APPLICATION OF THE INTEGRATED WASTE MANAGEMENT MODEL (IWM-1) INTO THE DECISION PROCESS

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EXECUTIVE SUMMARY

The paper discusses the two methods of implementing the Integrated Municipal Waste (IWM-1) model's results for the decision process. The first method relies on the integration of the calculated emissions based on the Polish emission fees and its toxicity ratios. As a result, the environmental impacts on water and on air are presented in monetary units. Such indicators can easily be further compared and combined with the economic data of the MSWM systems. The second presented method applies the procedures of the Life Cycle Impact Assessment (LCIA) and Interpretation. The results of inventory analysis are transformed after characterisation into the environmental profile. This profile consists of the calculated indicators for the chosen impact categories plus the economic performance and solid waste reduction indicators calculated directly by the IWM-1 model.

The authors applied the first version of the IWM model to analyze the present and the planned waste management systems in the two cities: Krakow, Poland and Stockholm, Sweden. The obtained results were integrated by the both discussed methods.

The results of the first method of integration show that the present cost of disposal of one kilogram of waste in Krakow is half the cost in Stockholm, but if the system in Krakow is modified, and the incinerator is built, the costs in the two cities will be nearly equal. Energy consumption by the systems in the two cities is negative. Because Stockholm has a very advanced recycling system and has an incinerator in which the energy recovery rate is higher than at the landfill site the total energy recovered from each kilogram of waste generated is nine times higher in Stockholm than in Krakow.

The second method of results' integration uses the LCIA. This approach does not lead to such direct comparisons of the analyzed systems as the first method, but gives a lot of information which is also important and can be used by the decision maker. The method requires some indirect transformation of the results from the IWM result table and the practical implementation of this transformation is presented in the article. In the analyzed cases the Krakow (future stage) MSW disposal system definitely results in higher impact on abiotic resources, (ADP), on global warming (GWP 100), on smog creation (POCP), on acid rain (AP) and on eutrophication process (EP). On the other hand, the Stockholm MSW disposal system puts more toxic stress on humans (HTP 100), on land (TETP 100), on ozone layer (ODP), on reciprocal of odour threshold value (1/OTV) and more stress on land use. The reasons for that are the higher amounts of waste generated (and incinerated) in Stockholm plus higher than in Krakow recycling rates. Also the use of biological

methods of waste disposal in Krakow results in high values of “biological” indicators. LCIA also shows that the impact on human health (HTP 100) is relatively the most important while the impact on ozone layer and on abiotic resources is relatively the smallest.

INTRODUCTION

The decisions in the area of municipal solid waste management are not only very capital intensive, but also difficult from environmental and social points of view. There is a need to develop, master and implement, a simple, but reliable tool that will help the decision makers in the analytical process. There are several mathematical models applied to help the decision makers in their tasks. In such models the main decision variable is cost. The environmental elements (the recycling schemes) started to appear in the models beginning in the 1980s (Jenkins, 1982; Clapham, 1986). Other group of models included the environmental factors in the form of constrains of the economic models (Chang at al, 1996). 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 (Chang at al, 1996) or on CO_2 emissions from vehicles (Wang at all, 1988).

The uncertainty of the parameters is also an important criteria for dividing models into different categories. Deterministic models such as linear programming (LP), mixed-integer programming (MIP), dynamic programming (DP) and multi-objective programming are used to analyze the problems where there is an assumption of the parameters’ certainty. To reflect the uncertainty, the models use the probability theory as well as the fuzzy and grey system theory.

Classifying the models from the decision areas, the models can estimate waste generation predictions as well as facility sites selection and facility capacity expansion or operation. Similarly, other groups include models which determine vehicle routing, manpower assignment, over-all system operation, system scheduling, waste flow, environmental performance or technology selection (Bjorklund, 1998).

A separate group of computer models apply the concept of Life Cycle Analysis (LCA). The example of such models are: the US-EPA (Barlaz at al, 1995), Integrated Waste Model IWM (White at al, 1997), MIMES/Waste (Sundberg, 1995), ORWARE (Eriksson at al, 2002), ISWM tool Canada, and WISARD (McDougall at al, 2001). These models are readily available applications but, with the exception of the IWM model, in practice, most models are still in the development or upgrading stages. The ORWARE and MIMES/Waste models are very difficult to be used because of their platform and complexity. Therefore, the potential user is left with the IWM models. At present there are two versions of the model IWM-1 and IWM-2 (McDougall at al, 2001). The two versions differ not only with the applied platform (IWM-1 is an Excel spreadsheet while the IWM-2 is a stand-alone program), but also the IWM-2 produces more accurate data and a more elaborate thermal treatment section. The choice of the platform results in the level of transparency of the two IWM models. IWM-1 is a transparent model and the experienced user can temper it with the coefficients, adjusting them to the local conditions, while the IWM-2 works in a closed environment. The lack of transparency inherent in the IWM-2 was the reason for using an IWM-1 in the presented project. The results of the IWM models are very fragmented hence not useful for the decision makers. Two methods of the results integration are presented. Both methods were applied to compare the MSWM systems in Krakow and Stockholm. Krakow develops its system from the traditional one to one very similar to the system used in Stockholm. What is called a present Krakow system is a system existed in the city in 2001. “Future system” is the system designed by the city planners now at the stage of the legal battle. The paper compares the present and future systems in Krakow, assuming the same waste stream, with the present Stockholm system.

DESCRIPTION OF THE SOLID WASTE MANAGEMENT SYSTEMS IN KRAKOW AND STOCKHOLM

Krakow, the former capitol of Poland, located in Southern Poland occupies 327 km^2 and has approximately 750 000 citizens. Some of them are permanent residents while a significant share are temporary residents such as tourists or students. The central part of the town is medieval and densely populated, but 52% of the town are green areas.

The waste disposal system in Krakow is a traditional one. The city has a recycling program with 150 recycling banks located around the town. They are prepared to collect metal, paper, PET bottles, and glass. Additionally, there are the “bring and earn” recycling centers where one can bring recyclables and collect money. These centers are mainly used by scavengers and by industry located within the city limits. The composting facility (6 000 tons) processes the green waste separately collected in the city. This is the waste from green areas, from the open markets and from the food and tobacco industry located in the city. The last share of waste has to be excluded from the analysis because this waste, according to Polish law, is not a MSW. Textile waste is separately collected by the charity organizations.

In the future, Krakow plans to build an incinerator (200 000 tons) and develop more extensive recycling and recovery programs. The number of recycling banks is planned to reach 450 and also the new material recovery facility (20 000 tons) and a new composting plants are planned for construction (6 000 and 9 000 tons). Also, the separate collection of wet and dry wastes in part of the city is planned for the future.

The subject of the analysis is the city of Stockholm, (Stockholm commune) the capitol of Sweden, which has 406 072 households and approximately 755 000 inhabitants. The medieval old town and friendly green areas attract many tourists which generate waste equal to 5 700 permanent residents (Bokota, 2004). A higher standard of living results in 1.8 citizens per household.

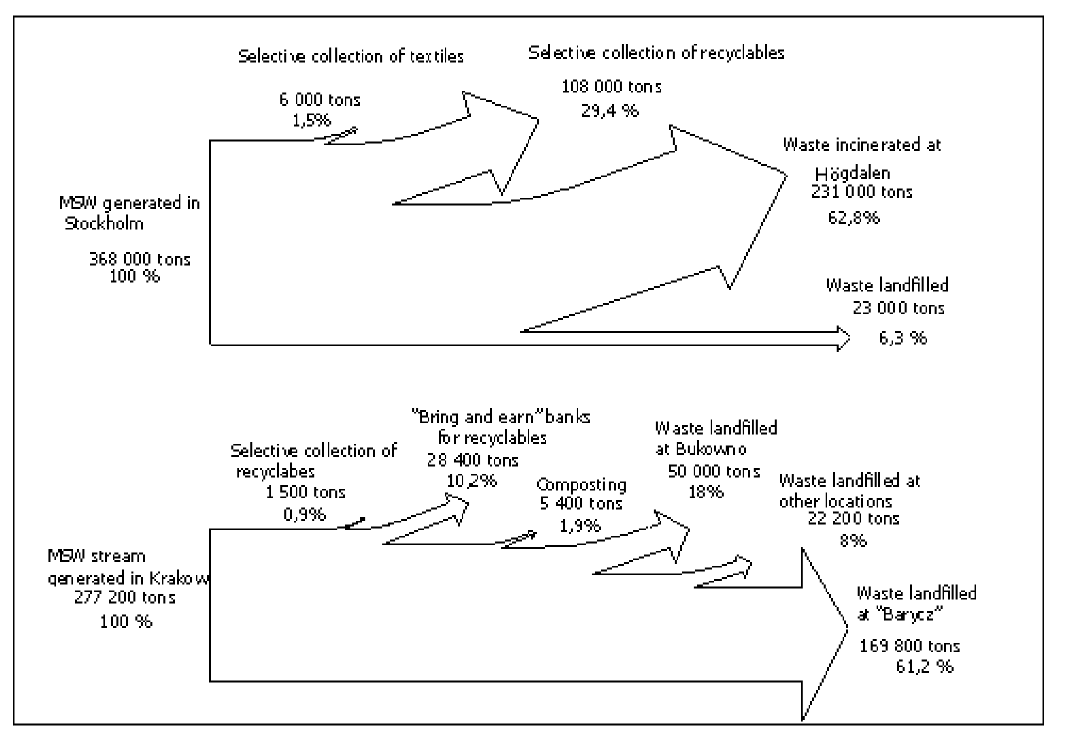

Fig. 1 MSW streams in Krakow and Stockholm (Bokota, 2004)

The waste disposal system is far more technically advanced and developed than in Krakow. The core of the system is the mass burn incinerator at Högdalen. The recyclables can be collected at kerbside plus in 300 collection banks or in the three recycling centers. There are also 22 household hazardous waste collection stations, and small composting and anaerobic digestion plants. These facilities because of their insignificant sizes and character have a negligible impact on the waste disposal scheme and therefore were not included in the model. Landfilling is seen as the last resource and used only for 6% of the waste stream. The waste stream flow in the two towns presents Fig.1

RESULTS OF IWM-1 MODEL

The IWM model presents the results in table form where number of emissions into air and water are presented. Also the origin of these emissions is indicated in the table. A few integrated indicators of the waste disposal system are also presented. Table 1 presents the results of the comparison for economic cost, energy consumption and landfilled to generated waste ratio. The unit cost of waste disposal in Krakow is half the cost in Stockholm. The main component leading to such high cost is the collection stage. A higher standard of living in Sweden accounts for the discrepancies in prices and wages between the two cities. For example, the cost of equipment and labor is higher in Stockholm, directly increasing the cost of the collection stage. Also the unit cost of waste landfilling is high in Stockholm in relation to Krakow. The table fails to indicate the actual unit cost of landfilling, presenting instead the ratio of waste generated, not landfilled waste. In Stockholm, thanks to the incineration, only a small portion of generated waste is being landfilled (6.3%) while in Krakow, the share of landfilled waste is much higher, at 87.2%.

Table 1. esults of the Krakow 2001 and Stockholm MSW systems analysis with the application of IWM-1 model

Stage of the process	Economic unit cost of waste disposal [€/kg waste gen.]		Energy unit consumption [GJ/kg waste gen.]		Ratio of landfilled waste to waste generated [kg/kg of waste gen]	
	Stockholm	Krakow	Stockholm	Krakow	Stockholm	Krakow
Collection	0,068	0,0318	0,00043	0,00056	-	-
Composting	-	0,001	-	0,000006	-	-
Incineration	0,020	-	-0,0084	-	-	-
Landfilling	0,028	0,0277	-0,00006	-0,00075	0,2516	0,8925
Recycyling	0,012	0,003	-0,00183	-0,00110	-0,0096	-0,0077
Total	**0,128**	**0,0635**	**-0,00986**	**-0,00129**	**0,2420**	**0,8848**

Energy consumption by the systems in the two cities is negative. Meaning the energy recovered at the landfill site in the form of the landfill gas or in the incinerator plus the energy saved thanks to the recycling program is larger than the energy needed for processing the waste. Because Stockholm has a very advanced recycling system and has an incinerator in which the energy recovery rate is higher than at the landfill site the total energy recovered from each kilogram of waste generated is nine times higher in Stockholm than in Krakow.

One of the main goals of the waste disposal system is to minimize the waste stream which enters the landfill. Landfilling is seen as the last resource of waste disposal. The ratio of landfilled waste to waste generated is an indicator of the efficiency of the waste disposal system. In Stockholm, this indicator is four times higher than in Krakow. This is mainly due to the application of the incineration technology as the main technology of waste disposal. Also the extensive application of recycling has positive, but marginal impact on waste disposal ratio.

INTEGRATION OF THE EMISSIONS FROM THE MSW DISPOSAL SYSTEMS

The IWM model also delivers information about the environmental emissions generated during the whole stage of waste disposal. Applying different technologies of waste disposal results in a different spectrum of emissions. Such information is too fragmented to allow any analysis of different disposal systems. According to the literature (US EPA, 1995) there are 12 methods of characterizing the impact of man’s activity on human health, ecosystems and/or natural resources. Not all methods can be used in all cases and some are more appropriate for assessing specific impact categories.

The method of Environmental Standards Relation (ESR) seems to be the best suited for the analyzed case. The purpose of ESR is to assess chemical releases to air, land, and water based on their relative potential ecological and human impact. The emission fee was used as a media specific weighting factor. If the emission fee fully covers the external cost of the pollution, by calculating the total fee one obtains the total cost to the environment caused by each option of MSW disposal.
The Polish law implements the emission fees, but it occurs that if treating the maximum allowable concentration of the pollutant in the ambient air as an indicator of the components toxicity, the emission fees for different pollutants are inconsistent with their toxicity. Assuming that the maximum allowable concentration of different pollutants is the good indicator of the toxicity the new, modified emission fees were calculated. For each component, the modified emission fee was defined as: the product of the emission fee for the sulphur dioxide and the ratio of the imission standard of this component to the imission standard of sulphur dioxide. In case of emissions to

water, the reference component was not sulphide dioxide, but standards for sulfates discharged with the effluent.

Such method leads to one indicator for the environmental impact of the whole waste disposal system. Additionally, applying such a method both, economic and environmental impacts are in the same, monetary units and that fact allows the direct comparison of the systems. The comparison of the MSW disposal systems in the two cities is presented Table 2. The present and planned MSW systems in Krakow are compared with the existing system in Stockholm. The future system in Krakow was analyzed assuming it is processing the present amount of waste.

Table 2. Comparison of the Krakow and Stockholm MSW disposal system.

	Krakow – 2001	Krakow – future stage	Stockholm 2001
1. Waste stream [kg]	277 151 634	277 151 634	369 434 219
2. Economic cost of waste disposal according to IWM-1 model [€]	17 575 166	34 898 808	47 261 528
3. Economic unit cost [€/kg] (2 ÷ 1)	0,06	0,1259	0,13
4. Environmental unit cost (per kg of waste) [€/kg]	-0,000051	0,00032	0,00021
5. Total disposal cost per kg of waste [€/kg] (3 + 4)	**0,06**	**0,1262**	**0,13**

The present cost of disposal of one kilogram of waste in Krakow is half the cost in Stockholm, but if the system in Krakow is modified, and the incinerator is built, the costs in the two cities will be nearly equal. The present environmental impact of the Krakow system is negative. That means that the avoided emission, thanks to the recycling, is larger than the emission caused by the restwaste collection treatment and disposal. This positive effect will vanish if the new system is introduced; the environmental cost of the restwaste treatment will be larger than the environmental benefits from the recycling. The emissions in Krakow and in Stockholm will be comparable, but the emissions in Krakow will be larger mainly due to lower efficiency of the recycling programs.

Environmental impact, expressed in monetary terms, is insignificant to the economic cost of the waste disposal. Environmental cost is less than 1% of the economic cost. The economic cost alone seams to be <u>the</u> indicator of the waste disposal system.

LIFE CYCLE IMPACT ASSESSMENT (LCIA) AND INTERPRETATION

The second method of using the results from the IWM model, proposed by the authors, were one or even two next phases of the Life Cycle Assessment (LCA). These are: Life Cycle Impact Assessment (LCIA) and Interpretation. In LCIA phase the results of the inventory analysis is further processed and interpreted in terms of environmental impacts and societal preferences. To do so, the list of impact categories and indicators are defined. The indictors are weighted sums of selected items from the IWM result table. Finally, the category indicator results can be grouped and weighted to include societal preferences of the various impact categories.

According to the literature (Guinée, 2002) there are three types of the impact categories lists:

- Baseline impact categories, included in almost all LCA studies
- Study-specific impact categories, which may merit inclusion depending on the goal and scope of LCA study,
- Other impact categories, categories not having the baseline characterisation methods and which require further development before are used in the LCA.

To compare different systems of MSWM the following impact categories and indicators have been chosen:

Table 3. Selected impact categories for comparing different MSW disposal systems.

Impact category	Characterization factor	Unit of indicator results
Baseline impact categories		
Depletion of abiotic resources	Abiotic depletion potential **(ADP)**	kg (antimony eq.)
Climate change	Global Warming Potential **(GWP 100)**	kg(carbon dioxide eq.)
Human toxicity	Human toxicity potential **(HTP 100)**	kg (1,4-dichlorobenzene eq.)
Ecotoxicity: freshwater aquatic	Freshwater aquatic ecotoxicity potential **(FAETP 100)**	kg (1,4-dichlorobenzene eq.)
Ecotoxicity: terrestrial	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.)
Impact of land use	Land use	m^2yr
Study-specific impact categories		
Odour malodorous air	Reciprocal of odour threshold value **(1/OTV)**	m^3 (air)

The indicators (characterization factors) for the selected items from the inventory table have been adopted from Guinée (2002). Unfortunately, not all mentioned in the Table 3 impact categories can be calculated directly from the IWM-1 inventory table. The IWM-1 model gives no information about the land used, extracted minerals and fossil fuels.

The necessary values for calculating the impact categories such as: depletion of abiotic resources, impact of land use, and odour malodorous air need to be estimated. For estimation the above impact categories several assumption were made:

- Characterizing factors for depletion of abiotic resources (ADP) impact category were estimated by weight share of the particular element in the whole compound. For example: for SO_2 the weight share of sulfur was 16/48 and ADP factor was calculated by multiplying that share by ADP factor for sulfur taken from Guinée (2002).
- Impact of land use was estimated as follows: total volume of landfilled waste [m^3] was divided by an average landfill depth (15 meters assumed) and multiplied by an average occupation time (70 years assumed; 20 for exploitation and 50 for monitoring) and characterization factor (1 for all land use types) (Guinée, 2002).
- Odour malodorous air impact category was calculated on the basis of IWM-1 inventory table and additional information about the principal trace gas components concentration in the landfill gas (Young, Heasman, 1985).

During calculation of all assumed impact categories appears the same problem: the information given in the IWM-1 table is simplified, limited to certain number of the compounds and groups of compounds. In reality {and also in the tables with the characterizing factors given by Guinée (2002)} there are many more compounds responsible for the environmental impacts. For example: the IWM-1 lists only 22 compounds of air emissions while in the literature landfill gas consists of 26 compounds (Kreith, 1994). Leachate is characterized in the literature by 42 parameters (Bagchi, 1994) while the IWM-1 model characterizes only 23 parameters.

IWM-1 lists, for some groups of compounds, aggregated emissions. To calculate environmental impacts these combined emissions had to be substituted by a single compound. Table 4 shows the assumed substitutions.

Table 4. The leading substance for defining the characterizing factors for some compounds from IWM-1 emission table.

In IWM-1 model	Assumed for further calculations
NOx	NO_2
SOx	SO_2
HC	Propane
Chlor. HC	Trichloroethylene
Dioxins/Furans	2,3,7,8-TCDD
Chromium	Chromium (VI)
BOD	COD (1mg BOD = 2mg COD)
AOX	2,3,4,6-tetrachlorophenol
Chloride	Chlorine
Fluoride	Fluorine
Sulfide	Sulfur

Global Warming Potential (GWP 100) for carbon monoxide (CO) was assumed to be 1. The reason for that is that carbon monoxide (CO) survives in the atmosphere for a period of approximately one month and it is eventually oxidized to carbon dioxide (CO_2) (www.airquality.co.uk). For calculating GWP 100 – Global Warming Potential in 100-year-horizon – it seemed appropriate.

RESULTS OF LCIA

The calculated results of Life Cycle Impact Assessment for Krakow (future stage) and Stockholm MSW disposal system are given in Table 5.

Table 5. Comparison of the impact assessment indicators for Krakow (future stage) and Stockholm MSW disposal system (per Mg of processed waste and per capita).

	unit	Krakow future stage (per Mg)	Stockholm (per Mg)	Krakow future stage (per capita)	Stockholm (per capita)
ADP	kg antimony eq.	3,94E-04	2,25E-04	1,47E-04	1,10E-04
GWP 100	kg CO_2 eq.	7,63E+02	3,64E+02	2,85E+02	1,78E+02
HTP 100	kg 1,4-DCB eq.	1,61E+04	1,75E+04	6,03E+03	8,55E+03
FAETP 100	kg 1,4-DCB eq.	-1,09E+00	-9,64E-01	-4,08E-01	-4,71E-01
TETP 100	kg 1,4-DCB eq.	1,29E+00	1,40E+00	4,83E-01	6,85E-01
POCP	kg ethylene eq.	-8,59E-02	-4,53E-01	-3,21E-02	-2,21E-01
AP	in kg SO_2 eq.	7,84E-01	-1,98E+00	2,93E-01	-9,67E-01
EP	kg PO_4^{3-} eq.	4,87E-03	-1,68E-01	1,82E-03	-8,20E-02
ODP	kg CFC-11 eq.	1,51E-05	3,26E-05	5,63E-06	1,59E-05
1/OTV	m^3	-2,23E+06	7,84E+05	-8,33E+05	3,83E+05
Land use	m^2*yr	7,58E-01	9,60E-01	2,83E-01	4,69E-01

To allow the direct comparison of the two different waste streams from the two cities the environmental impact results were calculated per Mg of processed waste or per capita (Table 5).

Krakow future MSW disposal system performs worse than the Stockholm system if measured by: abiotic depletion potential (ADP), global warming potential (GWP 100), photochemical ozone creation potential (POCP), acidification potential (AP) and eutrophication potential (EP). All these indicators are measured per Mg of processed waste. All other indicators (HTP100, TETP100, ODP, 1/OTV, land use) are better for the Krakow system. The reasons for that are probably the higher amounts of waste generated (and incinerated) in Stockholm plus higher than in Krakow recycling rates. The planned composting facility in Krakow results in lower land use in Krakow. Also the use of biological methods of waste disposal in Krakow results in high values of “biological” indicators. If the performance of the systems is measured and compared not per Mg of waste, but per one city dweller the rating in different categories are generally the same with the exception of FAETP 100. This is because the index is negative and the fact that the waste stream in Krakow is smaller made the FAETP 100 per capita better in Stockholm than in Krakow.

NORMALIZATION

The next step of the LCIA is normalization. ISO 14042 defines normalization as “calculation of the magnitude of indicator results relative to reference information” (Stypka, 2005). The reference information can be the indicator which refers to the whole community, country, continent or even the world. Normalization is not mandatory, but strongly recommended step of any LCIA. As a result of this step the environmental profile is transformed into the normalized environmental profile in which the indicators are substituted by the ratios of the indicators to the values referring to the reference areas.

Because they still do not give a simple ranking of the analysed options there is a pressure to further aggregate the obtained results. Further aggregation is possible by “grouping” or by “weighting” process. These steps are optional, (but not recommended by ISO 14042) in LCA (Stypka, 2005). There is no specific methodology of weighing or grouping recommended by ISO 14042 and, if weighing is applied for comparative assertions the results can not be disclosed to the public. Dramatic drop of the objectivity of the results is the main problem (McDougall, 2001). The different LCIA indicators are of different level of objectivity, but the weighted scores are significantly below that level (Stypka, 2005).

Table 6. The normalized environmental profile for Krakow(future stage) and Stockholm MSW disposal system (in reference to West Europe, 1995 factors per year).

Characterization factor	Unit	Krakow future stage	Stockholm
ADP	kg antimony eq./yr [*E-08]	1,0	0,8
GWP 100	kg CO2 eq./yr [*E-08]	4 471	2 845
HTP 100	kg 1,4-DCB eq./yr [*E-08]	59 045	85 328
FAETP 100	kg 1,4-DCB eq./yr [*E-08]	-60	-71
TETP 100	kg 1,4-DCB eq./yr [*E-08]	757	1 094
POCP	kg ethylene eq./yr [*E-08]	-289	-2 029
AP	kg SO2 eq./yr [*E-08]	793	-2 667
EP	kg PO_4^{3-} eg./yr [*E-08]	11	-496
ODP	Kg CFC-11 eq./yr [*E-08]	5	15

The authors conducted normalisation with the reference information factors for West-Europe, 1995 from Guinée (2002). Table 6 presents the normalized environmental profile for Krakow (future stage) and Stockholm MSW disposal systems.

The most important impact on the environment is the impact on human beings (HTP100) and it has lower value in Krakow than in Stockholm. This is mainly because the incinerated waste stream in Krakow is smaller than in Stockholm (dioxins). The impact on global warming (GWP 100) is also relatively significant and twice as big in Krakow as in Stockholm. The full energy recovery from the incineratior which is used not only for generation of electricity, but also for district heating is the main reason. The MSWM systems impact on abiotic resources, on ozone layer as well as on eutrophication process is the smallest. The MSWM systems impact on air quality (acidification, smog) is relatively average and can be negative (as in Krakow) or positive (as in Stockholm).

CONCLUSIONS

The IWM model is one of few of the shelf models allowing the evaluation of the integrated MSWM systems, however if the results of the analysis are to be used by the decision makers the further integration of the results is necessary. The two potential methods of integration are presented. The first method based on Environmental Standards Relation shows that the environmental cost of the MSWM system is very small comparing to the economic cost.

The present cost of disposal of one kilogram of waste in Krakow is half the cost in Stockholm, but if the system in Krakow is modified, and the incinerator is built, the costs in the two cities will be nearly equal.

Energy consumption by the systems in the two cities is negative. Because Stockholm has a very advanced recycling system and has an incinerator in which the energy recovery rate is higher than at the landfill site the total energy recovered from each kilogram of waste generated is nine times higher in Stockholm than in Krakow.

The present environmental impact of the Krakow system is negative. This positive effect will vanish if the new system is introduced. The environmental cost of the restwaste treatment will be larger than the environmental benefits from the recycling. The emissions in Krakow and in Stockholm will be comparable, but the emissions in Krakow will be larger mainly due to lower efficiency of the recycling programs.

The second method of using the IWM-1 model in the decision process is to use the result table for the Life Cycle Impact Assessment and Interpretation. This approach does not allow such direct comparisons of the analyzed systems as the first method, but gives a lot of information which is also important and can be used by the decision maker.

Using this method requires some indirect transformation of the results from the IWM result table. The practical implementation of this transformation was presented. In the analyzed systems Krakow (future stage) MSW disposal system definitely results in higher: abiotic depletion potential (ADP), global warming potential (GWP 100), photochemical ozone creation potential (POCP), acidification potential (AP) and eutrophication potential (EP). The Stockholm MSW disposal system puts more stress on human toxicity (HTP 100), terrestrial ecotoxicity (TETP 100), ozone layer (ODP), reciprocal of odour threshold value (1/OTV) and impact of land use. The reasons for that are probably the higher amount of wastes generated in Stockholm combined with lack of biological treatment of waste. The impact on human health (HTP 100) is relatively the most important while the impact on ozone layer and on abiotic resources is relatively the smallest.

The next step of the second method of data integration should be the process of grouping and weighting obtained indicators to include societal preferences of the various impact categories.

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REFERENCES

Bagchi, A. (1994): *Design, Construction and Monitoring of Landfills*. John Wiley & Sons, Inc. NY.

Barlaz, M.A., Ranjithan, R., Weitz, K.A. and Nishtala, S.R. (1995): *Life-Cycle Study of Municipal Solid Waste Management, System Description.* US Environmental Protection Agency, USA.

Bjorklund, A. (1998): *Environmental Systems Analysis Waste Management with Emphasis on Substance Flow and Environmental Impact.* Licentiate Thesis. Stockholm University of Technology.

Bokota, A. (2004): *Comparative Analysis of MSW Management in Krakow and Stockholm (in Polish).* M.Sc.Thesis. Cracow Univeirisity of Technology.

Chang, N.B., Shoemaker, C.A. and Schuler, R.E. (1996): *Solid Waste Management System Analysis with Air Pollution and Leachate Impact Limitations.* Waste Management and Research 14, pp. 463-481.

Clapham, W.B. (1986): *RECYCLE: A Computerized Planning Tool to Improve Municipal Solid Waste Management.* Ohio Journal of Science 86(4), pp. 189-199.

Eriksson, O., Frostell, B., Bjorklund, A., Assefa, G., Sundqvist, J.O., Granath, J., Carlsson, M., Baky, A., Thyselius, L. (2002): *ORWARE-a simulation tool for waste management.* Resources, Conservation and Recycling 36 (2002), pp. 287-307.

Guinée, J.B. (2002): *Handbook on Life Cycle Assessment – Operational Guide to the ISO Standards.* Kluwer Academic Publishers.

Jenkins, L. (1982): *Parametric Mixed Integer Programming: An Application to Solid Waste Management.* Management Science **28**(11), pp. 1270-1284.

Kopacz, K. (2003): *Analysis of Integrated Solid Waste Management in Krakow (in Polish).* M.Sc.Thesis. Cracow Univerisity of Technology.

Kreith, F. (1994): *Handbook of Solid Waste Management.* McGraw Hill.

McDougall, F., White, P., Franke, M., Hindle, P. (2001): *Integrated Solid Waste Management: A Life Cycle Inventory*. Blackwell Publishing.

Stypka, T. (2005): *Adopting the Integrated Waste Management Model (IWM-1) into the decision process.* International Polish-Swedish Seminar on Sustainable Municipal Solid Waste Management. Kraków 2005, (in printing.)

Sundberg, J. (1995): *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, Sweden, 28-29 Sept. 1995, pp. 252-258.

US Environmental Protection Agency (1995): *Life – Cycle Impact Assessment: A Conceptual Framework, key Issues, and Summary of Existing Methods.* EPA 452/r-95-002, USA.

Wang, F.S., Richardson, A.J., Roddick, F.A. (1988): *SWIM – A Computer Model for Solid Waste Integrated Management.* Computers, Environment and Urban Systems 20(4/5), pp. 233-246.

White, P.R., Franke, M., Hindle, P. (1997): *Integrated Solid Waste Management - a Lifecycle Inventory.* Blackie Academic & Professional.

(www.airquality.co.uk) The UK National Air Quality Information Archive (Jan. 10, 2006)

Young, P.J., Heasman, L.A. (1985): *An assessment of the odour and toxicity of the trace components of landfill gas.* GRCDA 8th International Landfill Gas Symposium San Antonio, Texas, April 1985, pp. 93-113.