Economic aspects of thermal treatment of solid waste in a sustainable WM system

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Abstract: This paper offers a systematic review of the literature of the last 15 years, which applies economic analysis and theories to the issue of combustion of solid waste. Waste incineration has attracted the interest of economists in the first place concerning the comparative assessment of waste management options, with particular reference to external costs and benefits. A second important field of applied economic research concerns the market failures associated with the provision of thermal treatment of waste, that justify some deviation from the standard competitive market model. Our analysis discusses the most robust achievements and the more controversial areas. All in all, the economic perspective seems to confirm the desirability of assigning a prominent role to thermal treatments in an integrated waste management strategy. Probably the most interesting original contribution it has to offer concerns the refusal of categorical assumptions and too rigid priority ladders, emphasizing instead the need to consider site-specific circumstances that may favour one or another solution.

Keywords: Incineration; environmental economic assessment; cost-benefit analysis; waste-to-energy

JEL: Q53; L43; L99

1. Introduction

As a trained economist and a professional applied economist, I often find myself at odds when trying to explain to non-economists what precisely economics is about. Most people, including scholars of non-economic disciplines, have some idea that economics concerns the valuation of costs and benefits, and thence expect an economist to have some contribution to offer to policymakers when they face mutually exclusive choices. Although Murphy laws trivialize economists as scientists that "know the cost of everything and the value of nothing", even non-economists are now familiar with the concept of social value, including financial as well as external costs and benefits – which means that economic valuation can promise to encompass a much wider set of issues than purely commercial ones.

Yet valuation by no means exhausts the range of application of the economic analysis - waste management policies included. A look at the table of contents of a recently published book – the "Handbook of Waste Management" edited by two leading economists in the field, Tom Kinnaman and Kenji Takeushi – offers an insight on the most popular themes: these include behavioral studies on the approach of individuals to separate collection, studies focusing on the market structure of recovered materials, issues related to transnational waste shipments, regulatory issues relative to the industrial organization of waste management services, application of monetary incentives and other economic instruments to promote waste policy targets, and so on.

It is quite surprising to me that such an influential volume, collecting the frontier of academic research about waste management economics, has no single chapter - and hardly any discussion – about waste combustion and waste-to-energy (WtE), as it does not about landfilling. It seems that the "old economy" of waste – that concerned by the traditional practices of collection, handling and disposal – is not an attractive field of research compared to the "new economy", whose keywords are material recovery, closing-loops and community involvement.

This intuition is confirmed by the literature review that we have performed on the most popular and widely known academic search engines - ScienceDirect, Scopus and Ideas – using incineration as a keyword. While the theme is widely covered in the literature of environmental engineering and management sciences, only a few contributions are genuinely "economic" in the approach and methodology.

An influential "new wave" of thinking in the field of waste relegates incineration in the past, ranging from the zero-waste gurus to the supporters of "good old" landfill: the former because it is assumed to be much better for environment and climate change, the latter because it is by far cheaper.

Yet a look at the real world experience tells a different story. In Europe and in Japan at least, waste combustion in different forms – at either mass-burning facilities or co-combusting refuse-derived fuels (RDF) with solid fossil fuels in industrial plants - still concerns a fraction from 30 to 60% of the total waste flow. Even more interesting, the share is highest in the countries that have reached the highest rate of landfill diversion – a hint, if not a demonstration, that WtE is complimentary, rather than opposite, to recycling, in the effort of phasing-out landfills (CEWEP, 2014)

Critics of incineration interpret this as a proof of the power of an influential lobby of construction industry, technology manufacturers and large multinational companies, blamed for using incineration as a Trojan horse for the sake of monopolizing the waste management industry. Others argue that incineration involves lock-in processes that "trap" those countries that have engaged in it, preventing them from making more sustainable choices. Without dismissing too early such arguments, our paper aims at providing a comprehensive analysis of the mainstream economic thinking about waste combustion, through a detailed literature review.

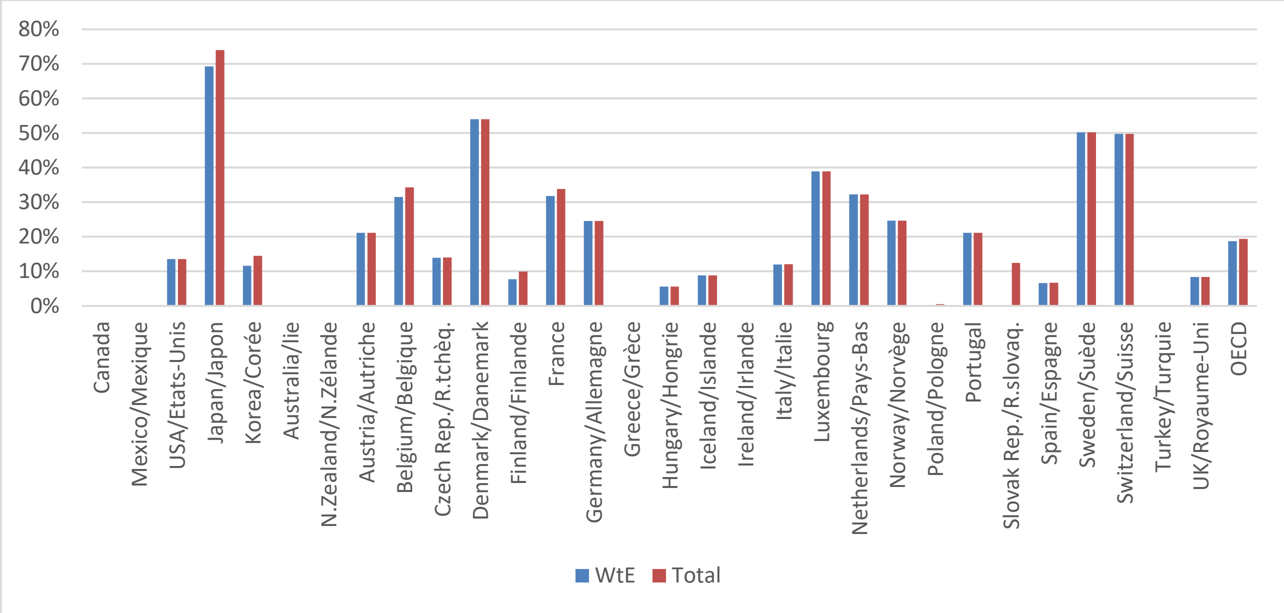
We shall present at the start a short overview of the main facts that characterize incineration today. We shall review the literature concerning the analysis of financial costs (par. 3) and external costs (par. 4), examining the degree of convergence and discussing the critical assumption that explain differences. Par. 5 illustrates more in detail the results of alternative assessment techniques, analyzing the role assigned to waste combustion and again the most critical assumptions. Par. 6 surveys analyses that compare different combustion solutions, with particular reference to the alternative between mass-burning facilities and the processing of waste aimed at extracting marketable combustibles. Finally (par. 7) we'll move to the issue of economic regulation of the waste combustion market, discussing the possible existence of market failures that justify state intervention. In the concluding chapter we shall underline the questions and issues that in our opinion are still open to economic research in the future.

2. Some stylized facts about waste incineration

On average, in the OECD, 19% of municipal waste are incinerated, nearly all of them with some kind of energy recovery. Four countries (Japan, Denmark, Sweden and Switzerland) incinerate more than 50%, while in other countries (Germany, France, Netherlands among others) the share is between 30 and 50% (fig. 1). The largest part of incinerators are also equipped for energy recovery.

It is interesting to see from fig. 2 that there is a clear correlation between incineration, recycling and landfilling. Countries that managed to nearly eliminate landfills (less than 2 kg/yr per capita) adopt a balanced combination of incineration and material recycling; while countries that do not incinerate must rely on landfill for more than 30% of their MSW.

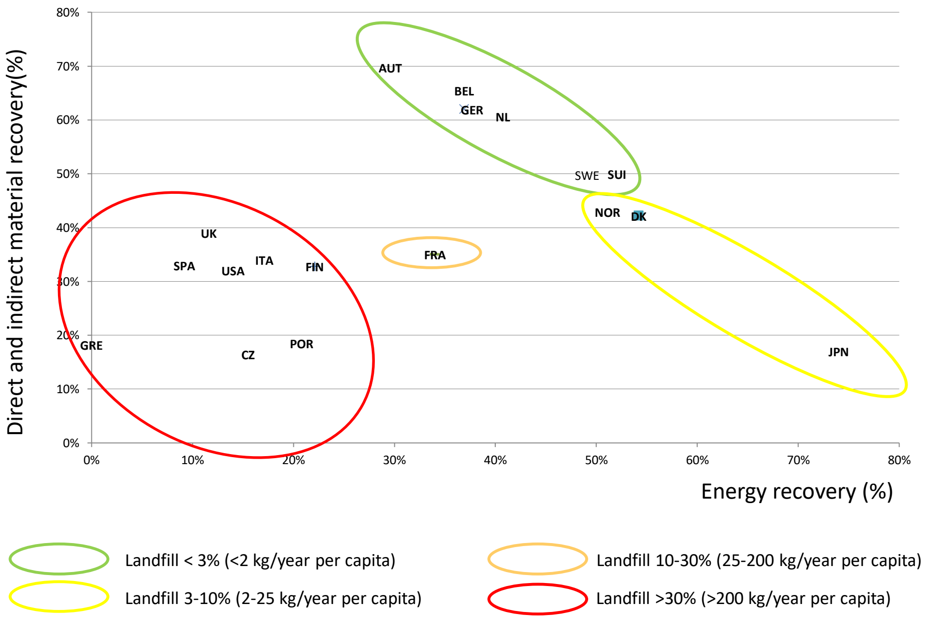
Figure 1 – Fraction of MSW incinerated in OECD countries



WtE: incineration with recovery of electricity and/or heat (waste-to-energy); Total: total fraction incinerated

Source: our elaboration on Oecd

Figure 2 – Material recovery, energy recovery and landfill phaseout



Source: our elaboration on Eurostat and OECD

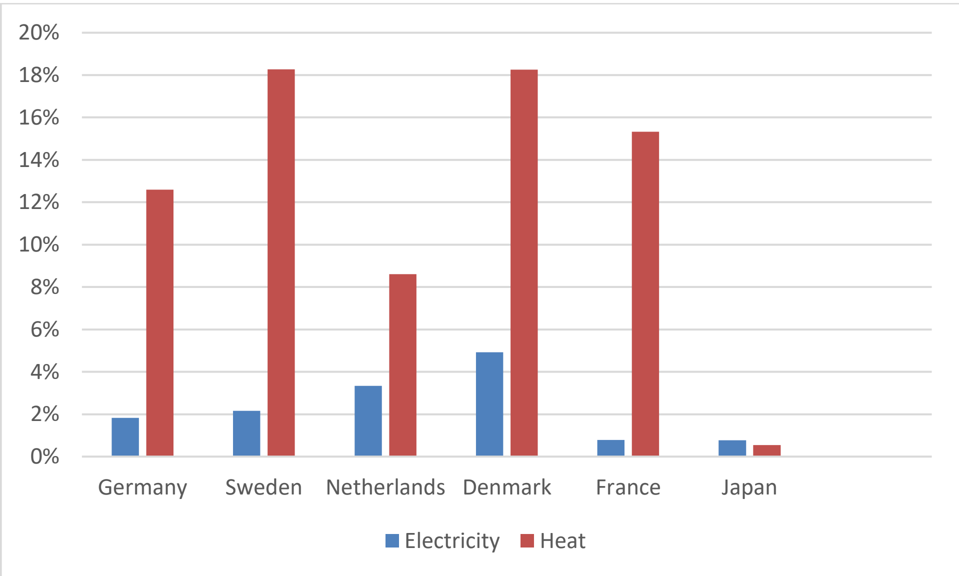
Seemingly, countries that achieved a higher share of incinerated waste for a long time show a mature situation, which does not seem open to further increase of volumes incinerated. Tendential decoupling of waste generation from economic growth, together with technical and organizational innovation in the field of recycling actually determines a further limitation to the growth of this market; in fact, countries with the higher installed capacity experienced an excess of supply during the last decade. Upgrading, adaptation to environmental regulation and revamping are the most typical drivers of new investment.

On a global scale, however, the incineration market still exhibits significant growth trends (Eckhard, 2013). Market analysts forecast the global Incinerator market to grow at a CAGR of 5,54% over the period 2013-2018.

This is particularly the case in some EU countries that are experiencing a transition dominated by the aim of phasing out landfills as much as possible (e.g. Italy and the UK); and even more, of emerging economies and the BRICS. In China, for example, where still something like 7 billion tons or waste are still untreated, the XII Five Year Plan 2011-2015 sets a target of 90% for harmless treatment, 35% of which will be provided by incineration facilities (China Research and Intelligence, 2014).

The contribution of waste combustion to the overall energy balance is quite limited. According to the IEA, 2013, only 0.4% of electricity at the global scale is obtained from waste. This share is more than three times higher (1.3%) in Europe. However, overall global shares are misleading, since they average out countries that rely on incineration in a very different way. Focusing on those that incinerate the highest quantity of waste, these figures increase significantly. Figure 3 shows that the share for electricity is 4.9% in Denmark, 3,3% in the Netherlands and 2.2% in Sweden; even higher rates are obtained for heat, with a share above 18% in Sweden and Denmark.

Figure 3 – Share of electricity and heat obtained from WtE in some of the global WtE leaders (year 2011)



Source: our elaboration on data from International Energy Agency

According to CEWEP estimates, the potential production of energy from waste by 2020 amounts to 196 TW (76 for electricity and 120 from heat), and thence nearly double of actual figures. Considering that energy from waste is 50% due to renewable materials contained in the waste flow (biomass etc), this also means that WtE offers a substantial contribution to the production of renewable energy and to the reduction of CO2 (Manders, 2009).

We have not been able to find any comprehensive analysis of the institutional framework of WtE. According to Antonioli and Massarutto (2013) and Hall and Nguyen (2012), incineration started more or less everywhere in the public sector, with the notable exception of France, where delegation to private companies constitutes a typical feature in all local services. In the last decade, however, a substantial trend towards a wide involvement of the private sector can be detected. This is driven by privatization and merger of local public companies, especially in the energy field (e.g. Germany, Netherlands, Italy), but also by a wider recourse to public-private partnerships (PPP) for new initiatives (e.g. UK). The World Bank (1999) provides a detailed description of the most frequent management models and contracts, which range from the most classical procurement contracts to more sophisticated arrangements, in which the contractor's duties extend to design, operation and assumption of financial risks. Table 1 illustrates their most typical features.

TABLE 1 HERE – SEE BELOW

3. Costs and benefits of incineration

3.1 Financial costs

How much does incinerating waste cost? Such a seemingly simple question hides many conceptual and methodological difficulties and pitfalls, which deserve a careful understanding.

In the first place, there is a need to distinguish between financial costs and external cost (the social cost, which is the relevant one for WM policymakers, being the sum of both).

Financial costs involve the monetary expenditures for construction and operation of a waste combustion facility. They include investment costs and operational costs: both are likely to depend on local circumstances and national regulations, although to some extent standardization is possible.

The literature we have surveyed is quite heterogeneous. Many studies rely in fact on previous work, or provide meta-surveys of original studies. It is not always easy to trace back the analysis to the original source, since many applied studies use second-hand data quoted from works that on their own rely on other sources. At the same time, comparing figures from one survey to the other is hampered by a lack of clarity about the reference year of monetary figures.

Table 2 compares the results of some of the most cited reference works, often officially released in the name of public institutions and national research centers in a number of countries. Since some of them present regression functions and not punctual data, we have applied them to reference plants. Where figures are provided in graphics and not in precise digits, we have made an approximation based on the published graph, which is reasonable for the purpose of making a rough comparison. All values have been converted in €$_{2012}$ using the inflation rate of the country where the study is based, assuming that where no information was available the reference year is the year prior to publication. The $/€ conversion rate is 1,30. The "gross cost" is the total financial cost, while the "net cost" column takes into account the revenues from sale of energy and by-products.

As we can see, figures are not very consistent; however, if we restrict the analysis to the most recent ones and to the EU context, figures become more comparable. The gross cost of an up-to-date facility which respects the very severe EU norms and exploit the economies of scale lies in the range between 100 and 130 €/t.

Table 2 – Financial costs of MSW incineration – survey of some reference studies (all values in €$_{2012}$)

TABLE 2 HERE – SEE BELOW

Source: our elaboration on the quoted studies

Very generally, two approaches may be used for quantifying investment and operational costs. The first one – engineering model – is based on a desktop study, possibly supported by experts in the field, aimed at designing a standard facility. The desktop study usually considers overnight costs for greenfield plants, thence leaving apart site-specific issues (such as land price, monetary and in-kind compensations paid to local communities) and financial considerations. These aspects are later standardized considering a normal situation and making assumptions about the time horizon and the interest rate.

As a facility with large initial investments and a long expected economic life, financial assumptions are particularly critical. Desktop models assume a standard operation time, which accounts for temporary closure for ordinary maintenance and eventually longer stops for planned refurbishing or revamping. Yet these variables are highly variable depending on

Economopoulos (2010) estimates that for the same facility with identical overnight costs, the final full cost would vary by 80% (table 3).

Table 3 – Full-cost recovering gate-fee for a mass-burning facility under different interest rate assumptions (€$_{2012}$/t)

	r = 5%	r = 14%
Small facility (50 kt/yr)	100	180
Big facility (500 kt/yr)	63	113

Source: adapted from Economopoulos, 2010

The second approach uses data collected directly on the field from actually operating facilities. This approach has the merit of a higher degree of realism. However, data collection is not always reliable: information about costs is normally a sensitive data that operators are not willing to disclose, unless they are obliged to do so by economic regulators (this is generally not the case).

Clearly, net costs also depend on the value of recovered resources. In the case of incinerators, these are for the most part electricity and eventually heat. Further revenues may arise from the recovery of materials (eg metals) and from the reuse of ashes as an inert for the construction industry.

Energy prices have their own degree of variability:

- Energy recovery efficiency depends on a number of variables (technology, quality of waste). An optimized plant treating pre-selected waste may recover 2-3 times more electricity and heat than a more traditional plant treating raw waste (Consonni et al., 2011).
- Market price of electricity may depend on national market conditions, since the average market price in each country is a function of the technology mix (for example, in Italy it is significantly higher than in Germany)
- Market value of heat depends on local circumstances, namely the existence in the surroundings of industrial premises that can effectively use heat and/or the feasibility of district heating. Climatic aspects obviously matter, since they affect patterns of heating demand. Many studies conclude strongly in favor of cogeneration, but presume that the recovery of heat does not imply extra costs.
- On top of market prices, eventual subsidies have to be considered (Cossu and Masi, 2013; Dubois, 2013): for example, some countries assimilate waste to a renewable resource and qualify WtE for green energy subsidies; others apply a tax on incineration. In China, the power grid is compelled to purchase electricity generated by incinerators at a special price that includes a subsidy (China Research and Intelligence, 2014). In a social cost-benefit perspective, these subsidies should be omitted since they are clearing entries for the collectivity as a whole

As far as ashes are concerned, the feasibility of recycling and the very existence of a market willing to pay a positive price depend on national legislation. Some countries allow – in certain cases – to recycle them, e.g. as inert materials for the construction industry; while others consider ashes as hazardous waste. Massarutto et al. (2011), in a direct survey on the Italian market, find an average price for ashes of 20 €/t, which generates a total revenue of approx. 1.5 € per ton of waste incinerated (tW). In the sensitivity analysis, they consider the alternative hypothesis of handling ashes as hazardous waste, with a cost of 200 €/t (obtained from market survey as well). This translates into an extra cost of around 7.5 €/tW. Technological innovation in the treatment and recovery phase may have an important role to play in the future (Baciocchi et al., 2010; De Boom et al., 2011; Rocca, 2012; Arena and Di Gregorio, 2013; Bontempi, 2013).

Revenues from the sale of materials directly recovered from the process (e.g. metals) have, instead, a negligible impact on the total balance.

A third possibility is to refer to gate fees actually paid to WM facilities by those who wish to use them. This methodology is not recommended for a social cost-benefit study, for two reasons.

In the first place, financial costs might differ significantly from gate fees. This difference could depend on many factors (World Bank, 1999; Oecd, 2014):

- Deviation of real-world experiences from theoretical optimum: extra costs due to site-specific issues, problems in the start-up phase (e.g. startup delays, unplanned closure, lower than standard load factors). The latter factor alone may generate differences in the reach of 50% of the unit treatment cost.
- Market imperfections: facility owners may enjoy a monopoly position that enables them to charge a price that is higher than the marginal cost, thence obtaining a more-than-average profit
- Complexity of the transaction: one should for example distinguish between spot transactions from those that are framed in long-term contracts and include further aspects that can be assimilated to risk-sharing mechanisms, such as capacity payment and take-or-pay clauses
- Taxation and subsidies: the gate fee is obviously influenced by economic incentives offered by public policies, as already said. Taxation of alternative disposal techniques (e.g. landfill) and subsidies to recycling also have an influence.

Second, it is difficult to have access to a comprehensive set of data concerning price charged by disposal facilities, even because these are usually embedded in long-term contracts; contractors may not be true counterparts (e.g. facilities might be owned by the same municipalities that confer their waste or other public subjects).

Spot prices may be charged in case of an occasional use of the facility (e.g. due to temporary unavailability of other solutions), but are normally not the rule. In the past, incinerators (and especially mass-burning facilities) were typically realized within the frame of a regional plan, and the possibilities to sell spare capacity were limited by legal clauses (such as the proximity principle) and more objective reasons (high transport costs).

With these differences in mind, and being aware of the weaknesses associated to different estimate techniques, we can resume some key points that seem most consolidated.

- As for any other capital-intensive technology, the economic viability of incineration is highly concerned by financial aspects. The cost of capital is obviously influenced by risk-allocation patterns, while the most obvious factors that affect economic risk are time (not only the economic life but also the time lag between the start of the investment project and full-scale operation), the likelihood of accidents, tightening of emission standards and anticipated termination. Since most of these aspects depend on regulation, assumptions related to this dimension should be clearly spelled out.
- Energy recovery implies extra costs that vary in the range of 30% (World Bank, 1999). These costs are higher for combined heat and electricity recovery.
- The recovery of heat is able to improve dramatically the economic performance; yet most studies usually account only for the equipment installed in the incineration plant; while further costs that are needed in order to valorize heat adequately (e.g. district heating networks) are normally not considered.
- Pollution abatement is the most important cost driver for modern facilities. Abatement equipment may account up to 60-70% of the investment and operational cost.
- Economies of scale are fundamental. The efficient scale varies from one study to another, but it is apparent that investment and operational costs diminish at least until 500-600 kt/yr. The key point,

however, concerns the possibility to really use up a facility with this capacity; population density becomes a key factor, since transport costs have a meaningful incidence.

3.2 External costs

Externalities caused by incinerators are fundamentally due to (i) air pollution; (ii) further pollution eventually caused by the disposal of incineration by-products, such as ashes and (iii) disamenities. Furthermore, incineration has important consequences in terms of climate change, due to GHG emissions. Of course, since the recovery of electricity and heat displaces other energy sources, the analysis should account for the respective emissions as social benefits.

A very well-known family of models adopts the so-called impact pathway analysis approach. This foresees an engineering module describes the expected emissions; climatologic modules simulate the resulting ambient concentration; epidemiologic modules associate ambient concentrations with mortality and morbidity, and finally an economic module attribute a monetary value applying external cost valuation techniques. An example is provided by the ExternE project, funded by EU framework programs, and by its follower CAFE (Bickel and Friedrich, 2005))

The difficulty in generalizing results from these studies lies in the fact that emissions and related polluting potential depend on technology adopted and site-specific features.

A great number of epidemiologic studies show that older facilities (with no or insufficient flue gas treatment) were responsible for serious air pollution and related damages to health. Many literature review studies point out that this evidence is far from conclusive, often hampered by methodological weaknesses and by a lack of a serious consideration of confounding factors (Franchini et al., 2004; Giusti, 2009; Hu and Shy, 2013; Cordioli et al., 2013). Nonetheless, most applied economic valuations assume these results as a starting point, obviously concluding that the external cost is significant.

Table 4 – A review of economic studies quantifying external cost of incineration (all values in $€_{2012}$/t)

TABLE 4 here

Source: our elaboration on Eshet et al., 2006

Eshet et al. (2006) review 13 studies published between 1992 and 2003. Their results are summarized in tab. 4: it reports in the columns the monetary values per ton of waste incinerated, arising from some typical sources of impact. Benefits arising from the displacement of emissions from the replaced energy sources are also considered. The total does not necessarily correspond to the sum of the columns.

Although results of these studies are not immediately comparable – since they apply different methodologies and refer to different sets of facilities – it is interesting to note that the range of estimates oscillates between 8 and 39 €/t on average. The top figure is influenced by two studies, which arguably include climate change issues (CO2 emissions), whereas they also consider old facilities with no energy recovery. The median value varies between 4 and 10 €/t.

The comparative review also enables to notice the “Chinese whispers effect” that characterizes this literature: many articles do not rely (totally or partially) on original research, but adopt to a more or less wide extent a “benefit transfer” approach – that is, using numeric results derived from other studies; the same result are reprised and rebound from one study to another, often as third- or fourth-hand citations. The result is that in many cases even recent articles report value of external costs arguably referred to older polluting facilities. Moreover, while the figures remain, assumptions may be forgotten and the caveats set forward in the original study may easily be lost.

Anyway, evidence about modern facilities – e.g. those respecting the demanding targets imposed by EU Incineration Directive or by even more strict national standards adopted in Germany or in the Netherlands – shows that the differential impact above the bottom threshold of a standard urbanized area are almost nothing (Shrenk, 2006; Federico, 2010). Assuming this is true, it should imply that external costs associated to emissions should also be almost nothing.

In fact, such severe emission standards have been imposed precisely because of the political will of adopting a precautionary principle, following evidence from the past. Official statements by many public health authorities in EU member states affirm that the risk of damages to health in proximity of newly built and well managed MW incineration facilities is "exceedingly low and probably not measurable by the most modern techniques" (UK HPA, 2009); similar statements are expressed by UBA, 2008 and WHO, 2007.

It must be said that this official position is not shared by a minority (though very influential on media and NGOs). This still raises doubts about the potential harmful effects associated, in particular, with micro-pollutants conveyed by nanoparticles; however, at present no clearly convincing empirical evidence can be cited, at least from peer-reviewed studies, about the existence of a causal link. Recent studies on ambient concentration of nanoparticles show no evidence that incinerators represent a major cause, urban traffic and other sources being most likely candidates (Cernuschi, 2013; Buonanno and Morawska, 2014).

As already said, a further aspect that explains differences in results concerns the environmental benefits that depend from the displacement of alternative sources of electricity and heat generation. This is controversial, for two reasons. In the first place, once assuming that WtE can generate a certain amount of electricity and heat, the alternative sources depend on the specific patterns of energy supply that characterize each country. Avoided emissions are highest in the case of coal-fired power generation and oil-based heating, and lowest in case of renewable sources or nuclear plants.

Disamenities – or, better to say, perceived externalities – are a far less investigated field. Most disamenity studies in the literature are concerned with landfills, but a few of them also consider incinerators.

The most common valuation techniques are the hedonic price method (HPM) and contingent valuation (CV) (Brisson and Pearce, 1995; Huhtala, 1999). The former is based on the analysis of the behavior of markets whose values are likely to be affected by the presence of a point-source of externality (typically: land and real estate). The latter in turn uses direct interviews aimed at enlightening the willingness to pay (WTP) for a state of the world in which the externality disappears, or the willingness to accept a compensation (WTA) for the opposite. Most often, interviewed people express a strong preference for recycling and an opposition to incineration.

The main problem with incineration is that only a few dimensions of externality are directly perceived (e.g. landscape, noise, smell). For dimensions that are more closely related to human health, since the causal link is not obvious, perception is mediated by personal beliefs, which are likely be shaped by the social discourse (Gupta et al., 2011; Cavazza and Rubichi, 2013; Achillas et al., 2011) and, more generally, depend on the social attitude towards risk (Lima, 2004; Aldy and Viscusi, 2014). Socio-cultural aspects matter as well, as the "dustbin-effect" (the refusal of being treated as someone else's disposal site) outlined by Dente et al., 1998, or the refusal to accept economic compensation (Ferreira and Gallagher, 2010).

An emerging field of research concerns applied studies aimed at capturing the altruistic and ethical motivations of human behavior. Many recent studies provide evidence in favor of the existence of a "social norm to recycle", which actually means that individuals attach a positive value (warm-glow utility) to actions that result in a reduction of their impact on the global environment (Abbott et al. 2013; Aadlan and Caplan, 2005). Again, however, it is difficult to translate this "first order" preference into a "second order" one that more precisely concerns incineration, since the attitude towards waste-to-energy is seemingly framed by social beliefs. In other words, altruistic- and ethically-motivated individuals that consider a good thing to adopt a behaviour that combats climate change in general may be favorable or not to incineration in particular, depending on what their beliefs are about this technology.

4. Economics and integrated assessment

In the technical literature, combustion of waste aimed at energy recovery is mostly regarded as a key element of an integrated waste management strategy (Cossu, 2011; Brunner and Rechberger, 2014).

In the last decade, life-cycle assessment (LCA) has gained consensus as a reference valuation tool for integrated waste systems. A reason for its success is the capability of capturing in an integrated way ("from cradle to grave", i.e. considering all direct and indirect sources of emission that are involved by a given cycle) the five most important dimensions of environmental impact: energy demand, global warming, acidification, human toxicity and ozone depletion. LCA has rapidly conquered the favor of scholars, also thanks to the availability of ready-to-use software. In our survey, we have found more than 60 published studies that apply this methodology, most often on a material-specific basis or for the comparison of individual technologies and in some cases for integrated waste management scenarios.

For the purpose of the present paper, we can limit our discussion to a few meta-studies that provide a comparative assessment (Cleary, 2009; Profu, 2004; WRAP, 2010). LCA studies show in quite a consistent way that systems based on WtE dominate simple landfilling, even when landfills are available without bottlenecks. Recycling is generally superior when it concerns clean and well-separated materials such as plastics and paper.

Despite this widespread success, LCA is not exempt from criticism. Ekvall et al. (2007) identify the main methodological weaknesses in the specific of waste management as follows: lack of methodological standardization, with many key assumptions left to uncontrollable discretional decision of scientists; failure to consider time and space; linearity of effects; incapacity to consider site-sensitive variables and specific features of background systems; failure to consider non-environmental impacts. Through an appropriate design it is nonetheless possible to avoid or reduce such shortcomings: e.g. Clavreul et al., 2012, propose an approach to deal with uncertainty; even though, weaknesses remain, and do not allow to use LCA as a definitive support tool.

From an economic viewpoint, however, LCA has an even more decisive shortcoming, which is closely related to the way economics regards the concept of efficiency. In fact, LCA fails to consider the economic cost, and tends therefore to overrate systematically solutions that achieve a better score in terms of the environmental variables. Even when economic costs are considered – for example in the "life-cycle costing" approach – these are usually assumed as constant parameters and linear relations.

For economics, in turn, the crucial concept is that of *marginal cost*, i.e. the additional cost that is implied by an additional quantity of something. A typical assumption of economic models with this respect is that of *diminishing returns*, or *increasing marginal cost*. Beyond a certain threshold, all technologies tend to exhibit higher unit costs, due to some diseconomies that inevitably appear. In the case of waste, this applies in particular to recycling. If recycling clean and well separated plastics is clearly and uncontestably superior to incineration or landfilling, this cannot mean that all plastics should be recycled, since only a few fractions are easy and cheap to obtain in a clean and well-separated way.

Economics has developed its own approach to integrated evaluation, namely the cost-benefit analysis (CBA). However, the literature applying CBA to integrated waste management is surprisingly very thin. Despite the heavy criticism that economists normally raise against assessment methods such as the LCA, economics has failed so far to offer an analogously comprehensive tool.

To summarize in a sentence the economist's perspective, "recycling is a good thing, but not at any cost" (Porter, 2005). This is fundamentally due to rapidly increasing marginal costs for higher per-capita collection of recyclables (Kinnaman, 2014) and to the arguably lower quality of materials collected when the source separation ratio increases, thence entailing more costly processing and reduced marketing opportunities (Massarutto, 2012). Market imperfections in the downstream of recycling need also be considered (Massarutto, 2014)

Thorneloe et al., 2007, use the decision support tool elaborated by the US EPA to compare a set of alternative scenarios with varying degree of recycling and alternative options for the residual. They find the socially optimal rate of recycling in the range of 20%. Kinnaman (2014) uses a sample of Japanese municipalities to show that the optimal rate of recycling from the perspective of the municipality is 10%, and jumps to 36% when social costs are considered.

Using a similar conceptual framework, Pearce (2005) argues that the EU waste policy targets by large fail to pass a cost benefit test, even if the full social cost of landfilling and incineration is duly accounted for, also considering GHG emissions.

For the remaining residual waste, the extent to which incineration should be preferred to landfill is even more controversial.

Landfilling implies lower financial costs than incineration, which is in turn socially advantageous once climate change and pollution are duly considered (Thorneloe et al., 2007). Pearce (2005) compares the social costs and benefits of landfill and concludes that evidence is favorable to landfill, until absolute scarcity of location for new landfills does not lead to substantial scarcity costs. Dijkgraaf and Vollenbergh (2004) reach a similar conclusion for the Netherlands, and on this basis criticize the EU mandatory priority ranking of solutions, which privilege materials recovery in the first place and waste-to-energy with respect to landfill.

Barrett and Lawlor, 1997, show the importance of population density when establishing priorities: in low-populated areas, landfills may still represent a preferable option for residual waste. Scale and energy recovery capacity are also found to be a key driver in Ireland (Donovan and Collins, 2011).

Chang and Davila (2007) reach a similar conclusion by considering that a landfill-based waste management system, although it may benefit from low landfill costs when landfills are abundant, are very vulnerable to shocks in the supply of disposal facilities; an integrated strategy, with a balanced mix of composting, material recycling and WtE arises as the most recommendable option.

Cucchiella et al. (2014) estimate the potential benefits, in terms of both financial cost and social cost, of varying degree of substitution of WtE to landfill for unsorted waste, ranging from 25% to 75% of residual waste incinerated. Both indicators are clearly positive and increasing with the fraction incinerated under standard assumptions; nevertheless, the result is vulnerable to some key assumptions. The sensitivity analysis shows that a variation of ±2% in the interest rate may affect the total benefit by -100 - +200%, turning the output into a negative figure when the interest rate is above 7%; a variation of initial investment of ± 20% has an impact of -153% - + 105%, whereas +20% is the threshold for keeping the output positive. The result is more robust to other assumptions concerning the market value of electricity and heat and the lower heating value.

Marklund and Samakovlis (2010) show that it is not optimal to impose material recycling of all paper, even after separate collection; for a part of it, energy recovery performs better in economic and environmental terms.

Jamasb and Nepal, 2013, discuss the UK waste management strategy comparing a "business as usual" scenario with the full implementation of the EU waste directive. Their study largely relies on benefit transfer, since figures for external costs due to air pollution and GHG derive from the literature. Incineration involves higher financial costs than landfill or recycling and appears as a more expensive source of energy generation than coal; however, if both aspects are considered together, it turns out as the dominant technique in terms of social cost-benefit.

The meta-analysis of waste optimization models conducted by Juul et al. (2013) shows, more in general, that once the focus is placed on an integrated set of objectives, incineration emerges as a pivotal technique, although the mix of alternative solutions may vary.

Reich (2005) applies a LCC framework for comparing alternative scenarios for MSW treatment in Sweden; most of the examined scenarios are focused on incineration (with some material recycling of plastics and paper): all of them result significantly preferable to an "all landfill" one.

Massarutto et al., 2011, apply an LCC approach to a set of scenarios characterized by different targets of source separation of recyclables, with the further constraint of considering waste until the moment it is finally discarded into soil or air, thus including also the "secondary" waste flows that originate from preparation for recycling downstream of separate collection. The study concludes that under the key assumptions of (i) increasing marginal costs and (ii) high scarcity costs of landfills, the optimal strategy implies a combination of material and energy recovery in fairly similar shares. This result once more emphasizes complementarity – rather than the opposition – between material recycling and energy recovery: the social optimum implies a separate collection system based on drop-off containers, while the residual is treated in mass-burning facilities, optimized for the recovery of electricity and heat.

Most of the applied studies just cited provide an ex-ante normative analysis. An ex-post evaluation based on the performance of SWM services is still a poorly developed field of research. In their systematic review of the economic literature dealing with efficiency of waste management, Simoes and Marques (2012) find that most studies do not even consider the choice of techniques as an explanatory factor (Simoes and Marques, 2012)

Among the few original studies using this approach, Bel and Fageda (2010), focusing on Spain, find that municipalities using an incineration plant have a significantly lower overall waste management cost. Simoes and Marques (2011) obtain a similar result for Portugal. Antonioli and Massarutto (2013) argue that the efficiency of SWM systems that adopt incineration is higher when landfill becomes a residual option (because of explicit government policies, e.g. landfill bans, or due to absolute shortage); evidence showing substantial savings for municipalities that do not actually incinerate depends on the fact that subsidies to recycling paid by EPR systems are very high.

However, the Oecd (2014), affirms that a positive contribution of incineration to the overall efficiency of waste management systems crucially depends on the accurate planning of incineration capacity: if entry in the incineration market is excessive, for example because of incentives and subsidies offered by governments, the risk of over-capacity is patent; in that case, the actual average costs is much higher.

To sum up, there seems to be quite a convergence in the literature about the fact that the social convenience of destining a certain quantity of waste to combustion depends on (i) the diminishing returns of material recovery; (iii) the absolute scarcity of landfills and (iii) the safety for health of modern incinerators.

The first assumption is quite reasonable given the actual state of the art of waste generation, especially in urban settings. Although this evidence is contradicted by an emerging evidence of "zero-waste" champions – cities or neighborhoods that manage to achieve source separation levels of 80% or more, thence requiring little or no residual waste management – many concerns have been raised about the possibility to generalize such targets to all communities.

Extreme recycling scenarios can compete only if curbside collection systems allow reaching 75% or more of separate collection, and if quality of materials does not worsen too much. This is a realistic assumption for small cities and rural areas, but probably not for urban areas, where the source separation level cannot seemingly be higher than 50-60%: in this case, the social cost is significantly higher.

In addition, the quality of separately collected materials is usually likely to decline with the separation rate, thence reducing the value of sorted materials and the very possibility of redirecting them to the productive system. In other words, intensive separate collection aimed at maximizing source separation may simply shift the problem of waste downstream, rather than promoting effective recycling. In the best cases, material recovery of very impure streams means that waste-derived materials are "down-cycled" rather than recycled – whereas LCA implications are far less investigated.

The second condition – about landfill prices – is quite reasonable for densely populated areas, as is the norm in Europe or in Japan, but not necessarily in the US or elsewhere (Kinnaman and Fullerton, 1999). If the cost of landfill comprises only financial costs and externalities – assuming standard facilities with adequate technical solutions for minimizing pollution – the superiority of combustion is questionable; yet if absolute scarcity emerges, the market price may very suddenly increase. The key issue, therefore, concerns the ongoing capacity to develop new landfill sites or to transport waste elsewhere. The market price may not be an adequate proxy of the scarcity cost.

An estimation of the scarcity cost has been the object of only a few pioneering studies. In the Province of Udine – one of the largest in Italy, extended from the mountains to the sea – not a single location is to be found anymore, once the standards imposed by the requirements imposed by EU landfill directives are respected; while in Lombardia, the most densely populated region, landfill prices rocketed from 20-30 to 150-200 €/ton or more in a few years, due to the failure to find suitable locations for new facilities (Massarutto, 2012). Comparing the average spot market price over a longer period and comparing it with the financial cost of a standard facility, this study concludes that a scarcity cost in the reach of 50 €/t is a very plausible figure.

As discussed above, the epidemiological evidence about modern facilities seems to confirm the third assumption. Most of the studies found in the literature showing adverse effects on human health were actually based on older techniques, which have been completely phased-out by new ones. In fact, many CBA studies still adopt benefit-transfer values of externalities that are based on past evidence, rather than on modern facilities, and may seriously overestimate external costs. However, in case future evidence would reveal proofs of further damages to health – e.g. caused by nanoparticles – this consideration would be obviously not true.

A further critical point concerns the valuation of externalities that are due to the competing MSW management techniques. External cost assessment studies applied to different technologies use different methodologies and address different issues; thence their results are not immediately comparable. In the case of landfill, for example, most externality studies are concerned with disamenity, but nothing comparable to the very detailed impact pathway analysis performed for incinerators has ever been made. Contamination of soil due to leachate or hazardous substances, for example, is very difficult to model, due to non-linear site-specific effects.

Once again, one thing is to discuss the potential impact of uncontrolled dumping, and another one is to consider modern facilities, conveniently equipped with pollution control equipment, constructed in convenient sites, adequately managed etc. However, a trade-off seems to emerge between minimization of pollution and availability of sites. Geography, density of settlements and patterns of regional development are obviously key drivers. In Northern Italy, for example, suitable sites have been dramatically reducing in the last 20 years.

A related critical factor concerns benefits obtained from the displacement of alternative polluting energy sources. Seemingly, if energy obtained from waste combustion displaces polluting sources (coal, oil) advantages are much higher than when the alternative energy mix is based on renewable resources.

A final point concerns the evaluation of externalities once accounting for the existence of illegal activities (Kellenberg, 2010 and 2012). It is well known that the waste sector offers enormous spaces to criminal organizations. The economic rationale for this is the negative value of waste (owners are willing to pay to get rid of it) combined with a very strict legislation imposing high treatment costs (Massarutto, 2012). Although all waste management, techniques are in principle vulnerable to illegality, it is arguably easier to enforce legislation in the case of identifiable facilities with continuous monitoring of emissions – as incinerators ultimately are – than in the case of landfills, where pollution becomes manifest after a long time. Recycling, as well, can easily mask illegal dumping, particularly when the value chains are longer and more globalized (Yokoo, 2014).

5. Comparative assessment of WtE options

Waste-to-Energy can be obtained fundamentally in two ways. The first one requires that waste (either "raw" or pre-treated in some way) is utilized in dedicated plants. The second approach consists in the transformation of waste into something that can be used in already existing plants that burn other (solid) combustibles for their process. An example of the latter is the combined used of RDF in cement kilns, coal-fired thermo-power facilities, steel production etc.

The latter alternative has the advantage of using already existing facilities; yet its feasibility is constrained by the possibility of actually finding a reliable market that is capable of absorbing the entire flow of combustible waste. Moreover, it also implies that the materials discarded from the upstream treatment phases can find a proper destination as well.

According to the Oecd, 2014, market imperfections may reduce the economic attractiveness, while the relatively little importance of coal as a combustible for electricity and industrial power generation limits the set of available opportunities. It is also interesting to notice that the increase of the economic convenience, driven mostly by increasing landfill prices, has brought to a significant increase of exports, particularly towards northern and eastern Europe.

Massarutto (2013) provides an empirical proof of this thesis, through a detailed analysis of the Italian market for RDF in the last 20 years. He shows that despite theoretical expectations, this market has never really developed; transactions are mostly bilateral and only in a handful of cases concern industrial plants, while most of the times RDF ends in dedicated incinerators.

RDF looks a far more promising strategy in developing countries, particularly those in which solid fuels are already widely used. This is for example the case of Malaysia (Abd-Kadir et al, 2013)

When RDF has to be treated in dedicated incinerators, its attractiveness is largely reduced. In the already cited study on municipal waste scenarios, for example, Massarutto et al. (2011) show that it is largely more efficient to destine untreated residual waste to mass-burning facilities rather than searching for higher and better energy recovery rates by processing waste in mechanical-biological facilities or producing RDF, especially when these waste-derived combustibles are to be burnt in dedicated plants. The key advantage of RDF lies precisely in the possibility to use already existing plants.

However, this result strongly relies on the possibility to exploit fully the economic potential of mass-burning plants. For example, when economies of scale are not fully exploitable because of transport costs, the convenience of mass-burning plants becomes less dominant, and other solutions may gain.

Murphy and McKeogh (2004), focusing on Ireland, argue that the efficiency of mass-burning is strictly depending on the possibility of recovering heat in combination with electricity; where it is difficult to find a market for thermal products, the superior energy recovery efficiency of gasification may rank this solution better. The same authors (2006) also show the advantages of combining different waste streams in order to optimize the energy recovery potential of gasification. Field experiences seem to demonstrate this result. The most successful

Pre-treating waste in order to stabilize their heat potential allows to use smaller facilities (Trulli et al., 2013)

Hellweg et al. (2005) use the result of LCA for the construction of an "environmental cost-efficiency indicator", showing that (i) basic grate incineration provides a substantially better environmental impact than alternative solutions based on landfill (worst) and MBT, though at a higher financial cost. The study also shows that an enhanced "staged" incineration, with the treatment of ashes and slags, may substantially improve the environmental performance for the same financial cost

6. Waste incineration and the market

Waste incineration poses to economists a further important set of questions, about regulation and governance. Assuming that “some” incineration is desirable, who and how should provide it? Under which institutional regime? Should the issue be left to the interplay of supply and demand on the market, or should the public sector have a role?

This issue is clearly related to the capability of self-regulated competitive markets to provide “the right” supply of incineration capacity. What are the eventual market failures that impair this solution?

For example: should incinerators operate in a competitive market system, or should facilities be planned by the government and operate as a public service? And in the latter case, what are the most appropriate ways to select the operators in charge of developing the system? What is the role for the private sector? Will facility owners acquire a market power? Should waste producers be left free to choose whether to use or not to use the facilities? Should gate fees be fixed by direct negotiations between waste holders and facility owners, or should they be regulated, and how? Should incinerators be compelled to operate within a given area (as under the regime of the self-sufficiency principle) or left free to sell their capacity at convenience? Should the risks associated with the construction and operation be borne entirely by facility owners, or should the state participate, for example by guaranteeing subsidies, soft loans, capacity payment or similar?

Strange enough, these aspects, which are crucial in the economic analysis of any utility, from electricity to telecoms, from transport to water supply, have not met until now an analogous interest in the case of waste management and more specifically waste-to-energy. Economic regulation of waste management has deserved only limited attention, and contributions that dedicate a specific attention to WtE are even scarcer.

Anyway, mass burning WtE facilities exhibit many of the typical features that are normally associated with market failures on the supply side.

Economies of scale are very significant, and have dramatically increased due to the high fixed cost of equipment for pollution control and energy recovery. The economically efficient scale is at least 500-600 kt/year (World Bank, 1999; Dijkgraaf and Vollenbergh, 2005). Assuming a yearly production of 500 kg/person and that 30-50% of MSW is actually suitable for such a treatment, this corresponds to an equivalent population of 2-4 million inhabitants. Since transport costs are significant, this means that it is normally not economically feasible to have more than one plant serving the same area, unless in very concentrated urban regions.

Furthermore, the possibility to exploit efficiently the heat content depends on the availability of local uses of steam, either in industrial processes or for district heating. These conditions are more probably found in the proximity of large urban areas.

Capital anticipation is significant. Initial overnight investment costs can be estimated in the reach of 200-250 M€ for a facility of 500 kt/yr. Even small variations with respect to initial assumptions (concerning the start of operation, the load factor etc) have a dramatic impact on costs (and therefore on profitability. Since the load factor depends on the waste flow that municipalities will be willing to burn, it will be affected by the long-run evolution of waste generation patterns, as well as by the possible emergence of alternative solutions.

Many studies also demonstrate that the control of suitable disposal capacity represents a barrier to competition in the waste management sector as a whole. Waste collection, for example, could be easily opened to competition via auctions and tendering; yet this is feasible only when collectors; yet this is feasible only when all collectors have the same right to access to disposal solutions at analogous conditions (Oecd, 2014; Buclet and Godard, 2000; Massarutto, 2006).

Risk of lock-in is also significant, due to the high sunk costs involved by incineration. Due to this, an early investment in incineration capacity may slow down the development of more sustainable techniques in the future (Corvellec et al., 2013).

A further assumption that is typical of regulatory economics postulates that the absence of competition will reduce the incentives to operate efficiently, especially when facilities are allowed to charge their full cost to customers. Even though the topic is not frequently analyzed, there is some empirical evidence that confirms this.

In the EU context, for example, Marques and Simoes (2008) argue that the "sunshine regulation" model adopted in Portugal – based on the systematic publication of benchmarking information had a positive effect in promoting efficiency.

Using a DEA approach, Chen et al., 2014, find that Taiwan's incinerators have a 15% cost reduction potential, and suggest a regulatory scheme, based on the identification of efficient benchmarks; only firms operating at the benchmark would be allowed to recover their full cost.

Natural monopoly is clearly enhanced by the barriers to trade. Transport costs are the most obvious, and until recently the decisive one: until disposal could be obtained quite easily, there was also no economic justification for waste shipments outside the local market. Trade opportunities were rapidly exhausted in the reach of few tenths of km. The increasing economic value of waste management options, driven by the emerging bottleneck on the side of landfill capacity, has rapidly changed this picture.

However, another kind of barriers is still present, in the form of regulations and norms that limit or prohibit waste trading. In Europe, for example, the "self-sufficiency principle" establishes that handling of waste should be done in the same area where waste has been produced; this principle can nonetheless be derogated in case waste derived materials are intended for recycling or recovery of resources.

Most countries adopt restrictions to the free circulation of waste, at least for MW. Disposal and treatment facilities are rarely left free to supply their services on the open market; rather, regional plans often allocate, in a more or less binding way, MW flows to specific facilities, or adopt waste import bans, prohibiting facilities located in their territory to receive waste from outside.

Regional planning emerged as a dominant regime for MSW in the 70s (Buclet and Godard, 2000; Massarutto and Antonioli, 2013). Prior to that, disposal was effectively supplied by the market (landfill)

From an economic viewpoint, such restrictions are hard to justify. The essence of trade is precisely that of allowing each individual, company, region and country to specialize in the activities it is more suited for. In principle, there is nothing bad if a congested area – where the cost of land is higher because of its scarcity, health-related externalities are higher due to the higher population density etc – "buys" waste services from areas with an opposite vocation. Ley et al. (2002) estimate that the cost of imposing restriction to the trading of waste among US states could reach 18 $ per capita/year.

Pollution alone cannot be an answer: electricity generation and steel production pollute at least as much as waste incineration, yet no "self-sufficiency" principle has ever been advocated. If Italy imports electricity from France and steel from Germany or China, why couldn't the same countries burn some Italian garbage?

This is particularly the case of incinerators. However, in recent times this general rule has become more porous and open to exceptions. Focusing on Europe, the picklock is represented by the concept of waste valorization. While pure disposal is normally subject to territorial restrictions, waste trade is increasingly accepted whenever it can be considered as a resource for some other productive cycles. In the case of combustion, a conventional threshold has been established (1600 .

Massarutto (2012) argues that this distinction is problematic, due to the fact that valorization cannot be simply presumed by the fact that waste-derived by-products are effectively used by other processes. In many cases, material or energy recovery are not economically justified as such, but only because they represent a cost-effective alternative to otherwise more costly disposal practices. This obviously depends on the cost of the best available alternative to recycling/recovery, for example landfill. Waste owners may be willing to pay at least that amount to receiving processes: instead of paying for receiving valuable materials, the latter will thence be paid for providing alternative disposal services. Nor is it simply possible to infer recovery from monitoring the direction of payments, since it would be very easy to mask it via indirect exchange.

Many studies suggest that an important argument against waste trade lies in the difficulty of controlling their actual destination. Even though these studies have been initially focused on disposal into landfills, considerations can be similar in the case of incineration.

Copeland (1991) first had the intuition of considering asymmetric regulation and enforcement, concluding that waste flow controls may be welfare-improving when external costs of garbage are not internalized, and there is no uniformity in regulation and/or enforcement. Brusco et al., 1996, extend the concept of “regulatory capture” to enforcement and control, showing how low-quality waste disposal may displace high quality solutions forcing them out of market, benefitting from loose enforcement. Macauley at al. (1993) shows that allowing landfills the option to practice third-degree price discrimination (i.e. charging a lower price to local customers and a higher one to imported waste) may make both local residents and the rest of the world better off.

Even not considering illegal dumping and “pollution havens”, asymmetric regulations (and asymmetric enforcement) could easily trigger a “race-to-the-bottom” (Buclet, 2002). Countries and regions where available sites for landfilling are relatively abundant may have not felt yet the necessity to implement tight regulations; domestic waste production in less-developed economies. Foreign waste may rapidly exhaust their capacity; this could force domestic authorities to adopt stricter regulations.

Massarutto (2007) combines these arguments and concludes that conditions may be very diverse before and after the initial investment is made. Before the facility is built, the perceived risk is so high that the willingness of market operators to initiate a WtE project is limited (at least, without some protection from competition and some guarantee of the total revenues). Moreover, landfill owners may exploit their monopoly position and charge a high price (which would make incineration worthwhile), but could threaten to lower this price after the incinerator starts its operation, thereby reducing the convenience to use it. In order to create an incentive to invest, authorities should use a mix of policies that include some guarantee of the waste flows destined to the plant (compulsory planning, limitations or bans to waste trading, self-sufficiency principle) combined with instruments that discourage landfilling early in advance with respect to the moment when the scarcity cost will appear (e.g. landfill taxes).

However, once enough facilities are built, this situation is not anymore true. The risk of insisting in the above measures will easily lead to overcapacity. As also outlined by the Oecd, (2014) an excess capacity increases the risk of a lock-in, since efforts to minimize and recycle waste are worthless, given the low marginal benefit. In order to prevent such situation, the regulatory approach should change early enough, in order to reduce gradually the barriers to competition.

Massarutto and Silvestri (2014) develop a theoretical model with two areas, one served by an incinerator and one not. The institutional setting foresees that the WtE facility is legally bound to treat all the waste originated in the first area at a regulated price, and can sell the extra capacity on the market. Either area 1 or 2 have a strong incentive to reduce waste: in the first area, because more extra capacity will be sellable on the free market at a higher price; in the second area, in order to avoid paying the WtE gate fee. If both areas were equipped with a WtE plant, the same would not be true, and each area would be willing to fully exploit the plant’s capacity before engaging in efforts to reduce waste.

This theoretical prediction is confirmed by the observation of the real market development. It is not a case that the countries with the highest degree of reliance on waste combustion have also been the first to introduce direct competition in the market, progressively eliminating restrictions to waste trade (Berthoud, 2011).

It seems quite clear, in turn, that waste trade, at least within the EU, has been mostly motivated by an excess capacity available in some countries –Germany and the Netherlands among others (table 5).

Table 5 – Excess incineration capacity in Northern European countries in 2010

	Actual capacity in WtE plants	Available combustible material from waste	Excess capacity	
	Mt	Mt	Mt	%
BEL	2.7	2.5	0.2	7%
NL	7.4	6.3	1.1	15%
GER	24.4	20.2	4.2	17%
DK	3.5	3	0.5	14%
SWE	5.2	4.6	0.6	12%
NOR	1.8	1.5	0.3	17%
TOTAL	45	38.1	6.9	15%

Source: Tolvik, Prognos and Profu, 2011 (from REF-E, 2014)

The NL case is enlightening. After opening to transboundary trade of non-hazardous waste for incineration in 2007, imports have risen to more than 350,000 t in 2007. Around 15% of waste treated in Swedish WtE plants comes from other countries (Birnstegel, 2011).

7. Conclusions

Combustion of waste is a highly debated and controversial issue worldwide. The contribution of economics to this debate, in our opinion, though still insufficient and to some extent inadequate, has provided a number of valuable insights. The first and most important concerns the concept of efficiency, which is still too often misunderstood.

For example, the European Union's "roadmap to a resource efficient Europe", affirms the target of confining thermal treatments of waste to non-recyclable materials (European Commission, 2011). Yet what does "non-recyclable" mean in practice? Economists would contend that this is not just a matter of technical feasibility. Most things are technically recyclable, provided that enough efforts are put in place to separate, handle, process and prepare them for recycling. But is it really worth doing? If repairing a computer with a broken component requires more time and higher efforts than making a new one, what is really "a waste": to destroy it in environment-friendly way, or to allocate labor and capital for recovering it?

Economic reasoning is based on trade-offs and assumes diminishing returns in every productive activity, including the recovery of recyclable materials from waste flows. This is generally more than sufficient for postulating that recycling should not go beyond a certain threshold.

Many non-economists suspect that incineration and recycling are ultimately in conflict – at least for the materials with the highest caloric power, such as plastics, paper and wood. The economic viewpoint, in turn, reveals a persuading reason why this seeming conflict may turn into a complementarity: not because both solution address different material flows, but because they address different flows of the same materials: recycling suits those that are easier and cheaper to select, while WtE better suits the others.

Ex-ante analysis based on cost-benefit valuation assigns a crucial role to incineration, especially where significant bottlenecks arise from the side of landfills, and provided that the actual effectiveness of pollution control is as high as postulated. Both assumptions seem reasonable in the actual market conditions, but not necessarily once forever.

However, real market data show that it is not necessarily that simple. Modern incinerators emphasize the typical market failures associated to capital-intensive facilities; but on the other hand, the rapid expansion of the territorial size of the market – increasingly globalized – calls for innovative approaches to market regulation, which are still mostly to design. This will be, in our opinion, one of the most promising fields of research for economic analysis in the next years.

A further aspect that deserves more careful investigation concerns the interplay between recycling and energy recovery. The reason of the disappointing development of the market of waste-derived fuels, for example, is worth analyzing, possibly with the same approaches that have led to recognize material recycling as incomplete markets, calling for the development of innovative institutions such as those based on extended producer responsibility. The long value chains that are implied by processing of waste for recovery, and eventually for disposal as industrial waste once these downstream phases discard further materials is still not well understood. The interplay between municipal and industrial waste combustion is still not investigated.

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Table 1 – Applicable tendering and contractual models for waste incineration plants

Tender model	Client's obligation	Contractor's obligation	Advantages	Constraints
Multiple contracts	Financing. Function specifications, tendering, project coordination, and construction supervision. Ownership and operation	Supply and detailed design of individual parts for the plant	Full client control of specifications. Possible to create the optimum plant based on most feasible plant components	Absolute requirement for project m nagement and waste incineration skills in the client's organization
Turnkey contract	Financing. Function specifications, tendering, and client's supervision. Ownership and operation.	Responsible for all project design, coordination, and procurement activities	One contractor has the full responsibility for design, erection, and performance	Limited client control of choice of plant components
Operation contract	Multiple or single turnkey contract. Ownership. Supply of waste	Operation of the completed and functional plant in a certain period	Limited strain on the client's organization	Difficult for client to secure affordable tariffs, (put or pay contract), control finances, and monitor contractor's performance and service level
Build Operate	Financing, function specifications, tendering, and client's supervision. Ownership. Supply of waste	Detailed design, project management, contractor's supervision, operation, and maintenance	Contractor committed to well-functioning and effective solutions. Limited strain on client's resources	Difficult for client to secure affordable tariffs (put or pay contract), control finances, and monitor the contractor's performance and service level
Design Build Operate	Financing. Overall function specifications and tendering. Ownership. Supply of waste	Detailed design, project management, supervision, operation, and maintenance. Ownership.	Contractor committed to well-functioning and effective solutions. Limited strain on client's resources	Difficult for client to secure affordable tariffs (put or pay contract), control finances, and monitor the contractor's performance and service level. Limited client control of choice of plant components
Build Own Operate Transfer	Overall function specifications and tendering. Ownership after transfer. Supply of waste	Financing, design, project management, supervision, operation, and maintenance. Ownership until transfer	Contractor finances, constructs, and operates the plant for a period after which the plant is transferred to the client. Very limited strain on client's resources	Difficult for client to secure affordable tariffs (put or pay contract), control finances, and monitor the contractor's performance and service level. Limited client control of choice of plant components
Build Own Operate	Overall function specifications and tendering. Supply of waste	Financing, ownership, design, project management, supervision, performance guarantees, operation, and maintenance	Client does not need to finance the project. Contractor committed to well-functioning and effective solutions. Very limited strain on client's resources	Difficult for client to secure affordable tariffs (put or pay contract), control finances, and monitor the contractor's performance and service level. Limited client control of choice of plant components.

Source: adapted from World Bank, 1999

Table 2 – Financial costs of MSW incineration – survey of some reference studies (all values in €$_{2012}$)

Source	Ref year	Size	Gross cost	Net cost	Note

		kt/yr	€/t	€/t	
Dijkgraaf- Vollenbergh, 2004	2002	648	128	98	Best practice market reference in NL
	2009	50-500	113-188		Run by private enterprises
Economopuolos, 2010	2009	50-500	61-104		Run by municipal associations
EPA, 1993	1987		55-96		WARM model
World Bank, 1999		300	44-75		Configuration typical of SE Asia
	2006	80-115	116-126		
ENEA, 2006		400-600	85-90		Our elaboration assuming economic life = 20 years, r 5%, load factor = 90%
Eunomia and Ecotec, 2002	2001	200	134		
	2001	250		79	
Jamash and Nepal, 2013		250		59	Based on COWI, 2002

Source: our elaboration on the quoted studies

Table 4 – A review of economic studies quantifying external cost of incineration (all values in €$_{2012}$/t)

	CO2		NO2		Other (conventional)		Transport		Energy recovery		Leachate (ash)		Total	
	min	max	min	max	min	max	min	max	min	max	min	max	min	max
Tellus, 1992	-	-	-	-	-	-	-	-	-	-	-	-	0.94	4.69
CSERGE, 1994	1.03	10.06	-	-	1.54	3.10	0.16	1.54	6.46	22.15	-	-	5.41	18.58
Powell and Brisson, 1994	1.03	10.06	-	-	1.74	3.83	0.35	0.53	10.31	14.11	-	-	- 2.96	5.91
Econ, 1995	-	-	-	-	-	-	-	-	-	-	-	-	26.28	160.48
EC, 1996	-	-	-	-	-	-	-	-	-	-	-	-	1.22	1.22
Enosh, 1996	-	-	-	-	-	-	-	-	8.02	-	-	-	9.47	9.47
EMC, 1996	3.66	-	-	-	2.36	-	-	-	8.02	-	-	-	1.55	1.55
Miranda and Hale, 1997	-	-	-	-	-	-	-	-	-	-	-	-	4.85	29.56
Rabl et al, 1998	-	-	-	-	-	-	-	-	-	-	-	-	11.54	11.54
ExternE, 1998	-	-	-	-	-	-	-	-	-	-	-	-	14.08	86.34
EC, 2000	0.47	0.94	-	-	4.69	101.35	-	-	-	107.92	-	-	- 8.45	116.37
Eunomia, 2002	18.44	19.42	0.91	1.58	8.18	21.99	-	-	-	-	0.05	-	27.58	43.03
Dijkgraaf & Vollenbergh, 2003	-	-	16.20	-	-	-	-	-	21.23	-	26.92	-	16.49	16.49
Mean	4.93	10.12	8.55	1.58	3.70	32.57	0.25	1.04	9.01	48.06	13.49	-	8.31	38.86
Median	1.03	10.06	8.55	1.58	2.36	12.91	0.25	1.04	8.02	22.15	13.49	-	5.41	16.49

Source: our elaboration on Eshet et al., 2006