires a positive value; therefore, the price considered here is 0.

All hypotheses made are summarized in table 3. A sensitivity analysis will be provided for recycled commodities in par. 3.

<Table 3>

2.5 External costs and benefits

External costs and benefits have been evaluated with the approach of benefits transfer, namely by adopting the unit value derived from a detailed meta-analysis of the literature to the material and energy recovery that characterize each scenario.

As far as external costs are concerned, we have considered emissions in air (from incineration, landfill and collection vehicles), climate change (CO_2) and disamenities. For air emissions, the main sources are the thorough survey conducted by the European Commission (2000) and the more recent work of Rabl et al. (2007). This latter study has applied the well known ExternE model to a set of facilities located in urban areas in Europe. The ExternE approach is based on an impact-pathway methodology. Emissions per ton are first converted into ambient concentration of pollutants. With the aid of dose-response functions derived from a thorough meta-analysis of epidemiologic literature, marginal increases of ambient concentration due to the treatment of a ton of waste are converted into the consequent arising of pathologies. Finally, the economic cost of each pathology is estimated, again through a combination of methodologies that includes contingent approaches, health care costs etc.

For CO_2 emissions, we have adopted the average price of emission trading certificates during 2008-2009, equal to 19 €/t (Source: Italian power stock exchange, GME). Once again, since this assumption is delicate and can influence the result, we have considered alternative hypotheses in the sensitivity analysis.

<Table 4 >

Benefits have been evaluated considering the same emissions. We have assumed that energy recovery from waste displaces oil and coal - powered thermoelectric plants and oil and gas-fuelled domestic heating systems. However, the study of Rabl et al. (2007) assumes energy recovery rates that are far lower than our estimates; this occurs because they consider already existing facilities, while in our study we have referred to the best available technologies optimized for energy recovery (see table 4, where we have considered separately assumptions made by Rabl et al., (2007) and ours).

In order to transfer the results of Rabl et al. (2007) to our study, we have extrapolated from that study a value per unit of energy recovered, and then multiplied it for the actual recovery level that is assumed in our scenarios. Since the difference is relevant, this hypothesis has later been tested through sensitivity analysis (par. 3).

For landfill, we have assumed that no energy recovery is made, since in all scenarios landfills are used only for non biodegradable residues (ashes, inerts). External costs due to disamenities and leachate have been derived from European Commission, (2000), Rabl et al (2007) and DEFRA (2003).

For materials recovery, we have adopted the estimates provided by Bianchi (2008) in a detailed meta-analysis of the expected savings in terms of primary energy and CO_2 implied by the recycling of a ton of selected materials (glass, paper, plastics, metals and wood). These quantitative estimates have been converted into monetary values by adopting the same hypotheses as for electricity and heat.

In our study we do not consider further benefits that households may derive from separate collection as “warm-glow” utility (Kinnaman, 2006), namely the private non-monetary satisfaction arising from ethical fulfilment. Studies conducted in the US quantify this extra benefit in 4-7 $ per tonne of waste recycled. Since to our knowledge no similar studies have been conducted in Europe, we have preferred to omit this dimension.

4. RESULTS OF THE STUDY

4.1 Comparative analysis of scenarios

Following the approach outlined in the previous section, we have modeled the costs of each technology expressed in €/t. Each scenario implies a different flow of materials entering each phase; therefore, the total cost of each phase (e.g. collection of paper, incineration or sorting) for each scenario is calculated by multiplying the unit cost per the quantity of waste actually treated in that phase. The sum of all phases provides the total cost. Dividing these totals by the total amount of waste generated (150 and 750 kt respectively for the two management areas) we obtain the cost per ton of gross waste. We refer to this latter value as €/tGW in order to distinguish from the previous section in which all costs were expressed with reference to the quantity actually treated by each facility (€/t).

Table 5 shows the detail of the collection costs. As we can see, total collection costs per ton of waste are higher for the scenarios based on kerbside collection; yet this cost is rapidly diminishing with the increase of SSL. This can be understood, since collection systems fundamentally imply a fixed cost that can be better absorbed if arising are higher. We can also see that the break even between kerbside and drop-off collection systems is reached only for SSL around 85%.

Table 6 shows the results obtained for the various treatment phases. Here again, the total treatment cost of each scenario results from a different combination, with a higher weight of WtE in the first scenarios and a relatively higher weight of sorting, composting and inerts production for the latter ones.

Table 7 summarizes externalities, separating costs and benefits. It is important to point out that in all scenarios external benefits are higher than costs: this is due both to energy recovery and to materials recycling, and is highly dependent on the assumptions made for energy recovery.

Differences between scenarios are far smaller than one could expect. This is due in particular to the fact that the recovery of both electricity and heat allows to avoid emissions from power stations and direct heating.

Emissions from collection and transport are negligible, as well as disamenities.

Revenues from the sale of recovered resources are significant (table 8); here again the weight of electricity and above all heat is highest for D, while materials weigh more in the K scenarios. Revenues from plastics are particularly high (this value is counterbalanced by correspondingly higher collection and sorting costs).

Table 9 provides the final synthesis of results. We have adopted a scalar structure in order to enlighten how the final result is obtained. Financial costs are lowest for D35 and highest for K50. In general, D scenarios seem less costly than K ones. It is also interesting to note that while values for D scenarios are quite close, those for K depend highly on the SSL, with K85 costing up to 45% less than K50. This suggests that drop-off collection is less vulnerable to the actual recovery rates, while kerbside collection, implying high fixed costs for the collection system, once set up, needs that high SSL are actually achieved.

Considering revenues reinforces this statement, since revenues from energy allow D scenarios to obtain higher revenues. The recovery of heat seems to have a high impact; when it is feasible, our model suggests that it is far more convenient to push this solution as far as possible.

Once externalities are considered, D scenarios maintain and even increase their superiority; however, D50 results cheapest than D35: the higher external benefits outweigh the higher financial costs. Here again it appears clear that K scenarios are vulnerable to the SSL. In other words, establishing a kerbside system makes sense only if it manages to reach the highest diversion rates, otherwise it is dominated by D in all aspects (far lower cost, higher revenues, comparable external costs, higher external benefits).

Table 5

Table 6

One final remark concerns the destination of URW. In all scenarios, the model shows that direct incineration allows a significantly better performance than further treatment phases aimed at recovering bio-stabilisation and RDF. Since the calorific value of URW is high enough, the additional cost of separating organic materials is not compensated by additional revenues and benefits from higher energy recovery. Producing RDF destined to dedicated facilities again worsens the balance, since the extra cost is not corresponded by appreciable benefits. Even higher additional costs are implied by the production of high-quality RDF (destined to cement kilns and coal-fuelled industrial plants as a substitute for coke). This last solution has nonetheless the advantage of allowing the use of already existing facilities, and can therefore be recommended in case these facilities are available and the social consensus for building WtE plants is more problematic to achieve.

Table 7

Table 8

Table 9

3.2 Sensitivity analysis

Although carefully designed with a constant reference to the literature and the empirical data, scenarios analyzed above are obviously based on discretional assumptions, and can therefore be criticized (Eriksson and Baky, 2010).

With the aim of testing the robustness of the results against the most critical assumptions, we provide a detailed sensitivity analysis. Table 10 summarizes the tests that have been made, which concern assumptions on energy and materials recovery, market value of recovered resources, effective results in terms of SSL and actual recycling, improved efficiency of technological phases.

The tests have been made with the aim of verifying how far a certain assumption is critical, regardless the actual plausibility of the new assumption. In most cases, we have evaluated the outcome that is generated if the concerned variable assumes a value ranging from half to twice the value assumed in the baseline model. In the case of energy recovery, instead, we have considered an alternative scenario in which the recovery rate is the one of already existing facilities, as assumed by Rabl et al. (2007). Finally, we have done a more careful exploration of the assumptions on which the K75 scenario is based, where we have searched for a break-even combination of SSL and home composting (i.e. the level of SSL and home composting that would make this scenario eventually equal the cost of the best alternative).

In table 10, we examine the effects in terms of ranking of scenarios and the impact on the gap between them.

As we can see, the results of the baseline model are quite robust face to most assumptions. The preference to D scenarios is confirmed, although sometimes the order of preference between D35 and D50 is inverted. The gap is reduced, particularly in the case of higher prices for recycled commodities and as a function of the quality of materials collected separately. Alternative assumptions on external costs, and particularly CO_2, in turn, further emphasize the gap, since the same multiplying factor also concerns spared emissions.

The only truly critical assumption concerns the efficiency of energy recovery and, of course, the possibility to actually recover heat on top of electricity.

As we have noted above, assumptions on energy recovery in our model are quite generous (though realistic), since they are based (i) on the use of best available techniques and (ii) on the high calorific value guaranteed by the fact that what is actually incinerated is the URW and not the MSW. If we adopt more careful assumptions, results can vary significantly.

Once different assumptions are made (assuming that energy recovery rates are the lower ones derived from Rabl et al.), results change significantly. As far as social costs (including external costs and benefits) are concerned, the outcome is quite dramatic, since the "zero waste" scenarios, and particularly K85, become the preferred option. In terms of financial costs only, the preference is still for D35 and D50, but the gap with K75 and K85 is almost completely filled. What is confirmed in all cases is the low rank of K50 and K65, as a further confirmation that scenarios based on intense recycling can function only if the highest targets are achieved either in quantitative or in qualitative terms.

Finally, an important outcome concerns the efficiency improvements that can be obtained through an optimal use of the capacity, as well as by achieving economies of scale in the treatment plants, particularly WtE plants in the small area. In the sensitivity analysis, this efficiency improvement is represented by reducing to zero the excess capacity of each facility.

In order to make this assumption realistic, one should imagine that each facility is able to treat waste arising from other territories, i.e. that some trading of URW is allowed. If the self sufficiency principle is compulsory (as in the assumption made in the baseline, with no import nor export of waste from each area), it is more realistic to assume that facilities would have to be oversized in order to face peak demand and emergencies. Some possibility to trade waste among areas and eventually to share some facilities could instead allow a more efficient use of capacity, and at the same time promote the development of separate collection, since both the exporting and the importing area would have the incentive to improve recycling (Massarutto, 2007).

Table 10

5. CONCLUSIONS AND POLICY RECOMMENDATIONS

This study has provided a comparative assessment of alternative management solutions for MSW by adopting an innovative approach that imposes a materials balance constraint, and combines an assessment of integrated scenarios with more realistic assumptions on technology costs, that include diminishing returns of recycling and increasing unit cost for higher SSL.

The study emphasizes the need to consider waste management technologies as complementary parts of an integrated strategy, rather than alternatives. In this light, it emerges quite clearly that both materials recycling and energy (and heat) recovery through incineration of residual waste are needed in order to effectively minimize the waste flow addressed to landfill.

The critical threshold of optimal recycling seems to be a SSL of 50%: trying to push SSL beyond this level implies higher financial costs (in the range of 30-60%), and these are not compensated by positive externalities. Moreover, considering externalities reinforces this statement, at least when energy recovery is optimized and once the benefit arising from reduced emissions of the displaced energy sources are accounted for. This outcome contradicts the one of many LCA studies surveyed in our review, where recycling most often results as a preferable option.

The model also shows that intensive recycling – that requires a widespread and punctual kerbside collection – makes sense in economic terms only if the actual recycling rates are very high: which implies not only the highest SSL (above 75%), but also that the quality of materials collected is high as well. This target can sometimes be achieved, as some experiences in Northern Italy already show, at least in small centres; yet it is questionable whether they can be extended at higher territorial scales. In any case, they require a high degree of public participation, active involvement and education, since households are required not only to separate many different waste flows, but also to prevent waste generation by making smart shopping choices, sorting and washing of materials at home, domestic composting etc.

This somewhat extreme solution, therefore, seems practicable in specific circumstances, but cannot be generalized. In all other cases, it seems that a more balanced strategy, based on drop-off separate collection expanded up to its highest possible separation records and direct incineration of residuals is the most reasonable one. This result is further enhanced if (i) energy recovery efficiency is actually high; (ii) heat recovery is also feasible, which implies industrial facilities in the surroundings or a district heating network already operating.

The model shows that not only final costs are lower for D than even for the most fortunate K scenarios; but also that D ones imply lower vulnerability and higher robustness, since results remain quite constant under different assumptions on the actual SSL; while in the K scenarios, costs are highly sensitive to the actual recovery rate, and increase dramatically if (i) SSL are lower than expected or (ii) quality of recovered materials is poor, with the related higher percentage of residues after selection. This seems a more likely outcome in large urban areas, especially where neighbourhoods are more densely populated.

For managing residual waste (URW), direct incineration remains the most effective and economic option. Alternatives based on mechanical sorting, bio-stabilisation and RDF do not provide a cost-effective alternative, since WtE plants would have to be constructed anyway. Even if cost effectiveness is even lower, the production of high quality RDF (destined as an alternative to coal in industrial processes) could be ranked positively in case the location of a WtE plant is problematic.

Alternatives to energy recovery, such as the destination of treated dry materials to the production of materials for the building industry, find a limit essentially in the materials balance. Their feasibility is conditioned by the need to mix MSW-derived flows with industrial waste in much higher proportion, which does not seem to be the case if the scale grows significantly.

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