Environment Protection Engineering

Vol. 39 2013 No. 4

DOI: 10.5277/epe130412

TOMASZ STYPKA[1], AGNIESZKA FLAGA-MARYAŃCZYK

COMPARATIVE ANALYSIS OF MUNICIPAL SOLID WASTE SYSTEMS: CRACOW CASE STUDY

The evaluation method to compare municipal solid waste management (MSWM) systems has been presented. The results of the integrated waste management model (IWM-1), were used as the input data for the analysis. The results were integrated into life cycle analysis (LCA) impact categories. The authors present possible to calculate categories, and calculate them for the two MSWM scenarios. Next, the system performance was compared using a multicriteria method, called analytic hierarchy process (AHP). The hierarchical preference analysis on the World Wide Web software (Web-HIPRE) was applied to conduct the analysis. The criteria ratios for the AHP were assumed arbitrarily based on the best knowledge of the authors. Finally, the presented sensitivity analysis showed the confidence of the obtained results and pointed out the most important assumptions of the whole analysis. The two Cracow MSWM systems were used as a case study.

1. INTRODUCTION

There is a need to develop, master and implement a simple but reliable tool that would facilitate decision makers to select the MSWM system. Several mathematical models exist which can help in this task though the main decision variable in such models remains cost. Environmental elements (recycling schemes) appeared in the models in the 1980s [1, 2]. Another group of models included the environmental factors in the form of constrains of the economic models [3]. Some of the models conduct the life cycle analysis (LCA) of the waste disposal system while other only focus on different environmental elements such as noise or traffic [3] or CO_2 emissions from vehicles [4]. The review of various models can be found in literature [5–7] but so far no model meets the expectations of decision makers.

Probably the group of models which, in the best way, reflects the idea of sustainable development is a group of LCA models. The examples of such models are: the

[1]Institute of Thermal Engineering and Air Protection, Cracow University of Technology, ul. Warszawska 24, 31-155 Cracow, Poland, corresponding author T. Stypka, e-mail: stypka@gmail.com

US-EPA [8], integrated waste model IWM [9], MIMES/Waste [10], ORWARE [11], ISWM tool Canada, LCA-IWM [12], and WISARD [9]. IWM was first published in 1995 as an IWM-1 in a form of an Excel spreadsheet. IWM-2 was developed to improve certain aspects of IWM-1 and to include more recent global (rather than solely European) data. IWM-2 was published in SQL-Database which is more friendly for inexperienced users but less useful for in-depth analysis. Because of its transparency, flexibility, simplicity, the authors decided to use the older version of the IWM model.

The decision makers are interested in a clear and simple answer which MSWM system is the best. The IWM-1 model gives plenty of detailed results not useful for the decision makers. To solve this problem, the authors attempted to integrate IWM-1 results using the LCA impact categories and next, using simple but sound multiobjective AHP method, come up with the final answer. This method was applied to compare two MSWM systems. One scenario was based on a real system used in Cracow, Poland in 2001. The main feature of this system is the use of landfill as the main way of waste disposal. The other system is based on the assumption that the waste recycling is improved and the main way of restwaste disposal is incineration. This was the planned system for Cracow at the time.

2. DESCRIPTION OF THE MSWM SYSTEMS BEING COMPARED

The analysis compares the Cracow (pop. 742 000) waste disposal system, operated by the city in 2001 (the system A), with the hypothetical system B. The system B was planned at that time in Cracow, and was based on waste incineration and more intensive recycling program. All information concerning waste and description of the system is taken from Kopacz [14]. The key data for the MSWM analysis are the amount and composition of waste. Waste input data, the same for the system A and the system B, divided into categories required by the IWM-1 model are presented in Table 1.

Table 1

Composition of various wastes in analyzed Cracow case

Amount [t/year]	Household waste composition [wt. %]								
	Paper	Glass	Metal		Plastic		Textiles	Organics	Other
169 346	19.9	7.8	2.9		14.4		6.1	36.2	12.7
			Ferrous	Non-ferrous	Film	Rigid			
			65	35	44	56			
Amount [t/year]	Commercial waste composition [wt. %]								
	Paper	Glass	Metal		Plastic		Textiles	Organics	Other
107 806	45.0	5.0	4.1		12.0		1.0	30.0	2.9
			Ferrous	Non-ferrous	Film	Rigid			
			65	35	44	56			

In the system A, the city has a landfill as the main disposal method and 150 recycling material banks for metal, paper, PET bottles, and glass. Additionally, the city has the system of „bring and earn" collection points and the composting facility with the throughput of 6000 tons per year. The charity organizations run the system of collection points for the textile waste. In the system B, it is assumed commissioning the incinerator with annual throughput 200 000 tons of waste which allows generation of electricity with the 20% efficiency. Contrary to real plans, the waste heat recovery is not modeled, because there is no such option in the IWM-1 model. Additionally, the number of collection banks is increased up to 450, and thanks to the increase of public awareness, the amount of recyclables collected in each bank is increased by 25%. Material recovery facility ready to handle 20 000 tons of recyclables with two composting facilities for 6 000 and 9 000 tons of green waste are commissioned. In some parts of the town, „wet" and „dry" waste collection systems have been implemented.

Table 2

Streams of waste and recovered materials for analyzed scenarios

Characteristics	The system A	The system B
Final solid waste non-hazardous [kt] hazardous [kt]	 246.98 	
Total weight [kt]	0.39	
Total volume [m^3]	247.37	
Material recovery rate [%] average paper glass metal-Fe metal non-Fe plastic – film plastic – rigid textiles	 17 25 6 40 15 4 1 6	 36 54 20 56 21 5 22 11
Organic recovery rate [%]	1	4
Overall recovery rate [%]	10	22
Diversion from landfill [%]	10	80
Secondary materials [%] paper glass metal-Fe metal non-Fe plastic – film plastic – rigid textiles compost total	 20.22 1.12 2.36 0.53 0.92 0.14 0.71 1.02 27.56	 44.09 3.66 3.25 0.74 1.10 3.65 1.25 3.58 61.66

As a result of those changes, the amount and quality of waste disposed at the landfill changes significantly. The exact results for both systems, calculated using the IWM-1 model, presents Table 2. In the system B, the amount of waste disposed at the landfill is reduced by five times (from 247 ktons in the system A to 49 ktons in the system B). Also in the system B, the diversion from landfill is eight times higher than in the system A, and the amount of recyclables is increased more than twice. The economic and environmental impacts of the two systems are also very different.

3. METHOD OF INTEGRATION OF THE RESULTS OF IWM-1

The IWM-1 model gives tables of results estimating the air emission of 22 compounds plus emission of 23 compounds into water. Additionally, the basic statistical and economical data about the system performance is also presented. Such detailed results are not useful for the decision makers and need to be aggregated. The proposed integration method is based on impact assessment of LCA. To calculate these indicators the authors used the method described in detail in other papers [15–19]. The assumption was to use the maximum possible number of categories which could be calculated based on the IWM-1 results. The list of the selected categories is presented in Table 3.

Table 3

Selected categories of the life cycle impact assessment

Impact categories	Characterisation factor	Unit
Baseline categories		
Depletion of abiotic resources	Abiotic depletion potential (ADP)	kg (antimony eq.)
Climate change	Global warming potential (GWP 100)	kg (CO_2 eq.)
Human toxicity	Human toxicity potential (HTP 100)	kg (1.4-dichlorobenzene eq.)
Ecotoxicity: fresh water aquatic ecotoxicity	Freshwater aquatic ecotoxicity potential(FAETP 100)	kg (1.4-dichlorobenzene eq.)
Ecotoxicity: terrestrial ecotoxicity	Terrestrial ecotoxicity potential (TETP 100)	kg (1.4-dichlorobenzene eq.)
Photo-oxidant formation	Photochemical ozone creation potential (POCP)	kg (ethylene eq.)
Acidification	Acidification potential (AP)	kg (SO_2 eq.)
Eutrophication	Eutrophication potential (EP)	kg (PO_4^{3-} eq.)
Stratospheric ozone depletion	Ozone depletion potential (ODP steady state)	kg (CFC-11 eq.)
Land competition	Land use	m^2·year
Other impact categories		
Odour malodorous air	Reciprocal of odour threshold value (1/OTV)	m^3 (air)

Indicators for the various impact categories were selected based on the literature [20]. Unfortunately, not all recommended impact categories can be directly calculated from the IWM-1 table of results.

4. RESULTS OF THE ANALYSES

The IWM-1 energy balance for two analyzed MSWM systems presents Fig. 1. In both scenarios, the collection stage is the most energy consuming. The incinerator is a significant source of energy but in the system A the landfill is also the source of energy due to landfill gas (LFG) utilization. The waste incineration generates about eighteen times more utilized energy than landfill gas powered electricity generators. This is because the LFG production is less efficient due to gas losses from the waste body, and the LFG incineration units utilize only produced electricity while the waste heat is discharged into the environment. In the scenario B, waste incineration utilizes approximately 80% of the generated energy in the form of electricity or waste heat delivered to the municipal heating system. The total energy balance is negative for the system A because the energy generated and utilized at the landfill covers only half of the energy needed at the collection stage. The system B becomes the net energy producer since the incinerator generates far more energy than is needed at the collection stage and at the sorting and composting facilities.

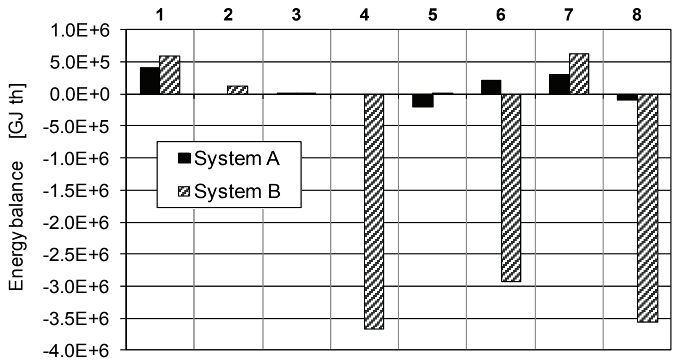

Fig. 1. Energy balance for two analyzed MSWM systems:
1 – collection, 2 – sorting, 3 – biological treatment, 4 – thermal treatment
5 – landfill, 6 – IWM model total, 7 – recycling savings, 8 – overall total

Both scenarios provide significant energy savings thanks to the recycling programs that can cut down on energy consumption at the paper, plastic and glass producing facilities. Because the recycling in the system B is much bigger, it results in higher energy consumption on the collection stage but it also saves more energy than the system A at the paper, plastic, and glass producing facilities. Looking from the LCA perspective taking into consideration the savings of the energy at the raw material production stage, both systems are net energy producers but the total balance in the system B is thirty five times higher.

One of the main reasons for introducing, in the system B, the waste incineration is to prolong the landfill's lifespan. The IWM-1 estimates that, in the system B in comparison with the system A, the amount of waste will drop four times in terms of both volume and mass load.

Abiotic depletion. The analysis shows that, regarding depletion, the system B is far worse that the system A (Fig. 2). During the thermal treatment, various elements and compounds are released into the environment in small portions. Those emissions are not necessarily dangerous to the environment but the elements and compounds are inevitably lost. Mercury emission has the biggest impact on abiotic depletion index. It makes 161% of the index at the combustion stage and 99% of the index calculated for the entire IWM model. This is possible because the total SO_x emission at the thermal stage of waste treatment is calculated as negative because energy produced by the incinerators "saves" the emissions at the regular power plants. Such negative emissions make the total value of the abiotic depletion index lower, and in the same time increase of the relative importance of remaining compounds. The same is observed in the system A, at the landfill site, when burned LFG produces electricity and substitutes SO_x emission from the conventional power plants. Because mercury is not transformed into the LFG, hence it is not released into the environment, the total value of abiotic depletion index at the landfilling stage in the system A remains negative.

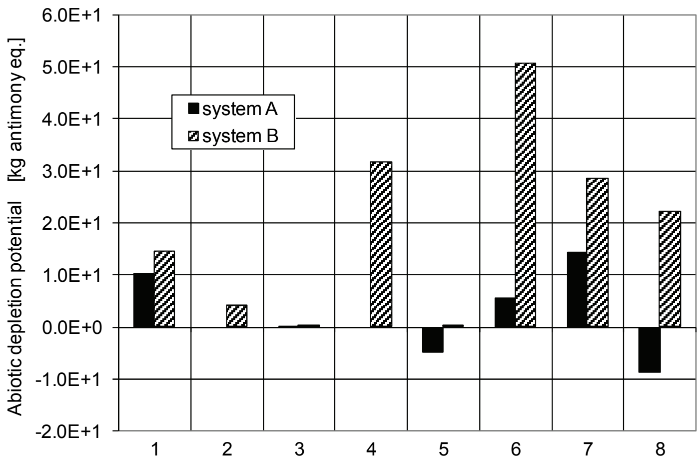

Fig. 2. Abiotic depletion for two Cracow MSWM systems
1 – collection, 2 – sorting, 3 – biological treatment, 4 – thermal treatment
5 – landfill, 6 – IWM model total, 7 – recycling savings, 8 – overall total

Resource savings obtained thanks to the extensive recycling planned in the system B do not change the overall perspective: in terms of abiotic depletion, the system A remains superior to the system B. A significant improvement of this index can be obtained by introducing intensive collection of source separated, household hazardous waste (HHW). This is particularly important now, when an intensive promotion of energy efficient but mercury containing fluorescent lamps takes place. Such lamps have to be collected separately, and transported to the manufactures for mercury recovery.

Global warming – climate change (Fig. 3). Incineration, landfilling and collection are three stages of waste disposal with the strongest impact on the climate change. Landfilling remains by far the most damaging one. Its impact in the system A is ten times bigger than the impact of the thermal treatment stage in the system B. The impact of the collection in both systems remains similar comparable with the impact of the thermal treatment. Incineration and landfilling cause methane and carbon dioxide generation. Specially methane (generated only in the landfill) is 21 more powerful than carbon dioxide and needs special attention. It is assumed that 40% of generated methane is captured and utilized in the CHP units. The remaining 60% of the LFG is released directly into atmosphere increasing the global warming impact of the landfill. The total impact of the landfill on the climate change is 920% higher than incinerator's. Recycling in both scenarios generate the minimal profits for the global environment and therefore globally, the system A has more deteriorating impact on the global climate than the system B.

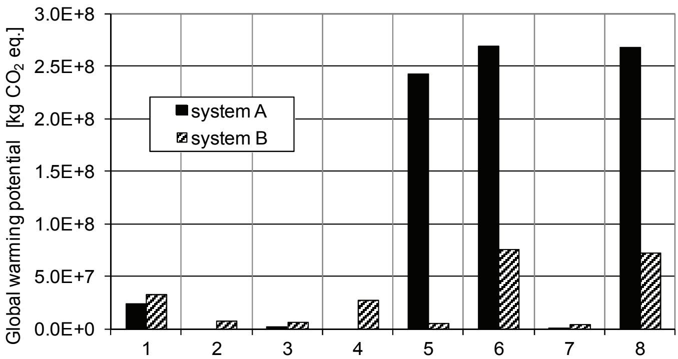

Fig. 3. Climate change for two Cracow MSWM systems
1 – collection, 2 – sorting, 3 – biological treatment, 4 – thermal treatment
5 – landfill, 6 – IWM model total, 7 – recycling savings, 8 – overall total

Recycling has a small impact on the global warming index because the borders of the analyzed system are drawn in such a way that although the energy savings at the collection stage are calculated by the model, the emissions associated with these savings are already beyond the scope of the analysis. As a result, the total impact of the two analyzed scenarios on the climate change, conducted from the local and LCA perspective are almost the same: the system B is better than the system A. The difference between the systems measured in carbon dioxide equivalent emission is 1.96×10^8 kg per year.

Human toxicity. This index covers the impact of the toxic substances present in the natural environment on the human health. Figure 4 clearly indicates that the system B has a negative impact on the human health. A detailed analysis of the results confirms that although all stages of the waste disposal system have a negative impact on human body, the impact of incineration is overwhelming. The impact of all stages of waste disposal equals 3.49×10^8 kg of 1.4-DCB eq. and the impact of the incineration is 3.48×10^8 kg of 1.4-DCB eq.

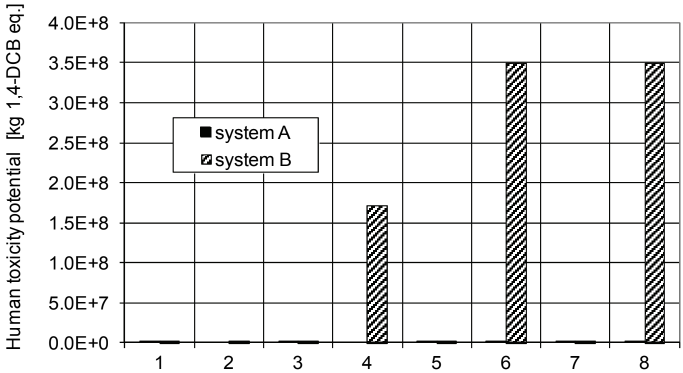

Fig. 4. Human toxicity potential of two Cracow MSWM systems
1 – collection, 2 – sorting, 3 – biological treatment, 4 – thermal treatment
5 – landfill, 6 – IWM model total, 7 – recycling savings, 8 – overall total

The impact of the landfilling depends on the collection stage. The main reason for such negative impact of the incineration is not the emissions of commonly feared dioxins and furans but rather of arsenic and chromium compounds. According to the IWM-1 model arsenic emission is a product of waste incineration. From each ton of burned waste 2.5 g of arsenic is discharged into atmosphere. This results in emission of 507 kg of arsenic per year which is equivalent to emission of 1.8×10^8 kg of 1.4-DCB. The other most toxic for humans emission is chromium. Chromium is released into environment with the landfill gas, landfill gas incineration products, flue gases from the incinerator and leacheate from the landfill. Based on the literature and unpublished reports about the chemical composition of the leacheate from Cracow landfill [21], the authors assumed that emission of chromium calculated by the IWM-1 model takes place mostly in the form of Cr(III) (97%). More toxic Cr(VI) makes only 3% of the total chromium emission. IWM-1 model estimates that each ton of incinerated waste results in 6.3 g of chromium emission and that leads to 1276 kg of chromium emission into the air per year. The toxicity of this emission is equal to emission of 1.3×10^8 kg of 1.4-DCB. Both emissions of chromium and arsenic constitute 89% of the entire human toxicity index.

Recycling reduces the toxicity of waste disposal to humans. In the system A, thanks to recycling, the total human toxicity of the entire system is reduced from 6.05×10^5 to 3.67×10^5 kg of 1.4-DCB eq. In the system B, thanks to more advanced recycling, the reduction is almost twice as big but the negative impact of incineration is such immense that this progress is not noticeable.

Freshwater aquatic ecotoxicity. Both analyzed systems show the negative effect at all stages of waste disposal but the impact of thermal treatment stage is the strongest (Fig. 5). Thorough analysis shows that emissions to air have bigger impact on total FAET value than emissions to water. In the system B hydrogen fluoride (HF) emission to air and phenol emission to water are the two most important contributors together making 65% of the whole FAET index. In the system A, by far the most dangerous is emission of adsorbable organohalogens (AOX – absorbable chlorinated organics; the equivalent amount of chlorine, bromine and iodine contained in organic compounds in water or wastewater, expressed as chloride) coming from landfill's leachate into ground water.

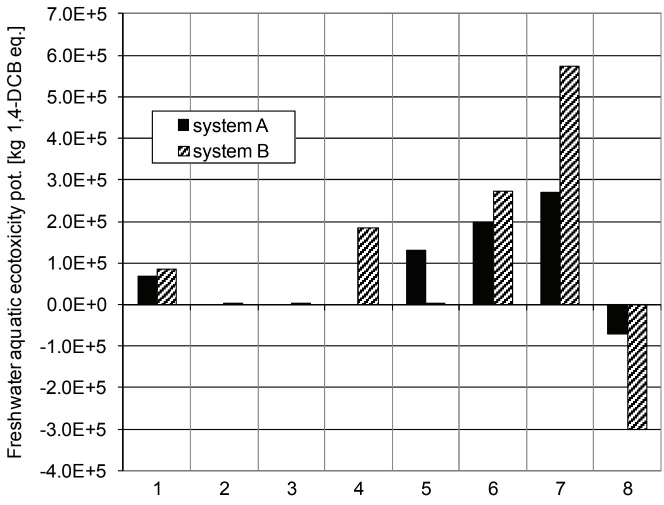

Fig. 5. Freshwater aquatic ecotoxicity of two Cracow MSWM systems
1 – collection, 2 – sorting, 3 – biological treatment, 4 – thermal treatment
5 – landfill, 6 – IWM model total, 7 – recycling savings, 8 – overall total

Phenol emissions observed in the collection stage of both analyzed systems has a negative impact as well. Phenol is emitted into atmosphere during the production of the diesel oil needed for the waste transporting trucks. Landfills affect the aquatic environment because of the leachate. In both analyzed systems at the landfill stage, leachate makes 99% of the FAET while the air emissions of the landfill gas make only 1% of the index value. The most toxic compound in leachete is AOX.

IWM-1 model assumes that 70% of the leachate is collected, and the rest is released directly into the water environment. It is estimated that in the system A, the directly released leachate contaminates water environment with 22 kg of AOX per year. AOX makes 89% of FEAT even the FEATP for AOX is very much comparable with mercury, cadmium, nickel, and copper. The final disposal of ashes from the waste incineration does not have any significant impact on the water environment.

Recycling and its development has a significant impact on the value of the FEAT. The direct emissions into the water makes 99.8% of the total value of the indicator, with AOX playing the main role. The source of these savings is lower emission of AOX obtained thanks to the reduction of paper production from the virgin material. It is estimated that thanks to more advanced recycling programs in the system B, the production of paper from the virgin material would drop by 44 090 Mg reducing AOX emission from the paper mills by 110 kg. Because the value of $FAETP_{AOX}$ is high (5.2×10^3 kg of 1.4 dichlorobenzene eq.) such drop in paper production equals the reduction of 5.73×10^5 kg of 1.4 dichlorobenzene eq. emission into the aquatic environment. This makes almost 100% of the index value. Phenol is the second important pollutant responsible for the aquatic toxicity. The recycling in the system B reduces the phenol emission by 1.17×10^2 kg of 1.4 dichlorobenzene eq., and that makes only 0.02% of the whole index.

Generally, a well organized system of waste management has a significantly positive impact on the water quality. In both analyzed systems, the positive impact of recycling fully offsets the entire waste disposal system and in the system B, positive impact of recycling is twice as big as negative impact of all stages of waste disposal. In the system B, incineration stage has more negative impact on water environment than has landfilling in the system A but more elaborate recycling programs in the system B fully compensate this negative impact. Increasing the efficiency of the leachate recovery has also a positive effect on the aquatic environment, if measured by the FAET index.

Terrestrial ecotoxicity. The impact on soil at various stages of a waste disposal system is very similar to the one observed for the human toxicity impact index (Fig. 6, cf. Fig. 4.). Both systems have negative impact on soil but on different scale. Total value of TET index for the system A equals 8.56×10^0 and for the system B 3.65×10^5 kg of 1.4 dichlorobenzene eq. The system B causes more than 10 000 times bigger damage to the soil than the system A. In practical terms, this means that the landfilling has an insignificantly small impact on land compared with the incineration.

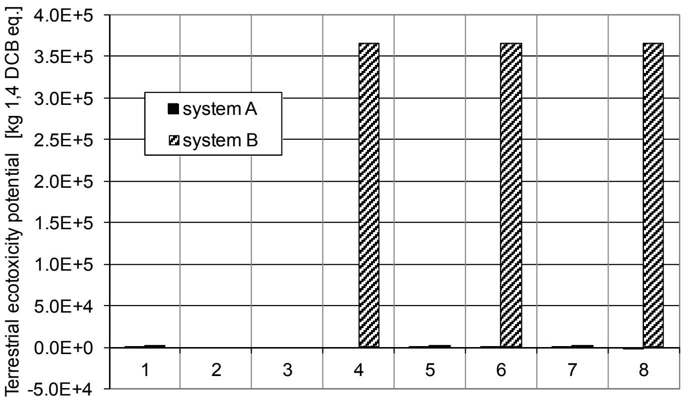

Fig. 6. Terrestrial ecotoxicity of the two Cracow MSWM systems
1 – collection, 2 – sorting, 3 – biological treatment, 4 – thermal treatment
5 – landfill, 6 – IWM model total, 7 – recycling savings, 8 – overall total

The incinerator affects the soil by flue gases and heavy metals, in particular. The IWM-1 model estimates that every year the incinerator emits 101 kg of mercury (Hg), 1280 kg of chromium (Cr), 507 kg of arsenic (As) and 507 kg of nickel (Ni) to the air. All these emissions have the negative impact on the soil but mercury contributes 89% into the terrestrial ecotoxicity index (TET). Chromium contributes to this index only 7% while arsenic 2% and nickel only 1%.

Recycling has a very positive impact, if measured by TET but in the system B it is absolutely not able to offset the negative impact of the thermal stage. In the system A, the positive impact of recycling measured by the TET index is 20 times larger than the negative impact of all stages of waste disposal. In the system B, this positive impact of recycling is twice as big as in the system A but 1000 times smaller that the total impact of all stages of waste disposal measured by the TET. The positive impact of recycling is caused by avoided emissions of mercury. In the system A, thanks to paper recycling, the mercury emission is reduced by 0.0623 kg of mercury/year (equivalent to 197 kg of 1.4 dichlorobenzene), which is twenty times higher than the emission from all stages of waste disposal in the system A. Improved recycling has a direct impact on the TET index.

Photochemical smog. The system A has a negative impact on environment if measured by the photochemical ozone creation, while the system B has a positive impact (Fig. 7). The smog forming compounds are created at the level of waste collection, biological treatment and landfilling. Smog at the collection stage is created by such emissions as hydrocarbons, nitrogen oxides, sulfur oxides, and carbon monoxide. These compounds are by-products of diesel oil combustion and plastic waste containers production. At the landfill site smog is generated as a result of emission of the landfill gas, methane in particular. On the other hand, electricity generated from the

LFG replaces the one generated in conventional power plants, which results in reduction of sulfur oxides, nitrogen oxides and reduction of the POC index of the landfilling stage. In the final balance of these two processes, the landfill generates 55 300 kg of ethylene eq. The energy utilized from the incinerator is much higher, resulting in larger avoided emission, which makes the POC at the incineration stage even negative (-1.82×10^5 kg of ethylene eq.)

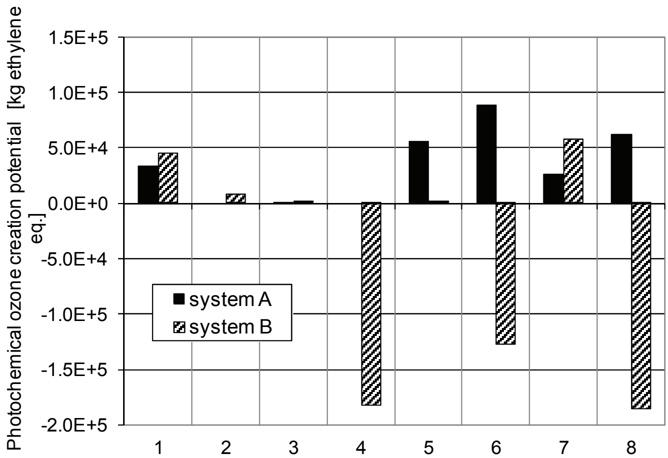

Fig. 7. Photochemical smog creation of the two Cracow MSWM systems
1 – collection, 2 – sorting, 3 – biological treatment, 4 – thermal treatment
5 – landfill, 6 – IWM model total, 7 – recycling savings, 8 – overall total

Development of the recycling stage also reduces the emission of the smog creating compounds. In the system B, with a far more advanced recycling system, the emission of the smog creating substances is reduced almost by half. Concluding, in terms of LCA, and from the local perspective, the system B reduces the total smog creation while the system A has a negative impact on air quality.

Acidification. In the analyzed case, it was assumed that the acid deposits will affect the city of Cracow, which is sensitive to such impacts as a place of a very high material and cultural value (lime stone historic buildings, steel constructions). For this reason the value of the AP was not reduced by any reduction coefficients. The obtained results are presented in Fig. 8.

The main stages of waste disposal which have impact on environment acidification are collection, thermal treatment and landfilling. Acidification is caused by diesel oil burning during the waste collection, and from the emissions during the waste incineration. Both, the incinerator and the landfill equipped with the LFG extraction system used to produce electricity are the sinks for the acidic emissions, because LFG powered generation unit discharges, and the incinerator, emit far less acidic emissions than are generated in the conventional power plants.

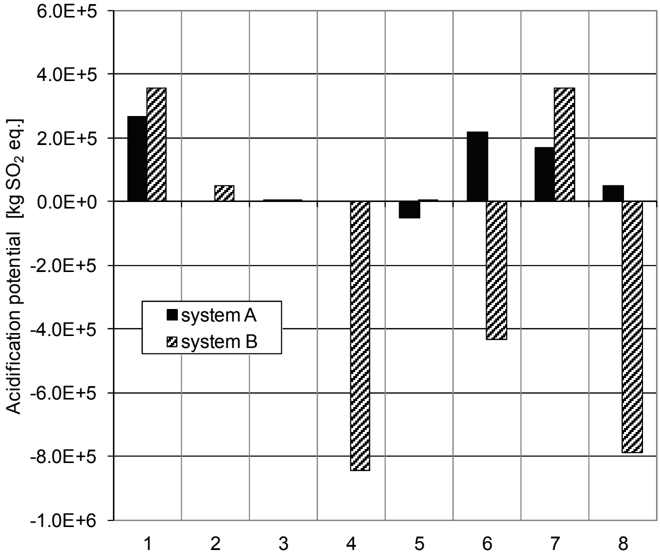

Fig. 8. Acidification in two Cracow MSWM systems
1 – collection, 2 – sorting, 3 – biological treatment, 4 – thermal treatment
5 – landfill, 6 – IWM model total, 7 – recycling savings, 8 – overall total

78% of the acidification index at the thermal stage originates from the negative emissions of sulfur dioxide. The observed “savings” at the recycling stage are mainly caused by the reduction of the energy demand in the production of plastic, paper, and metal from the recycling materials. This reduction includes the energy consumption needed for the transportation of the recycling material to the processing facilities. The development of the recycling program in the system B results in bigger “savings” during the recycling stage, and makes the system B more superior to the system A in terms of acidification, both from local and LCA perspective.

Eutrophication. Waste collection is the main stage where compounds stimulating eutrophication are generated. Eutrophication is caused by the emission of nitrogen oxides generated during the diesel oil combustion in trucks, collecting both recyclables and mixed waste. The IWM-1 model estimates that in the system A collection stage generates 2.57×10^5 kg of nitrogen oxides per year; it is 97% of the total value of the eutrophication index (Fig. 9).

Recycling reduces the total value of eutrophication both into the air and water. The positive effects come from different emissions, which occur at the production phase where either virgin or recyclable materials may be used. The emissions from the transportation of the recyclables into processing plant are also included in the calculations. The expected savings are significant. In the system B, thermal treatment is a significant sink for eutrophication stimulating compounds. The avoided emission is both to air, and to water but the "savings" into the air are 344 times higher. The reduction of the eutrophication effect is due to the reduction of NO_x and ammonia, with NO_x emissions being the most important. Calculated negative emission during the thermal treatment fully offsets the eutrophication emissions occurring during all other stages in the system B making the system B neutral regarding euthrophication. The advanced recycling program makes the system B friendly towards the environment from that perspective. In the system A the less efficient recycling does not totally offsets the collection, and the total impact of the system A is slightly negative.

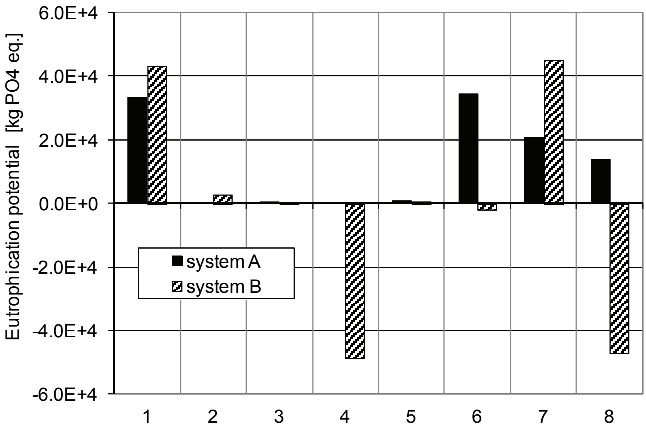

Fig. 9. Eutrophication in two Cracow MSWM systems
1 – collection, 2 – sorting, 3 – biological treatment, 4 – thermal treatment
5 – landfill, 6 – IWM model total, 7 – recycling savings, 8 – overall total

Odour. The IWM-1 estimates only three emissions of malodorous substances (Fig. 10). Nevertheless the obtained results confirm the widespread expectations that the incineration reduces the odour problem. In the system A, the landfill is the main source of odour. If the model calculated all the aromatic substances emitted to the air with the LFG the difference between the systems would be even sharper. Waste incineration reduces approximately 100 times the odour nuisance. This is mainly caused by the reduced emission of hydrogen sulfide (H_2S).

Recycling generates some avoided odour emissions mainly thanks to the reduction of hydrogen sulfide emissions during the paper production. In the system B, the avoided emission is larger than he odour emissions which occur during all stages of waste disposal. The problem with such compensation approach is that odour is a local problem and for the people living near one facility the potential reduction of odour at the other location is no an argument. Generally, the waste incineration, in comparison with the system A based on the landfilling, reduces the odour 45 times, when taking into account avoided emissions during the recycling the outcome is even more favourable for the system B.

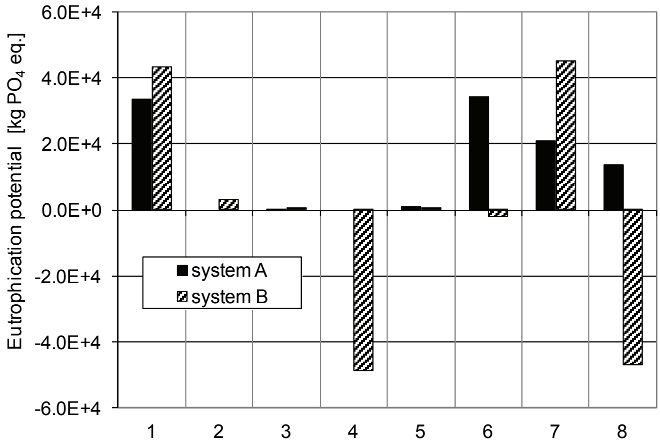

Fig. 10. Odour emissions from the two Cracow MSWM systems
1 – collection, 2 – sorting, 3 – biological treatment, 4 – thermal treatment
5 – landfill, 6 – IWM model total, 7 – recycling savings, 8 – overall total

The obtained results still do not give an instant answer about the superiority of one analyzed system over the other one. The final evaluation of the analyzed scenarios was made using the AHP method.

5. MULTICRITERIA ANALYSIS OF THE CRACOW MSWM SYSTEMS

The authors used the analytic hierarchy process (AHP) as a method for further analysis with the software offered on the website of the Hesinki University of Technology [22]. Prepared hierarchy of goals and assigned ratings are presented in Fig. 11. The ratings were assigned by the authors.

Figure 12 presents the final results of the analysis. The figure shows that the system B thanks to significantly better environmental performance is better evaluated than the system A. The overall score of the system A is 0.362 and that of the the system B is 0.638.

More detailed comparison of the environmental performance of the two MSW systems presents Fig. 13. The system B is more friendly towards all three bodies of the environment: water, soil and air. The system B is superior to the system A in all subcategories of air and water criteria. In subcategories of "soil protection", the system B is better only in "land use" subcriterium but this subcriterium is so important that the total evaluation of the system B in category of "land use" is better than the performance of the system A.

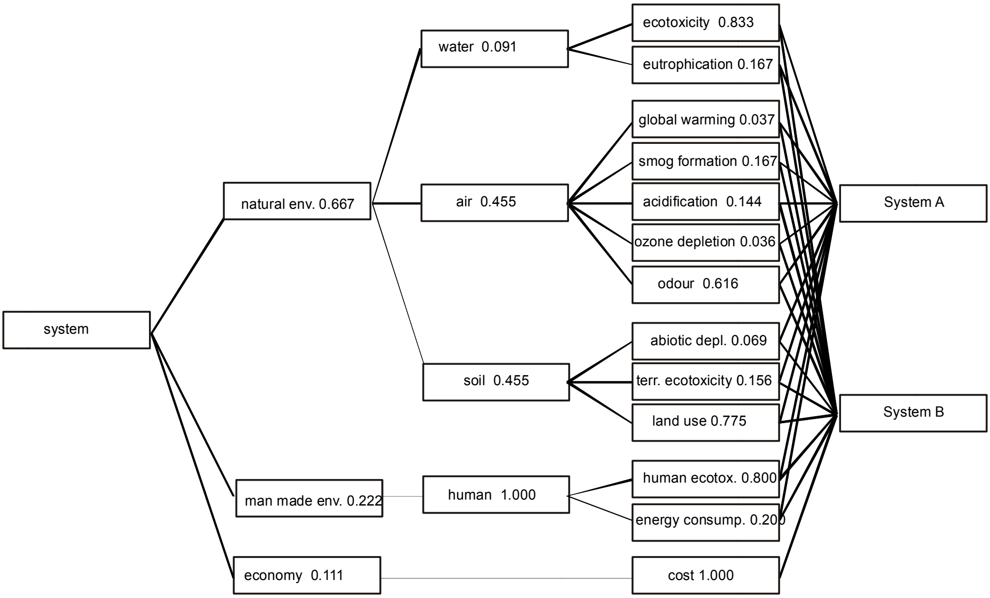

Fig. 11. Objective hierarchy and ratings for the Cracow analysis.
The ratings were assigned by the authors

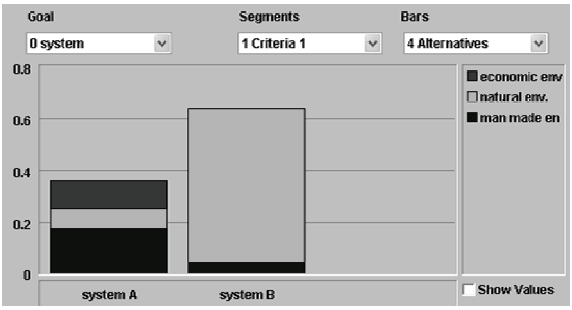

Fig. 12. Results of the AHP analysis for the two Cracow MSWM scenarios. Criteria 1

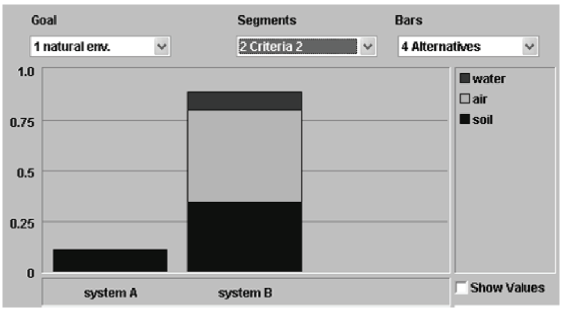

Fig. 13. Results of the AHP analysis for the two Cracow MSWM scenarios. Criteria 2

6. SENSITIVITY ANALYSIS

The final outcome of the AHP analysis depends on the assumed hierarchy of goals and on the assigned relative weights of the goals. The performance of the analyzed system has far more limited impact on the final result. If two options are compared, it is only important if the performance in each category is superior but not the level of this superiority.

The reason why the system B performs better than the system A is its significantly better performance in the category „impact on the natural environment". The relative weight of this category was assumed to be equal 0.67 but the sensitivity analysis indicates that if this weight is 0.48 both analyzed scenarios have equal value. If the weight for the natural environment equals 0.48, the other two weights should have the values 0.347 for the impact on the manmade environment and 0.173 for the economic performance. Also, the analysis shows that, if the importance of the economic criterion increases from the present ratio 0.11 to 0.30 this will result making the two analyzed systems equally good. The increased rating of the "manmade environment" from the present 0.22 to 0.47 will result in equalizing of the two systems performance. It is up to the decision makers to decide if such distribution of weights is possible.

Changing the ratings of all environmental bodies (impact on water, air and soil) will change the final evaluation score but will not change the rating of the systems' impact on the natural environment – for all ratings impact on natural environment of the system B is better than impact of the system A.

No change of water or air criteria ratings can change the two systems' performance in the subcategory "impact on water" and "impact on air". In category "impact on soil" changing the ratings of all subcategories can result in the switch of superiority of the two systems in the category "impact on soil" but the changes have to be substantial.

Generally, the sensitivity analysis shows that the biggest impact on the final score have the weights assigned at the top level of hierarchy (natural environment, manmade environment, economic impact) and the weights assigned to categories "impact on soil".

7. CONCLUSIONS

The presented evaluation method proved useful in decision process. There is no objective criteria which can help in the process of weights assignment but the presented AHP method can make the whole process more objective, and gives opportunity to present the points of view of different social groups. This transparent evaluation method gives the decision makers a chance to present, not only their preferences concerning criteria but also the decision makers are able to express their trust in the results of LCA. The analysis shows that upgraded system of waste disposal in Cracow (the system B) is a better solution than the old system (the system A). The system with the incinerator and extensive recycling performs significantly better taking into account its impact on the natural environment, while the system A is cheaper and puts less stress on the manmade environment. The system B performs better than the system A on the natural environment in all categories "impact on air" and "impact on soil" and "impact on water". The conducted sensitivity analysis shows the critical elements of the whole analytical process. Such elements are the relative weights assigned at the top level of goals hierarchy, and weights assigned within subcategory „impact on the natural environment". The decision makers should make an extra effort to increase the certainty in assigning these values, because different values can easily change final outcome of the analysis.

REFERENCES

[1] CLAPHAM W.B., Recyclace: *A computerized planning tool to improve municipal solid waste management*, Ohio J. Sci., 1986, 86 (4), 189.

[2] JENKINS L., *Parametric mixed integer programming: An application to solid waste management*, Manage. Sci., 1982, 28 (11), 1270.

[3] CHANG N.B., SHOEMAKER C.A., SCHULER R.E., *Solid waste management the system Analysis with air pollution and leachate impact limitations*, Waste Manage. Res., 1996, 14, 463.

[4] WANG F.S., RICHARDSON A.J., RODDICK F.A., *SWIM – a computer model for solid waste integrated managemen*t, Comput. Environ. Urban, 1988, 20 (4/5), 233.

[5] BJORKLUND A., *Environmental systems analysis waste management with emphasis on substance flow and environmental impact*, Licentiate Thesis, Stockholm University of Technology, Stockholm 1998.

[6] MACDONALD M.L., *Solid waste management models: a state of the art review*, J. Solid Waste Technol. Manage., 1996, 23 (2), 73.

[7] MORRISSEY A.J., BROWNE J., *Waste management models and their application to sustainable waste management*, Waste Manage., 2004, 24, 297.

[8] BARLAZ M.A., RANJITHAN R., WEITZ K.A., NISHTALA S.R., *Life-Cycle Study of Municipal Solid Waste Management, System Description*, US Environmental Protection Agency, USA, 1995.

[9] WHITE P.R., FRANKE M., HINDLE P., *Integrated Solid Waste Management. A Lifecycle Inventory*, Blackie Academic & Professional, 1997.

[10] SUNDBERG J., *Municipal solid waste management with the MIMES/waste model. A complementary approach to LCA studies for evaluating waste management options*, Workshop on LCA and Treatment of Solid Waste, Stockholm, 28–29 Sept. 1995, 252.

[11] ERIKSSON O., FROSTELL B., BJORKLUND A., ASSEFA G., SUNDQVIST J.O., GRANATH J., CARLSSON M., BAKY A., THYSELIUS L., *ORWARE-a simulation tool for waste management*, Resour. Conserv. Recycl., 2002, 36, 287.

[12] DEN BOER J., DEN BOER E., JAGER J., *Wider application options of the LCA-IWM waste management the system Assessment tool. Assessment of the environmental impact of waste minimization*, Efficient Management of Solid Waste, 7th Waste Forum 2007, Kalisz, 465 (in Polish).

[13] MCDOUGALL F. WHITE P., FRANKE M., HINDLE P., *Integrated Solid Waste Management. A Life Cycle Inventory*, Blackwell Publishing, 2003.

[14] KOPACZ K., *Analysis of integrated municipal solid waste management at Cracow*, Master thesis, Cracow University of Technology, Cracow 2003 (in Polish).
[15] STYPKA T., *Adopting the integrated waste management model (IWM-1) into the decision process. Integration and Optimization of Urban Sanitation Systems*, Proc. of Polish-Swedish Seminar, 2005, Cracow.
[16] STYPKA T., FLAGA-MARYAŃCZYK A., *Developing the evaluation criteria for the municipal solid waste systems. Cracow case study*, Chem. Chem. Technol., Lviv, Ukraine, 2010, Vol. 4.
[17] STYPKA T., FLAGA A., *Application of the integrated waste management model (IWM-1) into the decision process. Integration and optimization of urban sanitation systems*, Proc. of Polish-Swedish-Ukrainian Seminar, Lviv 2006.
[18] STYPKA T., FLAGA A., *Implementation of computer modelling in the waste management process*, *Efficient management of solid waste*, 7th Waste Forum, Kalisz, 2007, 435, (in Polish).
[19] STYPKA T., FLAGA A., *Integrating the results of the IWM-model using the LCA procedures*. *Cracow case study*, 7th International Waste Forum, Poznań, 2009 (in Polish).
[20] GUINÉE J.B., *Handbook on Life Cycle Assessment. Operational Guide to the ISO Standards*, Kluwer Academic Publishers, London 2002.
[21] LINAKA P.W., JEFFREY V., RYANA J.V., WENDT J.O.L., *Formation and destruction of hexavalent chromium in a laboratory swirl flame incinerator*, Combust. Sci. Technol., 1996, 116–117 (1–6).
[22] HIPRE software, <http://www.hipre.hut.fi>, Global Decision Support.