Life Cycle Assessment of Electronics

Conference Paper · January 2014

DOI: 10.13140/2.1.4893.1840

4 authors, including:

Otto Andersen
Vestlandsforsking

Geoffrey Gilpin
Høgskulen på Vestlandet

Anders S.G. Andrae
Huawei Technologies

Life Cycle Assessment of Electronics

Otto Andersen
Stiftinga Vestlandsforsking / Western Norway Research Institute (WNRI)
Fosshaugane Campus
Sogndal, Norway
oan@vestforsk.no

John Hille
Stiftinga Vestlandsforsking / Western Norway Research Institute (WNRI)
Fosshaugane Campus
Sogndal, Norway

Geoffrey Gilpin
Stiftinga Vestlandsforsking / Western Norway Research Institute (WNRI)
Fosshaugane Campus
Sogndal, Norway

Anders S. G. Andrae
Huawei Technologies Sweden AB
P.O. Box 54, 16494, Kista, Sweden

***Abstract*—Life Cycle Assessment (LCA) is a methodology for assessing the environmental aspects and potential impacts throughout a product´s life cycle from raw materials and energy extraction, components manufacture, assembly, distribution and sale, use and final end-of-life treatment such as disposal, recycling and energy recovery (i.e. cradle-to-grave). The environmental and resource impacts include climate change, stratospheric ozone depletion, toxicological stress on human health and ecosystems, the depletion of resources, water use and many others. This paper presents and discusses cases where LCA is used for assessing the environmental impact of electronics products and processes. Included are consumer electronics products, interconnect technology in electronics micro-integration, photovoltaic (PV) solar cells, and electric vehicles.**

Keywords—LCA; electronics; environmental impacts

I. INTRODUCTION

Life Cycle Assessment (LCA) is an approach / methodology for assessing environmental aspects and potential environmental impacts throughout a product´s life cycle. The life cycle usually comprise of raw materials and energy extraction, components manufacture, assembly, distribution and sale, use and final end-of-life treatment such as disposal, recycling and energy recovery (so-called "cradle-to-grave"). The environmental and resource impacts commonly included in a LCA are climate change, stratospheric ozone depletion, toxicological stress on human health and ecosystems, and the depletion of energy and material resources.

A. The ISO standard for LCA

There is a standard for LCA established by the International Standards Organization (ISO) that provides general principles for the methodology and content of a LCA [1]. The LCA phases and their interrelation are depicted in Fig. 1.

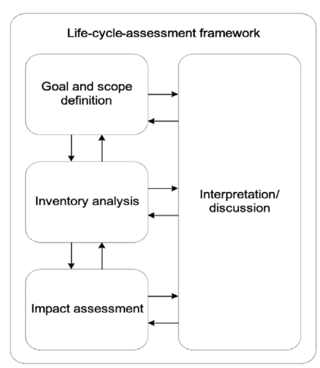

Figure 1. Methodological framework of life-cycle assessment according to the ISO 14040 series.

This work was financially supported by the European Commission through the project "Development of a competitive zero Global Warming Potential (GWP) dry process to reduce the dramatic water consumption in the ever expanding solar cell manufacturing industry – SOLNOWAT", grant agreement no. 286658.

Support was also obtained from the European Commission project "Functional joining of dissimilar materials using directed self-assembly of nanoparticles by capillary-bridging - HyperConnect" grant agreement no. 310420

In the *goal and scope definition* phase the scope is being defined sufficiently well to ensure that the breadth, depth and detail of the assessment are compatible and sufficient to address the goal. This implies definition of a so-called functional unit, to which the environmental impacts are expressed in relation to. Typical functional units for electronics products are “one year of usage” or “thousand units”. The functional unit is not necessarily just a quantity of material. Comparison of alternative types of electronics components can for example be on the basis of kg CO2-equivalents (CO2e) per m3 of packed and delivered product, or per service that the product provides.

The *inventory analysis* is the establishing of a Life Cycle Inventory (LCI) for the defined system to be studied. This involves data collection and quantification of material and energy inputs and outputs (e.g. emissions) for the system. Due to the complexity of gathering these type of data for all sub-parts of the system, it is common to use databases with life-cycle inventories for this purpose. The Swiss EcoInvent database is often used.

Impact assessment aims to identify the significance of the impacts of the established inventory. This is done by evaluating the inventory data according to the various environmental impact categories.

The *interpretation / discussion* phase is connected to the other three phases. It constitutes identification of significant issues based on the results of the work conducted in those previous phases. An evaluation of the results is undertaken, to determine the reliability of the results. This leads to conclusions, limitations and recommendations.

B. Attributional vs. Consequential LCA

When conducting an LCA of a product or technology, this is commonly done either as an attributional LCA (aLCA) or a consequential LCA (cLCA). In an aLCA, all the environmental impacts created in the life cycle of the product are detailed and summarized. A cLCA goes further. It aims at elucidating the consequences of a shift in a product or a technology, e.g. a shift from combustion based to electric vehicles. Thus, cLCA has the properties to function as a modeling tool for predicting the future environmental consequences of technology shifts [2].

II. CASE 1: CONSUMER ELECTRONICS PRODUCTS

Electronics products have undergone a tremendous change the recent decades with rapid technology development. Many LCAs have been conducted to determine the environmental implications of this development for consumer electronics products. The consistency between different LCA studies for desktop computers, laptop computers, mobile phones and television sets have also been assessed [3], [4].

A. Laptop Computers

The results from the various LCAs that has been carried out on laptop computers are not very consistent. This is shown in Tab. 1 where some of these results from the studies are compared.

TABLE I. LCAS OF LAPTOP COMPUTERS

Parameters	**Individual results from 5 different LCAs**				
Regional relevance	*Taiwan [5]*	*Japan [6]*	*Switzerland 7-9]*	*Global [10]*	*Europe [11]*
Weight of computer (kg)	2.3	2.3	3.2	1.5	2.5
Emissions (kg CO2e/kg)	23	113	206	273	100
% of emissions in various life cycle phases;					
production incl. transport	n.a.	44	93	41	36
use	n.a.	53	7	63	64
end-of-life	n.a.	3	0.1	-5	-0.4

First of all it is clear from the results in Tab. 1 that the value of 23 kg CO2e/kg for life cycle emissions of the Taiwanese laptop computer is likely to be too low. This result is not consistent with the fact that integrated circuits typically have life cycle CO2e emissions of at least 1000 kg per kg product, and that integrated circuits make up about 10% of the total weight of the computer.

Another inconsistency shown in Tab. 1 is that the study from Switzerland is very different from the others with respect to share of emissions in the various life cycle phases. The data presented for this, which is a compilation of three datasets from the EcoInvent database, indicated by far the most emissions occur in the manufacturing (including transport) phase, while for the others it is the use phase that is contributing the most.

B. Mobile Phones

The results from a comparison of different LCAs of mobile phones are shown in Tab. 2.

TABLE II. LCAS OF MOBILE PHONES

Parameters	**Individual results from 5 different LCAs**		
Regional relevance	*Finland [12]*	*Global [10]*	*Sweden/ China [11]*
Weight of phone (kg)	0.08	0.25	0.08
Emissions (kg CO2e/kg)	180	120	250
% of emissions in various life cycle phases;			
production incl. transport	71	93	80
use	29	13	20
end-of-life	0	-7	0

The results for the life cycle CO2e emissions in the studies of mobile phones shown in Tab. 2 are rather consistent in terms of life cycle stage shares, compared to the studies of laptop computers. However, there are large differences between the figures for the emissions per kg product.

III. CASE 2: INTERCONNECT TECHNOLOGY IN ELECTRONICS MICRO-INTEGRATION

Use of polymer particles in electronics micro-integration has been assessed for life cycle environmental impacts [4], [14]. Metal plated mono-disperse polymer particles (MPPs) can be integrated into lead-free solders through ball grid array (BGA) and chip scale packaging (CSP) technologies.

BGA is a surface–mount package that utilizes an array of metal spheres (or balls) as means of providing external electrical interconnection, as opposed to the pin grid array (PGA) which uses an array of leads for that purpose. The balls are composed of solder, and are attached to a laminated substrate at the bottom side of the package. The Si die of the BGA is connected to the substrate either by wire bonding or flip–chip connection. Metal plated MPP ball technology can replace conventional metal balls for BGA and CSP applications. The hypothesis is that the shift will be eco–efficient for several reasons. First of all, the polymer core particles increase the reliability by improving the interconnection compliance compared to compact metal cores [15]. Smaller sized MPP balls can achieve higher reliability in BGA/CSP [16]. A key issue is that smaller particles are less flexible than larger ones.

Moreover, it has been predicted that by introducing polymer cores, the reliance of toxic heavy metal use can be reduced by a factor of 3–7 for the same size balls [17]. In Fig. 2 the scope for an LCA of a common 256 pin BGA (BGA-256) is shown [14].

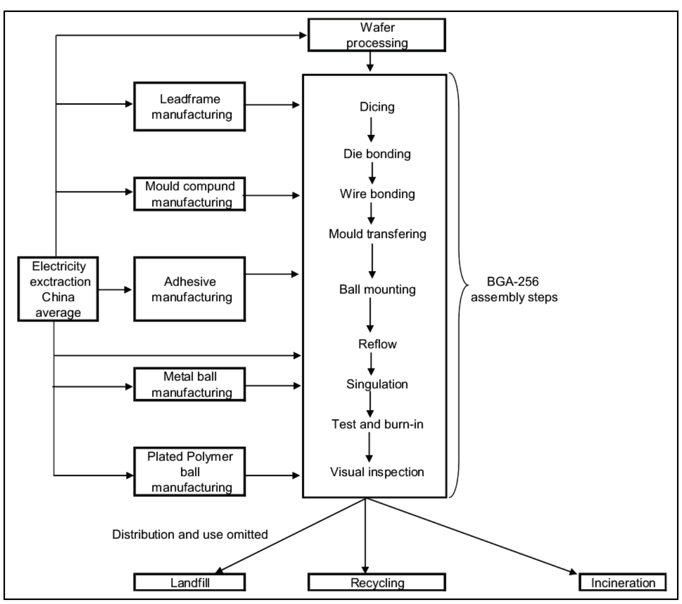

Figure 2. Scope of BGA-256 screening LCA

For this LCA it was used as basis that the conventional balls are produced in China with the Uniform Droplet Spray process, in which metals are melted and forced through an orifice, creating a laminar jet that is broken into uniform droplets [18], [19]. In [19], for each experiment about 200 g of solder alloy was filled into the crucible for spraying. Every experiment started with evacuating the chamber to 10 Pa and backfilling with pure nitrogen. The chamber pressure was maintained at 70 kPa after which the crucible was heated to 300 °C, holding this temperature for 15 minutes.

Metal plated MPP balls are produced by first preparing the polymer particles via the patented Ugelstad process, and then metallization of the polymeric spheres. Per ball these emulsion polymerization processes are very energy efficient compared to the Uniform Droplet Spray process. The results of calculations of GWP100 and Eco-Indicator'99(H) shares of BGA-560 microchips are shown in Fig. 3.

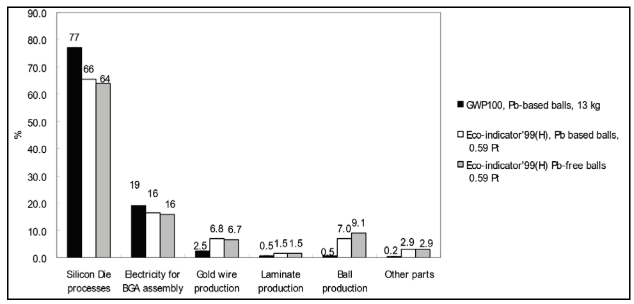

Figure 3. GWO100 and Eco-Indicator'99(H) of a BGA-560 microchip

From the results shown in Fig. 3 of the GWP100 and Eco–indicator'99 (H) shares of BGA–560 constituents a conclusion can be drawn that GWP100 scores for the life cycle of BGAs microchips are dominated by the CO2e-emissions from the silicon die. Solder balls have low CO2e-emissions, but Eco–indicator'99 (H) contribution from ball production are not negligible in metal BGA–560 microchip packages. The environmental impacts of changing from solid metal balls to metal plated polymer balls are shown in Fig. 4.

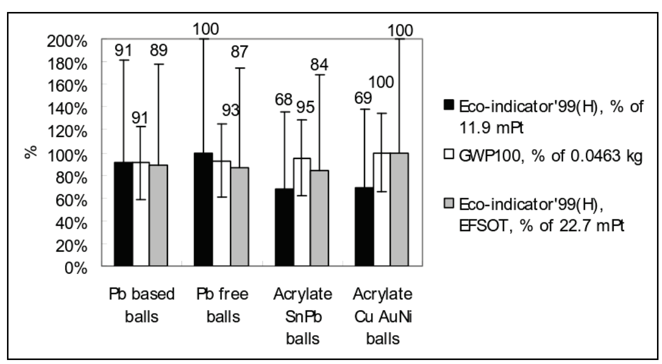

Figure 4. Environmental impact of changing to polymer balls

The polymer balls have larger life cycle GHG emissions, but lower Eco-Indicator´99(H) score than Pb-based and Pbfree balls. The updated version of Eco-Indicator´99(H), the so-called Environment Friendly Soldering Technology (EFSOT), includes gold and silver resource indices, give as result higher impact from the gold plated polymer balls.

It is the gold production that dominates the environmental impacts, both expressed as GWP100 and Eco–indicator’99(H), for gold plated balls. This is shown in Fig. 5.

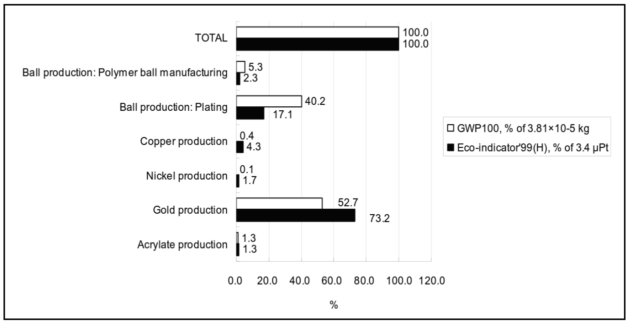

Figure 5. Relative contribution of environmental impacts from each sub-process in the life cycle of gold plated acrylate Cu AuNi balls

The gold production is thus the key component in the life cycle of gold plated polymer particles, from an environmental point of view.

The comparison of life-cycle impacts from the use of various types of metals is essential in making good choices during micro-integration technology development. When there is a possibility for making a choice between metal types, availability of life cycle impact data for the metals can constitute important decision support in green electronics.

We present here life-cycle data on some relevant metals found in Version 3 of the Ecoinvent database. Ecoinvent provides data on many processes that are or may be involved in the production of the metals, including extraction, refining and in most cases secondary processing from one or more kinds of scrap. The data shown in Fig. 6-8 are based on Ecoinvent’s datasets for “Market for copper”, “Market for nickel”, “Market for tin”, “Market for aluminum, primary, ingot”, “Market for silver” and “Market for gold”. The data for aluminum thus refer to 100 % primary metal. Underlying datasets show that the “Market for…” data for tin also refer to primary metal, while those for gold refer to a mix including 99.9 % primary metal and those for nickel refer to 96 % primary metal. However, those for copper refer to a mix including 38 % secondary metal from several processes, and those for silver to a mix dominated by secondary metal (98 %).

The primary metal entering into the “Market for…” datasets generally represents a mix of several sources and processes. However, one cannot be certain that the selection of mines and plants whose activities and environmental impacts have been studied, exactly represent an average of suppliers to the global market (or did so in 2011).

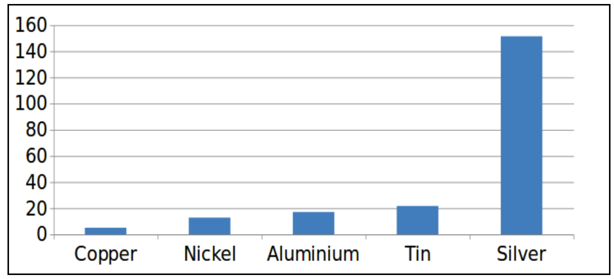

Figure 6. Global warming from the life-cycle of key metals used in electronics microintegration (GWP100/kg metal). Results are based on attributional LCA data in Ecoinvent category “market for” metals. For Ni, Al Sn and Au (Fig. 7), the data are indicated to be representative either solely or overwhelmingly (>96 %) to primary metal, whereas the “market” mix in the case of Cu and Ag includes large shares of secondary metal.

The advantage of the “Market for…” data is that they cover all processes from mining (or collection of scrap) up to and including distribution to an averaged global market.

The results shown in Fig. 6-8 are based on aLCA, where the impacts of processes leading to more than one product are split by allocation. Ecoinvent also has data based on cLCA, where it is taken into account what products are likely to be substituted for. In the cases of tin, aluminum ingots and gold, aLCA and cLCA yield quite similar results (differences of <10 % either way, for both GWP and Eco-Indicator´99(H)). For copper, cLCA yields significantly higher figures, and for silver much higher figures (almost six times higher in the case of GWP and over three times higher in the case of the Eco-Indicator´99(H)). For nickel, on the other hand, cLCA yields a significantly lower figure for GWP and in fact a slightly negative figure for the Eco-Indicator´99(H).

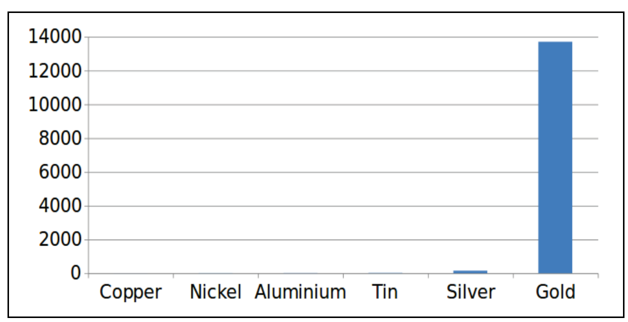

Figure 7. Global warming from the life-cycle of key metals used in electronics micro-integration (GWP100/kg metal). Same s Fig. 6 but with gold included, which would be “off the chart” in Fig. 6.

Ecoinvent provides separate figures for secondary metal based on aLCA for all metals considered here, except for tin. In the cases of aluminum, copper and nickel both GWP and the Eco-Indicator´99(H) are – unsurprisingly – very much lower than the data shown in Fig. 6-8, which refer to mixes containing 100 %, 62 % and 96 % primary metal respectively. The datasets considered most representative of secondary metal for the purpose of the comparison are “Treatment of aluminum scrap, post-consumer, prepared for recycling, at refinery, RER (RER=European conditions); “Treatment of copper scrap by electrolytic refining, RER”; and “Treatment of metal part of electronics scrap by electrolytic refining, in blister copper (output: 99,5 % Ni)”. As opposed to the “Market for..” figures, those for secondary metals do not include distribution to global markets, but this is hardly the main reason for their being very much lower. In the case of GWP the secondary-process figures are lower by a factor of 13 for Al, by a factor of 2 for Cu and a factor of over 100 for Ni.

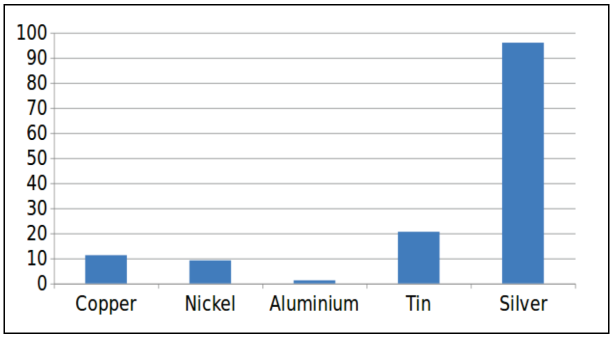

Figure 8. Eco-Indicator´99(H)scores of key metals used in electronics micro-integration. Gold is not included because, as was also the situation in the presented results in Fig.6, it would have been well “off the chart” with a score of above 25 thousand.

In the case of silver, however, “Treatment of precious metal from electronics scrap, in anode slime, precious metal extraction (silver), SE” yields a much higher GWP and only moderately lower Eco-Indicator´99(H) than the “Market for silver” results. This is not necessarily surprising, since most of the metal covered by “Market for silver” is secondary, from various sources, and the mentioned process based on electronics scrap represents only a single Swedish plant. Finally, “Treatment of precious metal from electronics scrap, in anode slime, precious metal extraction (gold) SE” – yields a GWP which is over twice as high, but an Eco-Indicator´99(H) score five times lower, than the “Market for gold” figures. The GWP figure in this case is somewhat surprising, since the “Market for gold” results are based on 99,9 % primary metal. The dataset for recovery of gold from electronics scrap represents the same Swedish plant as that for recovery of silver.

There are three main variants of the Eco-Indicator´99 indicator – “egalitarian”, “hierarchist” and “individualist”, the difference between them being the relative weight given to its three main components (ecosystem quality, human health and resources). In the case of the metals considered here, the “egalitarian” and “hierarchist” weightings yield quite similar results. Fig. 6-8 show results based on the “hierarchist” weighting, which the developers of the indicator also seem to regard as the most central or default option.

IV. Case 3: Photovoltaic (PV) Solar Cells

The case of LCA in photovoltaic (PV) solar cell production has basis in the fact that a new etching process for PV solar cells has been developed [20]. It is an alternative to the current wet chemical etching process, which has large environmental impacts in the form of high water consumption and emission of greenhouse gases with high GWP [21], [22], [23], [24], [25], [26].

A key problem with current PV cell production is the high water consumption in the etching of the crystalline silicon wafers. The wet chemical process, which is State of the Art (SoA) in PV cell production, involves the processing of the wafers in a series of consecutive and different chemical baths. Between each chemical treatment, the wafers need to be rinsed free from chemicals, using tens of thousands of liters of clean water in the process. In 2011, a leading USA solar cell company with manufacturing plants in Malaysia and the Philippines stated that their wet chemical-based process required as much as 15000 liters of high purity process water per minute for a 1.4 GW production facility for PV solar cells [20].

In order to compare the environmental impact of the new process to the corresponding impacts from state of the art wet chemical etching, a life cycle assessment (LCA) has been conducted [27].

The new PV etching process has been developed in the European Commission project *SOLNOWAT - Development of a competitive zero Global Warming Potential (GWP) dry process to reduce the dramatic water consumption in the ever expanding solar cell manufacturing industry* [20]. It has basis in previous research on plasma-based electronics etching [28, 29, 30, 31, 32, 33, 34, 35].

This new etching process is however taking place in atmospheric pressure and not in vacuum as has been common in dry plasma etching. It relies on the use of Fluorine (F_2) as the etching gas. This is a very toxic gas, which requires that emissions must be avoided. Therefore, regulatory emission control requirements and exposure limits for fluorine must also be taken into consideration in order to implement the necessary precautionary actions in the development of the process.

The goal of the study was to compare the texturing phase environmental impact of the dry- and the conventional wet-PV solar wafer etching processes, with particular focus on global warming potential and water consumption. The reason for doing this was to aid in the development of a dry alternative for the solar cell industry that would eliminate the very high water consumption and GWP emissions of current SoA process.

The life-cycle impact assessment results for two variants wet texturing process; acidic and alkaline, and the corresponding dry etching of two different type of wafers; with 4.3 watt and 4.5 watt per wafer efficiency respectively, are shown in Fig. 9. The results are normalized to the results of the method with highest score.

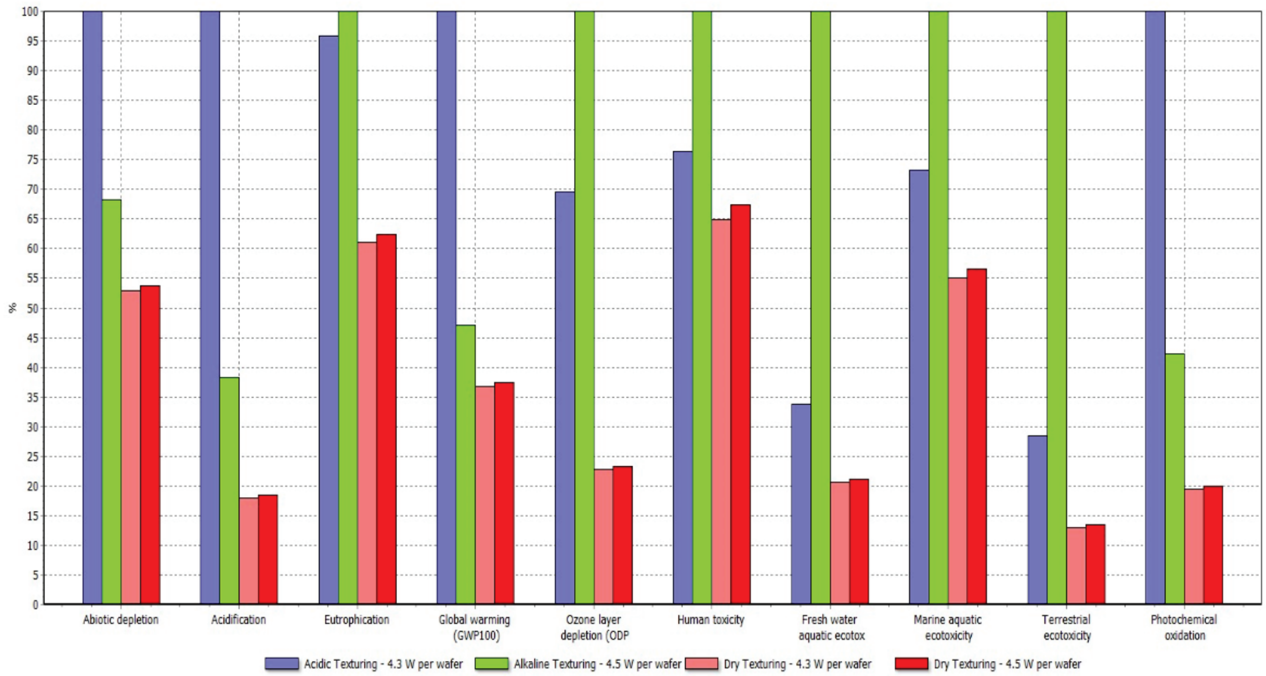

Figure 9. LCA results of the four etching methods

In Fig. 9 we see that the dry etching processes provide reductions for all impact categories with respect to their corresponding wet etching process. However, several of the impact categories have been determined with relatively large uncertainty, as in the results in Fig. 11. The results for the impact category GWP100 is shown in Fig. 10.

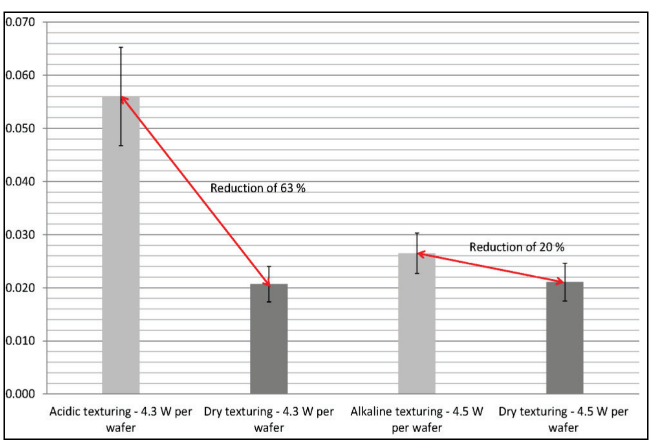

Figure 10. Life-cycle impact assessment results for GWP100 per wafer. Results normalized to the highest result of the four etching methods

In Fig. 10 it is shown that the dry texturing processes have potential for reductions of 63 % and 20 % with respect to their corresponding wet texturing processes, i.e. 4.3 W per wafer, and 4.5 W per wafer respectively. The uncertainty was determined to be within acceptable ranges.

Due to the fact that the dry process is based on the use of the very toxic etching gas fluorine (F2), it was important to include a closer look at toxicity in this assessment. Total toxicity, as presented, includes the results for the four impact categories 1)human toxicity, 2)freshwater aquatic eco-toxicity, 3)marine aquatic eco-toxicity, and 4)terrestrial eco-toxicity. The results are shown in Fig. 11.

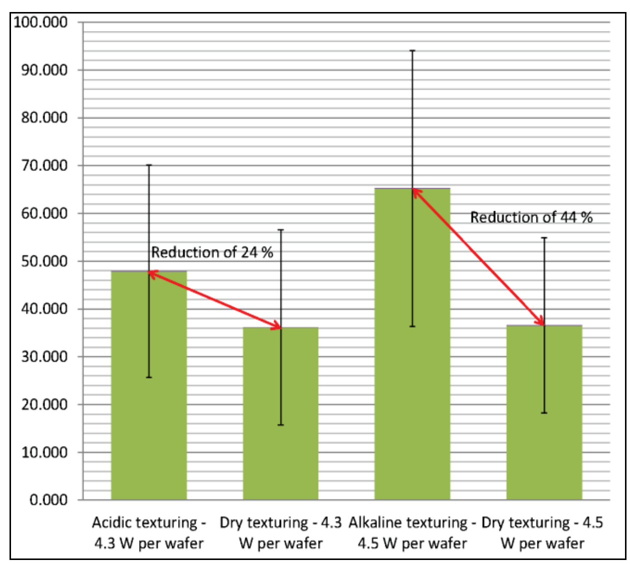

Figure 11. Life-cycle impact assessment for total toxicity (kg 1.4-dichlorobenzene (DB) equivalents per wafer) for all 4 texturing processes, including data uncertainty.

From Fig. 11 it can be observed that the dry texturing processes have the potential to lead to life-cycle toxicity reductions of 24 % and 44 % with respect to their corresponding wet texturing processes. The uncertainty was however determined to be high.

The results for the determination of water consumption are presented in Fig. 12.

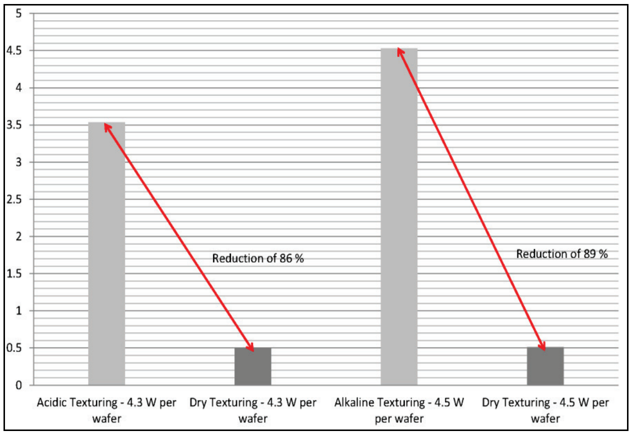

Figure 12. Life-cycle impact assessment for water consumption (liter per wafer) in the four texturing processes

In Fig. 12 it is shown that the dry texturing process has the potential for water consumption reductions of 86 % (v/v) and 89 % (v/v) for the two wafer types (4.3 W and 4.5 W) respectively, in comparison with the wet chemistry etching process.

Taking this all into consideration, it is possible to conclude that LCA results supports that the new alternative to current wet PV etching process has the potential to reduce the environmental impacts in the form of lower water consumption and emission of greenhouse gases. The LCA has shown that the dry process constitute a substantial improvement in the form of 85-89 % reduction in water use. The emissions of greenhouse gases are assessed to be reduced with 63 % and 20 % in comparison with current state of the art wet acidic and alkaline chemical etching respectively. Finally, the life cycle toxicity impacts appear not to be increased by the new process, even though the etching is facilitated through the use of highly toxic fluorine gas. However, this conclusion is based on relatively high degree of uncertainty, and the issue should be investigated further.

V. Case 4: Electric Vehicles

The last case is the application of LCA for electric vehicles. There are concerns that the production of electric vehicles will lead to negative consequences in terms of increased human toxicity, freshwater eco-toxicity, freshwater eutrophication, and metal depletion [36, 37, 38]. In particular, the production of electronic equipment necessary for an electrical vehicle requires a variety of metals, which poses a clear challenge for recycling and raises serious toxicity concerns. The life cycle human toxicity potential of electrical cars has been estimated to be 180–290 % higher than for cars with internal combustion engines [37]. This stems, to a large degree, from the high use of copper wires in electrical vehicles, and the use of nickel in cars with lithium-nickel cobalt-manganese batteries. The toxic emissions connected with the excavation of these metals occur, for the greater part, during the disposal of sulfide mine tailings. Similar elevated life cycle results from electric vehicles have been found for their eco-toxicity potential in freshwater systems.

Depletion of scarce metals is also a commonly sited concern with electric vehicles (see for example Gaines and Nelson [39, 40]. The life cycle metal depletion potential for electrical vehicles has been found to be 2–3 times higher than for combustion-based vehicles [37].

VI. Final Conclusions

The four cases in electronics illustrate and critically evaluate LCA as a methodology for assessing the environmental aspects and impacts in a product´s life cycle. This encompasses environmental and resource impacts, in the form of climate change, stratospheric ozone depletion, toxicological stress on human health and ecosystems, the depletion of resources, water use. This has been done for consumer electronics products, interconnect technology in electronics micro-integration, photovoltaic (PV) solar cells, and electric vehicles.

References

[1] International Standards Organization, Environmental management – life cycle assessment – principles and framework, Geneva, Switzerland: International Organization for Standardization, 2006.

[2] O. Andersen, “Consequential Life Cycle Environmental Impact Assessment,” in: Unintended Consequences of Renewable Energy. Problems to be Solved, London: Springer, 2013, pp. 35–45.

[3] A. Andrae and O. Andersen, “Life cycle assessments of consumer electronics — are they consistent?,” The International Journal of Life Cycle Assessment, vol. 15, no. 8, pp. 827–836, 2010.

[4] O. Andersen, H. J. Walnum, and A. Andrae, “Life Cycle Assessment of Electronics. Ugelstad-particles Ball Grid Array and Chip Scale Packaging,” Vestlandsforsking, Sogndal, Norway, 6/10, 2010.

[5] L. Lu, I. K. Wernick, T. Y. Hsiao, Y. H. Yu, Y. M. Yang, and H. W. Ma, “Balancing the Life Cycle Impacts of Notebook Computers: Taiwan’s Experience,” Resources, Conservation and Recycling, vol. 48, pp. 13–25, 2006.

[6] M. Tekawa, S. Miyamoto, and A. Inaba, “Life Cycle Assessment; An Approach to Environmentally Friendly PCs,” in IEEE International Symposium on Electronics and the Environment, pp. 125–130.

[7] Ecoinvent database, Name in db: Laptop computer, at plant/GLO U. 2008.

[8] Ecoinvent database, Name in db: Use, computer, laptop, office use/CH U. 2008.

[9] Ecoinvent database, Name in db: Disposal, laptop computer, to WEEE treatment/CH U. 2008.

[10] PE International, Environmental Footprint of ICT Equipment in Manufacture, Use, and End-of-Life. 2008.

[11] IVF, “Preparatory studies for Eco-design Requirements of EuPs. Laptop used in office,” 2007.

[12] Nokia, Integrated Product Policy Pilot Project. Stage I Final Report: Life Cycle Environmental Issues of Mobile Phones. Espoo, Finland, 2005.

[13] F. Bergelin, “Life cycle assessment of a mobile phone – a model on manufacturing, using, and recycling.,” 2008.

[14] A. Andrae and O. Andersen, “Life cycle assessment of integrated circuit packaging technologies,” The International Journal of Life Cycle Assessment, vol. 16, no. 3, pp. 258–267, 2011.

[15] J. Y. He, Z. L. Zhang, and H. Kristiansen, “Mechanical propert ies of nanostructured polymer particles for anisot ropic conductive adhesives.,” International Journal of Material Research, no. 98 (5), pp. 389–392, 2007.

[16] J. Y. He, Z. L. Zhang, M. Midttun, G. Fonnum, G. I. Modhl, H. Kristiansen, and K. Redford, “Size e ffect on mechanical propert ies of micron-sized PS–DVB polymer particles.,” Polymer, no. 49(18), pp. 3993–3999, 2008.

[17] D. Whalley and H. Kristiansen, “Development and application of polymer particles for ‘green electronics’.,” in Progress in Eco-Electronics, Warsaw, Poland: Radio and Tele Research Institute, 2008, pp. 61–67.

[18] W. Williams, “Production of uniform bronze powders by the uniform droplet spray process,” B.Sc., Massachusetts Institute of Technology, 1996.

[19] F. Yu, Z. Xia, and F. Guo, “Effects of rare-earth addition on properties and microstructure of lead-free solder balls,” Journal of Electronic Materials, no. 37, pp. 975–981, 2008.

[20] E. Duffy, “SOLNOWAT - Development of a competitive zero Global Warming Potential (GWP) dry process to reduce the dramatic water consumption in the ever expanding solar cell manufacturing industry. D7.3 Exploitation Plan (interim).” 2012.

[21] O. Andersen, Unintended Consequences of Renewable Energy. Problems to be Solved. London: Springer London, 2013.

[22] O. Andersen, “Solar Cell Production,” in Unintended Consequences of Renewable Energy. Problems to be Solved, London: Springer London, 2013, pp. 81–89.

[23] O. Zehner, “Unintended Consequences of Green Technologies,” in Green Technology, London: Sage, 2011, pp. 427–432.

[24] E. Alsema and M. de Wild-Scholten, “Environmental Impact of Crystalline Silicon Photovoltaic Module Production,” presented at the Materials Research Society Symposium, G: Life Cycle Analysis Tools for” Green” Materials and Process Selection, Boston, 2005, vol. Paper G3.3.

[25] V. M. Fthenakis, H. C. Kim, and E. Alsema, “Emissions from Photovoltaic Life Cycles,” Environ. Sci. Technol., vol. 42, pp. 2168–2174, 2008.

[26] V. M. Fthenakis and H. C. Kim, “Greenhouse-gas emissions from solar electric- and nuclear power: A life-cycle study,” Energy Policy, vol. 35, pp. 2549–2557, 2007.

[27] O. Andersen and G. Gilpin, “Environmental Assessment of Dry Photovoltaic Etching,” Vestlandsforsking, Sogndal, VF-Report 6/14, 2014.

[28] J. Rentsch, J. Jaus, K. Roth, and R. Preu, “Economical and ecological aspects of plasma processing for industrial solar cell fabrication,” in IEEE Xplore. Digital library., 2005, pp. 931 – 934.

[29] G. Agostinelli, H. F. W. Dekkers, S. De Wolf, and G. Beaucarne, “Dry Etching and Texturing Processes for Crystalline Silicon Solar cells: Sustainability for Mass Production,” presented at the 19th European Photovaltaic Solar Energy Conference, Paris, 2004, pp. 423–426.

[30] E. Lopez, I. Dani, V. Hopfe, H. Wanka, M. Heintze, R. Möller, and A. Hauser, “Plasma enhanced chemical etching at atmospheric pressure for silicon wafer processing,” in Poortmans, J.; European Commission, Joint Research Centre -JRC-: 21st European Photovoltaic Solar Energy Conference 2006. Proceedings. CD-ROM: Proceedings of the international conference held in Dresden, Germany, 4 - 8 September 2006, Dresden, Germany, 2006, pp. 1161–1166.

[31] E. Lopez, H. Beese, G. Mäder, I. Dani, V. Hopfe, M. Heintze, R. Moeller, H. Wanka, M. Kirschmann, and J. Frenck, “New developments in plasma enhanced chemical etching at atmospheric pressure for crystalline silicon wafer processing,” in The compiled state-of-the-art of PV solar technology and deployment. 22nd European Photovoltaic Solar Energy Conference, EU PVSEC 2007. Proceedings of the international conference, Milan, Italy, 2007.

[32] D. Linaschke, M. Leistner, G. Mäder, W. Grählert, I. Dani, S. Kaskel, E. Lopez, V. Hopfe, M. Kirschmann, and J. Frenck, “Plasma Enhanced Chemical Etching at Atmospheric Pressure for Crystalline Silicon Wafer Processing and Process Control by In-Line FTIR Gas Spetroscopy,” in EU PVSEC Proceedings, Valencia, Spain, 2008, vol. 2DV.1.22, Wafer-Based Silicon Solar Cells and Materials Technology, Mono- and Multicrystalline Silicon Cells and Materials, Processing Technology, pp. 1907 – 1910.

[33] Photovoltaics World, “Champions of Photovoltaics: Cells and Modules,” Photovoltaics World, no. October, 2011.

[34] B. Dresler, D. Köhler, G. Mäder, S. Kaskel, E. Beyer, L. Clochard, E. Duffy, M. Hoffman, and J. Rentch, “Novel Industrial single sided dry etching and texturing process for silicon solar cell improvement,” presented at the 27th European Photovoltaic Solar energy Conference and Exhibition, Frankfurt, Germany, 26-Sep-2012.

[35] P. Piechulla, J. Seiffe, M. Hoffman, J. Rentsch, and R. Preu, “Increased Ion Energies for Texturing in a High-Throughput Plasma Tool,” in Proceedings at the 26th European Photovoltaic Energy conference, Hamburg, 2011.

[36] O. Andersen, “Towards the Use of Electric Cars,” in Unintended Consequences of Renewable Energy. Problems to be Solved, London: Springer London, 2013, pp. 71–80.

[37] T. Hawkins, B. Singh, B. Majeau-Bettez, and A. Strømman, “Comparative Environmental Life Cycle Assessment of Conventional and Electrical Vehicles,” Int J Ind Ecol, vol. 17(1), pp. 53–64, 2013.

[38] J. Johnson, E. Harper, R. Lifset, and T. Graedel, “Dining at the periodic table: Metals concentrations as they relate to recycling,” Environ Sci Technol, vol. 41(5), pp. 1759–1765, 2007.

[39] L. Gaines and P. Nelson, “Lithium-ion batteries: Possible materials issues,” Argonne National Laboratory, Argonne, IL, USA, 2009.

[40] L. Gaines and P. Nelson, “Lithium-ion batteries: Examining material demand and recycling issues,” Argonne National Laboratory, Argonne, IL, USA, 2010.