MEMORANDUM

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From: U.S. Green Building Council, Board of Directors
Date: Monday, February 26, 2007
Subject: TSAC Report on PVC

Four years ago, the Technical and Scientific Advisory Committee (TSAC) undertook the exploration of a question posed by the LEED Steering Committee, and raised as an issue by members through the LEED for Commercial Interiors development process: what is the technical and scientific basis for a PVC-related credit within the LEED Green Building Rating System. Having completed their intensive study of the issue, TSAC's report is attached. The publication of TSAC's report concludes one process, and begins another.

Through the course of this assessment, larger questions became evident. TSAC has thoughtfully raised these in their recommendations.

The built environment itself is a complex system of systems. Therefore, it's no surprise that the study of any one material would offer no simple yes or no answers.

In order to apply TSAC's findings to a decision about credits within LEED, the report points to the fact that we must first address a series of policy issues raised through TSAC's research. These issues include:

- How should risks to human health and risks to the natural environment be reconciled?
- Should LEED offer credits for avoiding less desirable materials, or create credit incentives for the use of preferable, often innovative alternative materials or processes?
- Should LEED address individual materials through its credits, or should it focus on areas of impact?

TSAC's report will provide invaluable technical input to USGBC's policy-making processes, and will be applied together with our Guiding Principles:

- PROMOTE THE TRIPLE BOTTOM LINE: USGBC will pursue robust triple bottom line solutions to clarify and strengthen a healthy and dynamic balance between environmental, social and economic prosperity.
- ESTABLISH LEADERSHIP: USGBC will take responsibility for both revolutionary and evolutionary leadership by championing societal models that achieve a more robust triple bottom line.

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- RECONCILE HUMANITY WITH NATURE: USGBC will endeavor to create and restore harmony between human activities and natural systems.
- MAINTAIN INTEGRITY: USGBC will be guided by the precautionary principle in utilizing technical and scientific data to protect, preserve, and restore the health of the global environment, ecosystems and species.
- ENSURE INCLUSIVENESS: USGBC will ensure inclusive, interdisciplinary, democratic decision-making with the objective of building understanding and shared commitments toward a greater common good.
- EXHIBIT TRANSPARENCY: USGBC shall strive for honesty, openness and transparency.

Going forward, consistent with the 9-step process USGBC has defined for questions addressed by TSAC, the LEED Steering Committee will review the report and its recommendations, determine which policy issues to address first, and engage USGBC's Board of Directors and other member committees to develop policies and positions in response to the issues.

Regarding any proposed changes to LEED credits that may result from this process, USGBC's membership will be the ultimate arbiter through our consensus balloting process.

Assessment of the Technical Basis for a PVC-Related Materials Credit for LEED

February 2007

Authored by the LEED® Technical and Scientific Advisory Committee (TSAC) PVC Task Group:

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Preface

This report has been prepared under the auspices of the U.S. Green Building Council's LEED Technical and Scientific Advisory Committee (TSAC), in response to a charge given TSAC by the LEED Steering Committee to review the technical basis for considering a PVC-related materials credit in LEED. To undertake this assignment, the TSAC empaneled an *ad hoc* PVC Task Group (PVC TG), consisting of Scot Horst (Chair), Dr. Kara Altshuler, Nadav Malin, and Dr. Greg Norris; biographical data on the TG appears in the Appendix of this report.

TSAC has developed a nine-step process for preparing analyses of technical issues, and one of the most important elements of this process is obtaining input from the various stakeholders on an issue. That step occurred in stages: in 2004 on the methodology of the analysis, and in 2005 on comments to the draft report. As a result of this public input process, further significant analysis was done by the PVC TG which is reflected in this final report.

In addition to the revisions based on stakeholder input, the key elements of the analysis and findings were also peer reviewed by independent scholars and experts not involved in the work of the PVC TG (see Appendix J). The results of those peer reviews are also reflected in this final report.

This final report of the technical analysis performed by TSAC is being submitted to the LEED Steering Committee, which will make any policy determinations for LEED that need to be made as a result of the technical analysis.

TSAC wishes to express its deep appreciation to the members of the PVC TG for their diligent and ground-breaking work, to the peer reviewers, and to the many stakeholders for their input and engagement during the lengthy process of our work.

Malcolm Lewis, TSAC Chair
Joel Ann Todd, TSAC Vice Chair
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Table of Contents

1 Executive Summary

In the year 2000, during the process of developing the LEED for Commercial Interiors rating system, draft credit language was developed to provide credit for the avoidance of PVC materials. The U.S. Green Building Council (USGBC) decided that further technical knowledge was needed to determine the soundness of this proposed credit. Pending resolution of this issue, all PVC-related Credit Interpretation Rulings and potential Innovation and Design credits were put on hold.

In November 2002, at the request of the USGBC's LEED Steering Committee, the Technical and Scientific Advisory Committee (TSAC) established a PVC Task Group to assist TSAC in investigating the charge of:

> "…reviewing the evidence offered by stakeholders and independent sources, and advising the LEED Steering Committee on the availability and quality of evidence as a basis for a reasoned decision about the inclusion of a PVC-related credit in the LEED rating system."

Methodology

In response to its charge, the PVC Task Group identified several building applications in which PVC-based materials have a significant market share. Then, to explore whether there was a basis for a PVC-related credit, the Task Group ***investigated whether for those applications the available evidence indicates that PVC-based materials are consistently among the worst of the materials studied in terms of environmental and health impacts***. Four applications representing a diversity of uses of PVC in buildings were chosen for study. For each application, several alternatives were selected to represent a broad range of commonly-used alternative products. Since the study was not designed to determine which alternative is "best" in any application, only whether PVC is consistently among the worst alternatives, it was not necessary to include all potential alternatives. Note that some forms of PVC are commonly called "vinyl"—the two terms are used interchangeably throughout this report. The applications and alternative materials for each application that were studied are described briefly below; more detailed descriptions of each of the materials and the assumptions used in the analyses are included in Section 3.5 of the report and in Appendix C.

Siding

Four types of siding—vinyl, aluminum, wood, and fiber-cement—were analyzed for this report. For aluminum, wood and fiber-cement, a painting cycle of 6 years was assumed. The paint was assumed to consist of water-based 71% of the time and solvent-based 29% of the time[1]. Appropriate fasteners were included in the model.

- Vinyl siding was assumed to have a useful life 40 years.
- Aluminum siding is assumed to have a useful life of 80 years. It was assumed to have a recycled content of 21.4%.
- Wood siding was modeled as beveled cedar siding (clapboards) installed over battens, with a 40-year useful life.

[1] The specific numeric inputs used for analysis are cited here and throughout the report, while recognizing that there are levels of uncertainty associated with the data such that the level of precision is not as high as might be implied by the number of significant digits in the inputs and outputs.

- Fiber cement siding was modeled with a 50-year useful life.

Drain/waste/vent pipe

The three most common materials used to make drain/waste/vent (DWV) pipe for building applications were selected for study: PVC, ABS (acrylonitrile butadiene styrene), and cast iron. The life span of all the pipe products has been assumed to be 50 years.

- ABS is assumed to contain only the ABS polymer.
- PVC is assumed to contain only PVC polymer.
- Cast iron is assumed to be made entirely from scrap iron and steel (100% recycled content).

Resilient flooring

Two PVC-based resilient flooring products—sheet vinyl and vinyl composition tile (VCT)—and two non-PVC products—linoleum and cork—were selected for study. Styrene butadiene flooring adhesive was included in the models for all except cork, which was assumed to use a water-based contact adhesive.

- Sheet vinyl was modeled using several different compositions to reflect the variation in the marketplace. The presumed life span of sheet vinyl is 15 years.
- Vinyl composition tile was modeled with a useful life of 18 years.
- Linoleum was assumed to have an 18-year useful life.
- Cork flooring was assumed to have a 50-year useful life.

Window Frames

Windows in the study have frame materials made entirely from PVC, aluminum, or wood, with the associated ancillary materials needed for their assembly and installation. The impact of the window frames on energy use while a building is occupied was included in the analysis.

- Vinyl window frames were modeled with the following framing components: PVC (85%), fiberglass (6%), galvanized steel (5%), and Ethylene Propylene Diene Monomer (EPDM) (5%).
- Aluminum window frames used in the model contained no thermal breaks, and therefore have much higher use-phase energy use than the others.[2] The framing components include aluminum with 21.4% recycled content (86%), fiberglass (6%), galvanized steel (4%), and EPDM (4%).
- Wood window framing components include wood (86%), aluminum (4%), fiberglass (2%), galvanized steel (2%), and EPDM (2%).

[2]In 2003, aluminum frames were 10% of the residential market and approximately 60% of these frames had no thermal break. In the nonresidential market, aluminum was 87% of the total (including curtain wall, storefront and manufactured windows) and approximately 26% had no thermal break.

To compare the impacts of alternative materials choices, two assessments — an environmental **life cycle assessment** (LCA) and a **risk assessment —** were completed for each material in each application. The analytical process used is summarized below, followed by the findings. Such assessments can never be better than the data used as inputs to the process. This is the best that can be done at this time. To assess the quantity and quality of the data publicly available on these topics, a Web-based relational database was created to link available information resources to the materials, life cycle stages, and environmental and human health impacts addressed. The quality and nature of the knowledge conveyed is also characterized. A total of approximately 2,500 references were reviewed as part of this analysis and are cited in the database (the database is available at http://pvc.usgbc.org and was last updated on December 14, 2004).

Life Cycle Assessment (LCA)

Conventional LCA. A life cycle assessment endeavors to quantify and characterize all of the resource and pollution flows (inputs and outputs) associated with a particular material over its entire life cycle: from the harvesting or extraction of raw materials, through manufacture, installation, use, and reuse or disposal. The individual inputs and outputs are quantified in a life cycle inventory. They are then characterized according to their estimated contribution to environmental and health impacts. These impacts are grouped into a set of *impact categories*. The impact categories and units of measure used in this study are based primarily on the U.S. EPA's TRACI method and include:

Environment/Resource

- Acidification – damage to forest and freshwater ecosystems from "acid rain" due to emissions of air pollutants
- Ecotoxicity – damage to ecosystems from toxic pollutants
- Eutrophication – changes to aquatic ecosystems due to introduction of excess nutrients
- Fossil Fuel depletion – depletion of resources of coal, petroleum, and natural gas

Combined human and ecosystem health

- Ozone Depletion – damage to the protective ozone layer high in the stratosphere
- Photochemical Smog – formation of ozone in the air that people (and plants) breathe

Strictly human health[3]

- Human Health: Particulates
- Human Health: Cancer
- Human Health: Non-Cancer
- Human Health: Global Climate Change
- Human Health: Mercury

[3] As discussed later, our method to integrate occupational risk and use-phase risk results (both for cancer and non-cancer impacts) with the pollution-driven population risk results (which are standard impact categories in LCA) has delivered some noteworthy conclusions for the LCAs and risk studies reported here: the occupational impacts did not appear to be at all trivial by comparison with the population risk impacts in these life cycles. This is partly due to the conservative method we have used to estimate occupational exposures, but nevertheless, it indicates that occupational exposures should not be automatically ignored in LCAs in the future.

A conceptual illustration of the method is provided in the figure below.

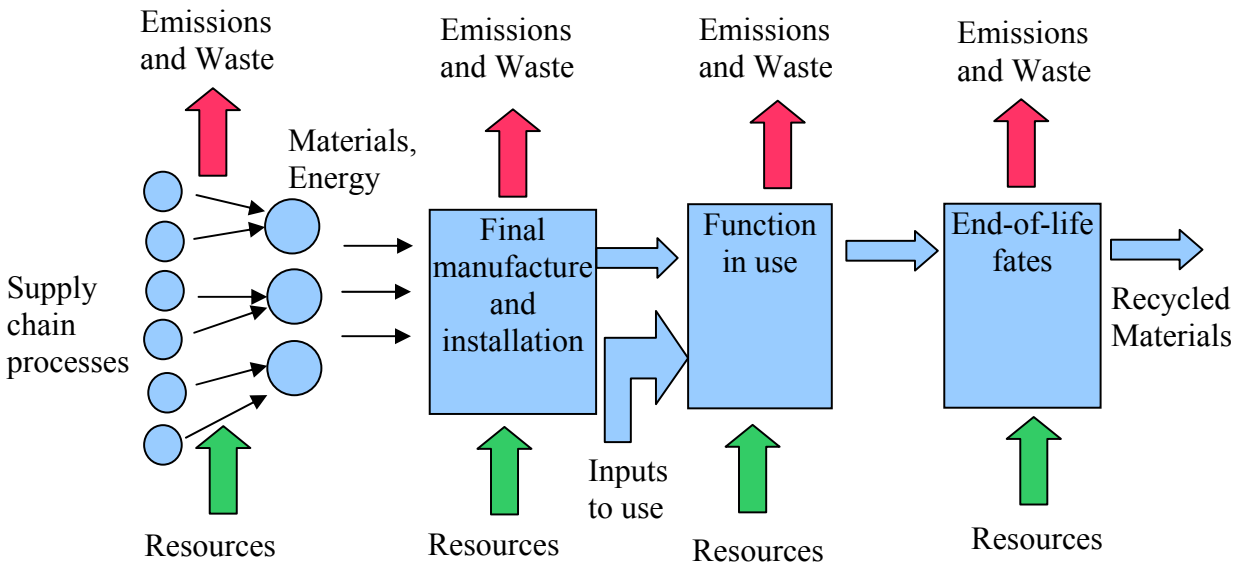

Figure 1-1: Scope of the life cycle studies in this report

Expanded End-of-Life Analysis. The Task Group obtained information indicating that accidental landfill fires and backyard burning during the end-of-life phase might be a significant source of localized dioxin concentrations. However, information on emissions for accidental landfill fires and backyard burning is very limited, and such accidental sources are usually omitted in conventional life cycle analyses. Given the potential importance of this life cycle phase in the total results, we developed methods to estimate the end-of-life impacts using available scientific knowledge in literature. Because the empirical basis for these models is limited, the end-of-life emission estimates are highly uncertain; we developed scenarios intended to represent upper and lower boundaries for dioxin emission factors per unit of material disposal and the results are presented for the high, low, and mid-range scenarios. The influence of these end-of-life dioxin emission uncertainties on our final conclusions and recommendations is addressed. These estimates were peer reviewed; see Appendix J.

Risk Assessment

Risk assessment quantifies potential risk of developing adverse health effects following exposures to environmental toxicants (compounds which have the potential for causing toxicity in living things). This quantification is performed by comparing doses of the toxicants a person receives through inhaling, ingesting, or absorbing a compound through the skin to a reference level or dose that a person can be exposed to on a daily basis with no anticipated adverse effects. The following risk assessment analyses were conducted:

- A human health risk assessment was conducted for occupational workers involved in the manufacture of PVC and alternative building materials addressed in our analysis (except

for wood and fiber cement siding, for which occupational risk data were not available). In addition, for resilient flooring materials only, potential risks from installation and use were modeled, based on a school classroom environment.

Exposure data and dose-response information are unavailable for wood and fiber-cement siding. However, it is known that certain types of wood workers get nasal cancers, and compounds (Portland cement, silica, etc.) in the production of fiber-cement siding contribute to adverse respiratory effects. Although the lack of occupational risk estimates for wood workers is of concern, there are no data to indicate that individuals living in areas around wood processing facilities are at increased risk for nasal cancers. The lack of data with regard to fiber-cement facilities is of greater concern, as respiratory problems are documented in neighborhoods adjacent to or downwind of Portland cement facilities.

- An assessment of exposure to phthalates in residential indoor air and dust was performed based on data gathered from several houses in Massachusetts and a risk assessment performed. A comprehensive risk assessment of end users in a home environment for all the building materials was not performed because comprehensive data on compounds emitted from the building materials was not available and/or exposure would be non-existent or negligible.

- Further, risk estimates were generated for residents potentially exposed to vinyl chloride monomer (VCM) whose homes were located in the vicinity of vinyl manufacturing plants in West Louisville, KY, and Baton Rouge and Calcasieu Parish, LA in order to determine how risks to the general population compare to occupational risk estimates.

Risk estimates were developed for both non-cancer and cancer effects. It should be noted that occupational risk estimates in this report were generated only for the purpose of comparing exposures and risks among the building materials considered, and tended to be upper-bound estimates because of the conservative methodology used. Risk estimates were not compared to any regulatory limits or guidelines and should not be interpreted in that context.

Peer Review

The TG actively sought out peer review of the revised draft report in an effort to verify validity of methodology and to gain constructive criticism of the approaches used. Sections of the report have undergone a peer-review process by recognized experts in the appropriate fields: phthalate toxicity and exposure, fenceline risk, analyses estimating dioxin emissions, and mercury exposure. See Appendix J for a complete discussion of peer reviews.

Findings

Integration of LCA and Risk Assessment

This section summarizes findings related to the study's goal: ***to determine whether, for the applications included, the available evidence indicates that PVC-based materials are consistently worse than alternative materials in terms of environmental and health impacts***.

Findings are summarized for the following types of impacts:

Human Health Impacts

- Cancer mortality from all pollutants, based on the life cycle assessment inventories, measured through disability-adjusted-life-years;
- Combined mortality associated with cancer, neurotoxic impacts of mercury, global climate change, and particulate matter, measured through disability-adjusted-life-years; and
- Other toxic risks, ranging from mild irritation through possible mortality, measured with Hazard Index.

Environmental Impacts

- Acidification
- Eutrophication
- Eco-toxicity
- Smog formation
- Ozone depletion potential
- Global climate change, and
- Fossil fuel depletion.

Findings are also summarized for different levels of ***life cycle comprehensiveness***:

- Cradle-through-use,
- With additional end-of-life impacts, and
- With additional occupational impacts.

Findings for the four applications are summarized below.

Window Frames

Human Health Impact

- Cradle-through-use: aluminum frames are worst among alternatives studied.[4]
- Addition of end-of-life including burning: aluminum frames remain worst for combined human health impacts, but PVC is worst for cancer-related impacts among alternatives studied.
- Addition of occupational exposures: PVC and aluminum are worst among alternatives studied.
- The rankings for other toxic risks are generally consistent with those for the mortality-related rankings.

[4] We analyzed aluminum windows with and without thermal breaks, using data on usage phase thermal performance from Lawrence Berkeley National Laboratory's RESFEN database, provided at http://www.efficientwindows.org. These data show that for an equivalent glazing type, thermally broken aluminum frame windows have a thermal efficiency that is intermediate between those of unbroken aluminum and vinyl frames. The results of life cycle comparisons indicate that the relative performance of wood, aluminum and vinyl window frames do not change when thermal breaks are added to the aluminum windows; their life cycle health and environmental impacts remain worse than those for PVC frames. A comparison of the results with and without thermal breaks is provided in Appendix K.

Environmental Impact. The aluminum window frames are worst overall for environmental impacts among alternatives studied due to their higher heat losses and gains and resultant higher energy use for heating and cooling during the use stage.

Pipe (drain, waste, vent)

Human Health Impact

- Cradle-through-use: cast iron is worst among alternatives studied.
- Addition of end-of-life: PVC is worst for cancer-related impacts, while cast iron is worst for combined health impacts among alternatives studied.
- Addition of occupational exposures: PVC is worst for cancer-related impacts, while cast iron is worst for combined health impacts among alternatives studied.
- For other toxic risks, cast iron is worst, followed by ABS, followed by PVC as the best among alternatives studied.

Environmental Impact. Cast iron pipe is consistently the worst material among alternatives studied for all environmental impact categories except eutrophication, for which ABS is the worst when end-of-life emissions are considered.

Siding

Human Health Impact

- Cradle-through-use: PVC is worst for cancer impacts, and PVC and aluminum are worse than alternatives studied for combined human health impacts
- Addition of end-of-life: PVC is worst for cancer impacts, and PVC and aluminum are worse than alternatives studied for combined human health impacts
- Addition of occupational exposures: PVC is worst for cancer impacts, and PVC and aluminum are worse than alternatives studied for combined human health impacts. Data on occupational exposures were available only for PVC and aluminum; data are not available for wood and fiber cement.
- For other toxic risks, aluminum tends to be worst among the alternatives studied, followed by PVC, with the conclusions depending on the emission factor uncertainties.

Environmental Impact. Among the alternatives studied, aluminum siding is found to be the worst material for many environmental impact categories, while fiber cement is the worst for fossil fuel depletion. When end-of-life impacts are included, wood is the worst for the smog effect.

Resilient Flooring

Human Health Impact

- Cradle-through-use: VCT and sheet vinyl are worst for cancer-related impacts; VCT and sheet vinyl vie with linoleum for worst on combined health impacts among alternatives studied.

- Addition of end-of-life: VCT and sheet vinyl are worst for cancer and combined health impacts among alternatives studied.
- Addition of occupational exposures: VCT and sheet vinyl are worst for cancer and combined health impacts among alternatives studied, except for the scenarios with lower-end estimates of occupational and end-of-life exposures.
- For other toxic risks, no material among alternatives studied consistently dominates. Vinyl is worst for the low-end emissions estimates, while cork is worst for the medium and high-end estimates.

Environmental impact. Sheet vinyl (not VCT) performs worst among alternatives studied on all impact categories except eutrophication, for which linoleum is worst.

Additional Analyses

Results from our combined review and analysis of evidence from life cycle assessment and risk assessment results, while as broad as possible, still do not cover all possible environmental and health impacts of the material alternatives. To capture some other possible health impacts, several additional analyses were performed. These included a review of the literature regarding the effects of phthalate exposure, health risks to populations living near vinyl production facilities, and the effects of PVC in structure fires.

Phthalate Exposure. Phthalates are used to plasticize vinyl, and thus are found in sheet vinyl and vinyl composition tile. Many phthalates, including DEHP, when administered in pregnant rats, result in adverse reproductive effects in male offspring which affect future reproductive success. It is speculated that exposure to environmental levels of phthalates in the environment may be contributing to decreased fertility in human males, but recent studies show that reproductive endpoints are not adversely affected when exposure to phthalates occurred only during adulthood, and that *in utero* and early post-natal developmental stages are primarily susceptible to phthalate-induced hormonal and cellular changes. Studies indicate that occupational exposures to phthalates are believed to be much greater than those in the ambient environment. Pregnant female workers are believed to be at risk, while males and non-pregnant females are not.

To estimate potential risks from residential exposures, measured levels of DEHP in the dust and air of homes in Cape Cod, Massachusetts were used. The risk estimates showed that, for dust, the childhood risk of non-cancer effects following ingestion of the maximum amount of DEHP in dust was three times the federal limit of 1.0, while risks for teens and adults were well below 1.0. DEHP in air did not pose a risk. The DEHP concentrations measured did not differentiate the sources of the phthalate. Further, because DEHP is used as a surrogate, the actual risk from exposure to plasticizers currently used in resilient flooring (sheet vinyl and vinyl composition tile) is expected to be much lower. Although some studies link phthalate exposure in homes with rhinitis and asthma in children, these studies cannot rule out other compounds that may have induced the effects. Recent reports indicate that phthalates may increase the allergic response from exposure to other compounds.

Health Risks to the General Population Living Near Vinyl Facilities. An assessment of data from Kentucky and Louisiana air monitoring stations was performed to evaluate the risk

from inhalation exposure to vinyl chloride monomer (VCM). Risk estimates for five different monitoring stations in Kentucky had cancer risk estimates that ranged from 1 in one million to 5 in one million. The cancer risks at five out of six air monitoring stations in the Baton Rouge, LA and Calcasieu Parish, LA areas also slightly exceeded U.S. EPA limits. These minor exceedances do not necessarily trigger any regulatory action by either the federal or state governmental agencies.[5]

PVC in Structure Fires. PVC has the potential to release toxic gases during building fires that, according to some analyses, can incapacitate building occupants and firefighters, reducing their chance of escaping unharmed. In particular, hydrochloric acid (HCl) can be generated from burning PVC materials, and from those materials before they have ignited. Highly toxic phosgene gas can also be released from burning PVC, and some hypotheses suggest this as the culprit in some otherwise unexplained health effects observed in firefighters. None of the scientific literature reviewed that included measurements of the actual compounds found in the air in structure fires showed problematic levels of either of these gases, however. It remains possible that HCl has the effects described in certain specific situations, but those do not appear to be typical. Since structural fires have many sources of toxic gases, any firefighter not using breathing apparatus would be taking an unnecessary risk, regardless of the specific materials present. Compared with other plastics, and other combustible materials, PVC may have a beneficial role in reducing injuries in structural fires, as it may reduce the chances of a fire igniting or spreading due to its relatively high ignition temperature. In summary, there is evidence that PVC contributes to hazardous conditions in building fires, but insufficient evidence to determine how widespread or consistent a risk that represents, how it compares to alternatives, how it relates to the possible fire retarding effect of such materials, and how these factors compare to the other human health and environmental impacts studied in this report.

Impact of these Additional Analyses

While each of the additional topics studied—phthalates, fenceline exposure, and structure fires—raises new questions and potential concerns, none provided information that compel changes to the basic findings from the combined LCA and risk assessment.

Summary of Findings

No single material shows up as the best across all the human health and environmental impact categories, nor as the worst. This primary finding from the integration of LCA and risk assessment is demonstrated in Table 1-1 and Table 1-2 which present the relative ranking of PVC and the alternatives studied for each application; more detail is provided below.

The life cycle performance of PVC relative to other materials depends upon two factors:

- Whether we focus on *human health impacts* or *environmental impacts*. (The Task Group refrained from weighting and aggregating the separate health and environmental results into a final overall "indicator" score because there is no scientific basis for so doing.)

[5] Risks of 1 in one million or less are generally considered negligible and acceptable by the federal government as well as many state governments.

- *Life cycle scope.* The performance of PVC relative to the alternative materials changes as we expand the life cycle scope from cradle-through-use, by adding end-of-life with accidental fires and backyard burning, and occupational exposures by integrating LCA and risk assessment.

The influence of these two scope dimensions on the relative performance of PVC is fairly consistent across the four product groups that were studied, and may be summarized as follows.

- Relative to ***human health impacts***, aggregated in terms of total risk of mortality and morbidity (including pathways of cancer, particulate inhalation, global climate change, and impacts of mercury exposures), the performance of PVC compared with other materials depends on the life cycle scope for the four product groups studied.
    - For a narrow life cycle "cradle-through-use" assessment, PVC performs better than some alternatives studied for window frames, siding, and drain-waste-vent pipe, while it performs worst among the resilient flooring materials studied.
    - When we add end-of-life with accidental landfill fires and backyard burning, the additional risk of dioxin emissions puts PVC consistently among the worst materials studied for human health impacts, unless the end-of-life emissions from landfill fires and backyard burning are near the lower end of the wide range of uncertainty about these emissions. When end-of-life emissions are near the mid-range value or nearer to the upper end of this range, landfill fires account for at least 80% of the total end-of-life dioxin emissions for PVC.[6]
    - When we also add occupational exposures that we were able to model (the literature was much less complete regarding occupational exposure data for manufacture of materials other than PVC), PVC remains among the worst materials studied for human health, although the data gaps affect the robustness of this finding.
- Relative to the ***environmental impact categories*** (acidification, eutrophication, ecotoxicity, smog, ozone depletion, and global climate change), PVC performs better than several material alternatives studied, regardless of the life cycle scope, for three of the four product groups; the exception is resilient flooring, for which sheet vinyl is consistently the worst material studied on all environmental categories except eutrophication.
- Risk estimates for ***residential exposures*** at air monitoring stations in Kentucky and Louisiana, which could not be integrated into the above findings, slightly exceed the state and federal acceptable cancer risk limits of 1 in one million, indicating that exposure of the general population to VCM in these two locations may result in an increased chance of developing cancer. These risk estimates are very conservative, because they are based on constant exposure from birth to adulthood and use the most conservative toxicity value for VCM. (Fenceline risk estimates were not generated for other building materials due to a lack of exposure data.)

[6] Risk assessments were not independently conducted for dioxins; therefore published toxicity values using a linearized multi-stage model were used. These toxicity values may be overestimates because non-linear models may be more appropriate given the non-genotoxic mechanism of dioxin-induced cancer (NAS, 2006). Non-linear models generally provide lower toxicity values than do linearized models. Thus, our risk estimates may *overestimate* the actual cancer risk associated with dioxin emissions in the life cycles of products in this report, which is especially relevant to uncontrolled burning of PVC.

Conclusions

In light of the findings summarized above, we draw the following conclusions about the expected impacts of a credit that rewards avoidance of PVC, for the four material alternatives studied and across the four product groups studied, and for life cycles that include risks from dioxin emissions from accidental landfill fires and backyard burning:

- ***Human Health Risk.*** The evidence indicates that a credit rewarding avoidance of PVC could steer decision makers toward using materials that are better for human health in the case of resilient flooring. If buyers switched from PVC to aluminum window frames, to aluminum siding, or to cast iron pipe, it could be worse than using PVC. Data on end-of-life emissions are highly uncertain and therefore there is a wide range of exposure possibilities; if end-of life emissions are close to the upper end of our range, then PVC is among the worst materials studied for health risk, but if end-of-life emissions are close to the lower end of our range of possible values, then PVC is among the mid or better materials studied for health risk in the product categories of window frames, pipe, and siding. Policies to prohibit backyard burning and reduce landfill fires would improve the profile of PVC in piping, windows and siding as compared to the other alternatives considered.
- ***Environmental Impact.*** The evidence indicates that a credit that rewards avoidance of PVC could steer decision makers toward using materials that are worse on most environment impacts, except for the case of resilient flooring, in which sheet vinyl and VCT are worse than the alternative materials studied for most environmental impacts.

The report discusses data gaps, or missing information related to PVC and competing materials, and provides a detailed assessment of the subject areas that, if information became available, could alter the results of the analysis.

Recommendations

The foregoing conclusions represent the TSAC's response to the charge assigned to it by the LEED Steering Committee. Additionally, based on its work on this issue TSAC has developed several recommendations related to how materials are assessed in LEED. These recommendations are separate from the formal conclusions and should be weighed with other factors as they are considered by the Steering Committee.

- ***Need for integrated methods for materials evaluation.*** The importance of the remaining data gaps, together with the demonstrated power of integrated analysis to find key chemicals and pathways within the life cycles of product alternatives, leads TSAC to recommend that the Steering Committee use the evidence provided in this report as a basis for working towards increased use of integrated methods for material evaluation, not only to pass judgment on a particular credit for a particular material.
- ***Need for credits based on a more complete assessment of environmental and human health concerns***. In the long-term, the Steering Committee is encouraged to consider developing credits informed by both LCA and risk assessment to address critical environmental and human health issues explicitly and more systematically. Such credits should use a comprehensive, whole-building approach to critical issues. Examples could

include a comprehensive whole building approach to issues such as bioaccumulative pollutants, particulate emissions and climate change.

- ***Need to address end-of-product-life phase of the life cycle.*** Our findings about the potential importance of the health impacts from accidental landfill fires and backyard burning argue for much greater attention to the end-of-product-life phase of the life cycle of PVC and other building materials. This means better data and modeling, and if the risk of major health impacts is confirmed by further empirical work, then policies to reduce this important source of health risk are recommended.
- ***Need to reward development and use of improved materials.*** Avoid the "blunt instrument" problem of material-based credits inadvertently steering decision makers to replace one high-negative-impact material with another, and instead create an ongoing market incentive for continuous development and improvement of building materials. This initiative includes two possible approaches: 1) by seeking out means for incentivizing the improvement of all buildings materials in terms of environmental and human health impacts; 2) incentivizing the substitution of problematic materials with others that are demonstrably better with regard to environmental and human health impacts over their life cycles.
- ***Need to gather and use information on occupational and life cycle impacts of products.*** The literature on occupational and fenceline risks during manufacturing is overwhelmingly focused on PVC relative to the other supply chains studied in this report, leaving large data gaps in assessments of those other product life cycles on these important topics. These data gaps need to be filled, and significant exposures then need to be included in product life cycle evaluations.
- ***Opportunity to engage Innovation and Design credits in LEED.*** Develop guidelines for approval of innovation credits that move the industry forward. Recognizing that there are many possible ways to address this challenge, the capabilities and motivation of the marketplace should be engaged as a resource. Without constraining the possibilities by our current perspective, guidance should be developed for encouraging and evaluating Innovation and Design credits that can benefit the industry with additional sources of information and new approaches to materials evaluation.

Table 1-1: Ranking of Alternative Materials Studied by Adverse Human Health Impacts

In this table, there are three entries for each alternative material, representing the high, mid range, and low estimates of impact, ranked with worst among those studied at the top. Note that only selected alternatives were studied for each Product Group or Application; therefore, *the results only can be used to assess the ranking of PVC relative to the studied alternatives, not to identify "best" or "worst" materials among all possible alternatives in any category.*

Product group	Cancer Only			All Human Health		
	Cradle thru Use	**+End of life**	**+Occupational**	**Cradle thru Use**	**+End of life**	**+Occupational**
Window Frames (PVC plus 2 alternatives included in study)	1. Alternative A high, Alternative A avg, Alternative A low, ***PVC high*** 5. Alternative B high Alternative B avg Alternative B low ***PVC avg*** ***PVC low***	1. ***PVC high*** 2. ***PVC avg*** 3. Alternative A high, Alternative A avg, ***PVC low*** 6. Alternative A low 7. Alternative B high, Alternative B avg, Alternative B low	1. ***PVC high*** 2. ***PVC avg*** 3. Alternative A high, ***PVC low*** 5. Alternative A avg 6. Alternative A low 7. Alternative B high, Alternative B avg, Alternative B low	1. Alternative A high, Alternative A avg, Alternative A low 4. ***PVC high***, 5. ***PVC avg*** Alternative B high ***PVC low*** Alternative B avg Alternative B low	1. Alternative A high, Alternative A avg, Alternative A low 4. ***PVC high,*** ***PVC avg*** ***PVC low*** Alternative B high Alternative B avg Alternative B low	1. ***PVC high,*** Alternative A high, Alternative A avg, Alternative A low 5. ***PVC avg*** 6. ***PVC low*** 7. Alternative B high Alternative B avg Alternative B low
Pipe (drain-waste-vent) (PVC plus 2 alternatives included in study)	1. Alternative C high, ***PVC high*** 3. Alternative C avg 4. Alternative C low 5. ***PVC pipe avg*** 6. ***PVC pipe low,*** Alternative D high, Alternative D avg, Alternative D low	1. ***PVC high*** 2. ***PVC avg*** 3. ***PVC low*** 4. Alternative C high 5. Alternative C avg 6. Alternative C low 7. Alternative D high Alternative D avg, Alternative D low	1. ***PVC high*** 2. ***PVC avg*** 3. Alternative C high 4. ***PVC low*** 5. Alternative C avg, Alternative D high 7. Alternative D avg 8. Alternative C low 9. Alternative D low	1. Alternative C high 2. Alternative C avg 3. Alternative C low, ***PVC high*** 5. ***PVC avg,*** Alternative D high, Alternative D avg, Alternative D low 9. ***PVC low***	1. Alternative C high, ***PVC high*** 3. Alternative C avg, ***PVC avg*** 5. Alternative C low, Alternative D high, Alternative D avg, Alternative D low, ***PVC low***	1. ***PVC high,*** Alternative C high 3. Alternative C avg, ***PVC avg*** 5. Alternative C low, Alternative D high, Alternative D avg 8. Alternative D low, ***PVC low***
Siding (Vinyl plus 3 alternatives included in study)	1. Alternative E high, ***PVC high,*** Alternative E avg 2. Alternative F high, Alternative G high, Alternative E low 7. Alternative F avg, Alternative G avg 9. Alternative G low, Alternative F low *11.**PVC avg*** ***12. PVC low***	1. ***PVC high*** 2. ***PVC avg*** 3. ***PVC low,*** Alternative E high, Alternative G high, Alternative E avg 7. Alternative G avg 8. Alternative F high 9. Alternative E low, Alternative F avg. 11. Alternative G low 12. Alternative F low	1. ***PVC high*** 2. ***PVC avg*** 3. ***PVC low*** 4. Alternative E high 5. Alternative E avg 6. Alternative G high 7. Alternative G avg 8. Alternative F high 9. Alternative E low, Alternative F avg 11. Alternative G low 12. Alternative F low	1. Alternative E high 2. Alternative E avg, Alternative F high, Alternative F avg, Alternative G high, Alternative F low, Alternative E low 8. ***PVC high,*** ***PVC avg,*** ***PVC low,*** 11. Alternative G avg 12. Alternative G low	1. Alternative E high, ***PVC high*** 3. Alternative E avg, Alternative G high, ***PVC avg,*** Alternative F high, Alternative F avg, Alternative F low, Alternative E low, 10. ***PVC low,*** Alternative G avg, 12. Alternative G low	1. ***PVC high,*** Alternative E high 3. Alternative E avg, ***PVC avg,*** Alternative G high, Alternative F high, Alternative F avg, Alternative F low, Alternative E low 10. ***PVC low*** 11. Alternative G avg, 12.Alternative G low
Resilient Flooring (2 Vinyl plus 2 alternatives included in study)	1. ***Sheet Vinyl high*** 2. ***VCT high*** 3. ***Sheet Vinyl avg,*** ***VCT avg*** 5. Alternative H high, ***Sheet Vinyl low***, Alternative I high ***8. VCT low*** 9. Alternative H avg 10. Alternative I avg 11. Alternative H low 12. Alternative I low	1. ***Sheet Vinyl high*** 2. ***Sheet Vinyl avg,*** ***VCT high*** 4. ***VCT avg*** ***5. Sheet Vinyl low,*** ***6. VCT low*** 7. Alternative H high 8. Alternative H avg 9. Alternative I high 10. Alternative I avg 11. Alternative H low 12. Alternative I low	1. ***Sheet Vinyl high*** 2. ***VCT high,*** ***Sheet Vinyl avg*** 4. ***VCT avg*** 5. ***Sheet Vinyl low*** 6. ***VCT low*** 7. Alternative H high, Alternative I high 9. Alternative I avg, Alternative H avg 11. Alternative H low 12. Alternative I low	1. ***VCT high*** 2. ***VCT avg,*** Alternative H high, ***Sheet Vinyl high*** 5. ***Sheet Vinyl avg,*** Alternative H avg 7. ***VCT low*** 8. ***Sheet Vinyl low,*** Alternative H low 10. Alternative I high 11. Alternative I avg 12.Alternative I low	1. ***VCT high*** 2. ***Sheet Vinyl high*** 3. ***VCT avg,*** ***Sheet Vinyl avg*** 5. Alternative H high 6. Alternative H avg 7. ***VCT low*** 8. ***Sheet Vinyl low*** 9. Alternative H low 10. Alternative I high 11. Alternative I avg 12. Alternative I low	1. ***VCT high*** 2. ***Sheet Vinyl high*** 3. ***VCT avg,*** ***Sheet Vinyl avg*** 5. Alternative H high 6. Alternative H avg 7. ***VCT low,*** ***Sheet Vinyl low*** 9. Alternative H low 10. Alternative I high 11. Alternative I avg 12. Alternative I low

Cradle through Use: health impacts through environmental pathways from life cycle inventory
+End of Life: adds health impacts from end of life disposal including backyard burning, landfill fires, incineration
+Occupational: adds occupational impacts including installation

*Table 1-2: Rankings of Alternative Materials Studied for Environmental Impacts**

In this table, there are three entries for each alternative material, representing the high, mid range, and low estimates of impact, with worst among those studied at the top. Note that only selected alternatives were studied for each Product Group or Application; therefore, *the results only can be used to assess the ranking of PVC products relative to the studied alternatives, not to identify "best" or "worst" performing materials among all possible alternatives in any category*. Numbers in parentheses refer to impact categories noted in footnote to the table.

Product group	Cradle through use	Add end-of-life with burning
Window Frames (PVC plus 2 alternatives included in study)	1. Alternative A high (1,1,1,1,1,1,1) Alternative A avg (2,2,2,2,2,2,2) Alternative A low (3,3,3,3,3,3,3) 4. Alternative B high (5,4,5,4,4,7,5) Alternative B avg (7,6,7,6,8,8,7) ***PVC high*** (4,7,4,7,5,4,4) ***PVC avg*** (6,8,6,8,6,5,6) Alternative B low (9,6,9,6,9,9,9) ***PVC low*** (8,9,8,9,7,6,8)	1. Alternative A high (1,1,1,1,1,1,1) Alternative A avg (2,2,2,2,2,2,2) Alternative A low (3.3.3.3.3.3.3) 4. Alternative B high (6,4,4,4,4,4,7) Alternative B avg (7,6,5,6,5,8,8) ***PVC high*** (4,5,7,5,7,5,5) Alternative B low (9,8,6,8,6,9,9) ***PVC avg*** (5,7,8,7,8,6,5) ***PVC low*** (8,9,9,9,9,7,6)
Pipe (drain-waste-vent) (PVC plus 2 alternatives included in study)	1. Alternative C high (1,1,1,1,1,1,1) Alternative C avg, 2,2,2,2,2,2,2) Alternative C low (3,3,3,3,3,3,3) 4**. *PVC high,*** (4,7,4,7,4,7,4) Alternative D high, (7,4,7,4,6,4,5) Alternative D avg, (8,5,8,5,7,5,6) Alternative D low, (9,6,9,6,8,6,7) ***PVC avg,*** (5,8,5,8,5,8,8) ***PVC low*** (6,9,6,9,9,9,9)	1. Alternative C high, (1,3,1,1,1,1,1) Alternative C avg, (2,4,2,2,2,2,2) Alternative C low (3,4,3,3,5,2,3) 4. ***PVC high,***(4,6,4,4,3,6,4) ***PVC avg,*** (5,8,5,5,4,7,5) Alternative D high, (7,1,7,6,7,3,7) Alternative D avg, (8,2,8,8,7,4,8) Alternative D low, (9,7,9,9,8,5,9) ***PVC low*** (6,9,6,6,9,8,6)
Siding (Vinyl plus 3 alternatives included in study)	1. Alternative E high (1,1,1,1,1,3,1) 2. Alternative E avg (2,4,10,3,2,7,2) Alternative G high (6,2,5,2,6,5,4) Alternative F high (3,6,2,6,3,1,7) Alternative F avg (5,7,3,7,4,2,8) Alternative F low (7,8,4,8,5,4,9) 7. Alternative G avg (9,3,7,4,11,6,5) Alternative E low (11,9,12,9,7,12,3) ***PVC high*** (4,10,6,10,8,8,10) ***PVC avg*** (8,11,9,11,9,10,11) Alternative G low (12,5,7,5,12,9,6) ***PVC low*** (10,12,11,12,10,11,12)	1. Alternative E high (1,1,2,1,1,2,1) 2. Alternative E avg (3,4,10,3,2,7,2) Alternative G high(7,2,1,2,6,5,4) Alternative F high(4,6,3,6,3,1,8) Alternative F avg, (5,7,4,7,4,3,9) Alternative F low(6,8,6,8,5,4,10) Alternative G avg(9,3,5,4,11,6,5) ***PVC high***(2,10,7,10,8,8,7) Alternative E low (11,9,12,9,7,12,3) ***PVC avg***(8,11,9,11,9,10,11) Alternative G low (12,5,8,5,12,9,6) ***PVC low*** (10,12,11,12,10,11,12)
Resilient Flooring (2 Vinyl plus 2 alternatives included in study)	1. ***Sheet Vinyl high,*** (1,4,1,1,1,2,1) ***Sheet Vinyl avg,*** (2,5,2,2,2,3,2) ***VCT high,*** (5,8,5,3,7,6,5) Alternative H high (6,1,4,6,6,1,4) 5. ***Sheet Vinyl low,***(5,8,5,3,7,6,5) Alternative H avg, (7,2,7,8,8,4,7) ***VCT avg*** (4,7,6,5,4,12,6) 8. ***VCT low*** (8,9,9,10,5,7,9) 9. Alternative H low (9,3,8,12,9,8,8) 10. Alternative I high (10,10,10,7,10,9,10) 11. Alternative I avg (11,11,11,9,11,10,11) 12.Alternative I low (12,12,12,11,12,11,12)	1. ***Sheet Vinyl high,*** (1,4,1,1,1,2,2) ***Sheet Vinyl avg,*** (2,5,2,2,2,3,3) ***VCT high,*** (3,6,3,4,3,5,1) Alternative H high, (6,1,4,6,5,1,7) ***Sheet Vinyl low*** (5,9,5,3,8,6,5) 6. Alternative H avg, (7,2,7,8,7,4,8) ***VCT avg*** (2,5,2,2,2,3,3) 8. ***VCT low*** (5,9,5,3,8,6,5) 9. Alternative H low (9,3,8,12,9,11,11) 10. Alternative I high (10,8,10,7,10,7,9) 11. Alternative I avg (11,11,11,9,11,8,10) 12.Alternative I low (12,12,12,11,12,12,12)

* Note: rankings with respect to each separate environmental impact category are presented in parenthesis, with the following impact category order: acidification, eutrophication, smog, ozone depletion, global climate change, fossil fuel depletion, ecotoxicity). The materials appear in this table in the order of their average normalized performance across the seven impact categories. The order of appearance does *not* indicate an overall environmental score, since such an overall score would require value-based weighting across the impact categories.

1

2 Introduction

This report has been prepared in response to a charge to the LEED® Technical and Scientific Advisory Committee (TSAC) from the LEED Steering Committee to evaluate the technical basis for a PVC-related materials credit. In the year 2000 during the process of developing the LEED for Commercial Interiors rating system, draft credit language was developed with a requirement to avoid PVC materials as well as chemicals listed in the *Annual Report on Carcinogens*, IARC *Monographs*, or OSHA Toxic and Hazardous Substances. The U.S. Green Building Council decided that further technical knowledge was needed to determine the soundness of this requirement for PVC. Pending resolution of this issue, all related Credit Interpretation Rulings and potential Innovation and Design credits were put on hold.

This report was supported by an online database for those seeking further information. The database was developed by the PVC Task Group as a way to organize its research for this report and to ensure maximum transparency of its information sources and methods. The Task Group is available to work with the Steering Committee to assist in a full understanding of the issues presented in this report and is willing to present on the reasons for its recommendations.

2.1 Task Group Charge

The U.S. Green Building Council's Technical and Scientific Advisory Committee charged its PVC Task Group with the following:

> "The Task Group is charged with reviewing the evidence offered by stakeholders and independent sources, and advising the LEED Steering Committee on the availability and quality of evidence as a basis for a reasoned decision about the inclusion of a PVC-related credit in the LEED rating system."

2.2 Background

The formation of the Task Group was precipitated by a series of events beginning in 1999. In December of that year, the LEED Commercial Interiors Committee initiated credit language relating to PVC. The initial language, which appeared in a February 2000 un-balloted draft Commercial Interiors LEED rating system as Materials and Resources (MR) Credit 9, read as follows:

Materials Credit 9: Alternative Materials	INTENT: Reduce use of products containing toxic and/or hazardous substances and encourage use of comparable alternatives. REQUIREMENT: • Eliminate the use of virgin PVC • Eliminate the use of any chemical listed in the OSHA Toxic & Hazardous Substances. TECHNOLOGIES/STRATEGIES: Require Material Safety Data Sheets (MSDS) for all products specified.

It was modified soon after for the March 2000 draft to read as follows:

Materials Credit 9: Alternative Materials	INTENT: Reduce use of products containing toxic and/or hazardous substances and encourage use of comparable alternatives. REQUIREMENT: • Eliminate the use of virgin PVC • Eliminate the use of any chemical listed in the National Toxicology Program (NTP), "Annual Report on Carcinogens"; the International Agency for Research on Cancer (IARC) "Monographs"; or 29 CFR 1910, subpart Z, OSHA Toxic & Hazardous Substances. TECHNOLOGIES/STRATEGIES: Require Material Safety Data Sheets (MSDS) for all products specified.

In November 2000 a meeting was held in Washington, DC between the LEED-CI Committee and interested parties. At this meeting a full discussion of the issues surrounding this "no-PVC credit" took place and the matter was referred to what was to become the TSAC, a LEED committee established by the USGBC Board of Directors to support the LEED Steering Committee in dealing with complex technical issues. In November 2002, TSAC chose Scot Horst to chair the PVC Task Group and by March 2003 the Task Group was selected and began work.

The Task Group was chosen to represent expertise in particular areas that were necessary to accomplish the charge. Individuals with experience and a knowledge base in LCA, risk assessment, green building techniques, the LEED rating system, materials alternatives and life cycle impact assessment were selected. Each of the Task Group members was approved by the USGBC Board of Directors. The Task Group includes the following members:

Scot Horst, Chair	The Athena Institute
Kara Altshuler, PhD	Private Environmental Consultant/Regulatory Toxicologist
Nadav Malin	BuildingGreen, Inc.
Greg Norris, PhD	Harvard University School of Public Health, Sylvatica

The Task Group was assisted greatly by Yurika Nishioka, Ph.D, Harvard University School of Public Health. Brief biographies of the Task Group members are presented in Appendix L.

2.3 Objectives

The following objectives of the USGBC's TSAC and PVC Task Group are relevant to the purpose of this report:

Task Group: The objective of the Task Group was to provide a report to TSAC that presented a clear picture of the current state of knowledge on this issue to inform the decision making process for the LEED Steering Committee.

TSAC: It is the objective of TSAC as per its charter to ensure the technical soundness of LEED, to serve as a scientific advisory committee to the Council for LEED, and to review difficult or controversial technical issues.

2.4 Scope

Characterizing the environmental pros and cons relating to PVC building materials and competing alternative materials is a mammoth undertaking. Thus, the Task Group was forced to make numerous assumptions and limit the scope of its work in several ways. A few of the more significant limitations are discussed in the sections that follow.

2.4.1 Past manufacturing practices

Past manufacturing practices are not reviewed in this report. An example might include worker exposure to vinyl chloride monomer (VCM) in PVC manufacturing. Even though workers were exposed to VCM extensively in the past, it is now highly controlled in PVC facilities in the United States. In order to characterize the potential influence of credits related to material selection in the near-term and future, current rather than historical manufacturing practices are considered.

2.4.2 Foreign manufacturing practices

With the exception of information already integrated into life cycle inventory databases and other source materials, foreign manufacturers were not researched for this report. Very little information is available from foreign production facilities. While the Task Group recognizes that materials from foreign production facilities are purchased and used in buildings in the U.S, it was not possible for the Task Group to obtain enough specific information about issues such as risk and production methods to analyze them for this report.

2.4.3 Materials

PVC is widely used in buildings, for many different applications. It was not feasible to study both the PVC-based materials and competing materials for all of these applications, so a set of four representative applications were selected for analysis. These are siding, drain/waste/vent pipe, resilient flooring, and window frames. Potential implications of this streamlining measure are discussed below in Section 3.5 on materials studied.

2.4.4 Morbidity associated with PVC and competing building materials

There has been substantial concern about potential morbidity in humans exposed to the toxic compounds used in the manufacture of both PVC and non-PVC building materials. A challenge posed by the broad concept of morbidity, defined as “the presence or incidence of disease,” is that such impacts range widely in severity from minor irritation to paralysis. For example, many of the compounds used to manufacture building materials cause morbidity: volatile organic compounds result in headaches, nausea, burning eyes and nasal tissues; compounds used in the manufacture of fiber-cement siding and sand molds for cast iron pipes can cause respiratory diseases including silicosis; compounds used to make both PVC and ABS resin are carcinogenic; and metal smelting, in the manufacture of aluminum sheet and steel used in window frames, is associated with respiratory diseases and cancers of the lung and bladder. Because different pollutants are associated with morbidity impacts that can vary widely in severity, comparisons of the risks of “morbidity impacts” are not very meaningful. This report follows the LCA convention of aggregating morbidity impacts with a composite “hazard index”. This hazard index cannot be directly compared with the mortality risks – risk of premature death – associated with onset of cancer, inhalation of particulates, and other impact pathways.

2.5 Transparency

2.5.1 Ensuring transparency

There are several aspects of transparency that relate to this report: transparency of process, of methodology, and of information sources. Transparency of process has been determined by TSAC through the use of its nine-step "TSAC Review Procedures for Specific Issues."

The nine steps are outlined in the procedures as follows:

1. Define the charge
2. Form a Task Group (TG)
3. Solicit stakeholders
4. Solicit written input from stakeholders
5. Review stakeholder submissions
6. Synthesize information and prepare draft report
7. Solicit stakeholder comment on draft report
8. Prepare final report and recommendations
9. Develop USGBC position

Stakeholders were solicited to respond to several documents posted on the USGBC website. They were also invited to respond to the TG's methodology in a meeting that was held in Washington, DC on February 18, 2004. Written responses to the methodology and additional reports for the TG to consider and include in its list of sources were solicited until April 2, 2004. A draft report was prepared and distributed in December 2004. Comments were solicited and accepted until February 16, 2005; further assistance on several topics was solicited and accepted until July 8, 2005. In response to those comments, the TG has undertaken additional research and prepared this final report, representing step eight in TSAC's nine-step process.

Transparency of methodology is achieved in part through the presentations and documents that were shared publicly as part of the nine-step process, and in part with the full disclosure of methods and assumptions provided in this report. In addition, spreadsheets that were used to calculate the quantitative results are available for download from the USGBC Web site.

Transparency of source data used in this effort is accomplished through the publicly available online database, which lists all the sources used to develop the initial public comment draft, and Appendix M, which lists all the sources cited in this report.

2.5.2 How stakeholder comments on the draft report were used to inform this report

In response to the first public draft of this report, stakeholders submitted 562 comments, many of which were substantive and constructive. All of these comments are viewable online at http://pvc.usgbc.org/comments.php.

Stakeholder input on the first draft stimulated a second round of work by TSAC, including a reevaluation of methodology, analysis scope, analyses conducted, sources of data, and scope of recommendations and conclusions. While it was not feasible to respond to each comment individually, key themes from the comments were collected and addressed in a document entitled "Stakeholder Comment Themes" that was distributed on August 25, 2005. That document

outlined the actions that TSAC intended to take in response to each theme or concern. This section of the report summarizes and updates those responses.

We have not included as a "theme" comments that are supportive of the draft report, although those were also considered in the revisions process, as evidenced by the fact that the fundamental approach, of quantitative analysis that integrates impacts over the life cycles of functionally comparable product alternatives, addressing both environmental and human health risks, was retained.

The input themes addressed here are as follows, sorted roughly in order from more general to more specific:

1) Influence of report on LEED and momentum for market transformation
2) Questions about methodology (TG's hybrid of LCA and risk assessment), in terms of the scope of impacts it is able to address, its transparency of method and data, its complexity, its relation to precautionary principles, general validity in light of the fact of its novelty, and the need for peer review.
3) Concerns about whether and how the "data gap" issues are adequately accounted for in the conclusions and recommendations.
4) The need to explicitly address the Stockholm convention, persistent organic pollutants (POPs), and persistent, bioaccumulative and toxic substances (PBTs).
5) Recyclability and end-of-life fate of PVC
6) Fence line exposures for residents near PVC manufacturing facilities
7) Role of fires (building, backyard burning, other) in life cycles of PVC building materials
8) Stakeholder input on data sources and assumptions covering several areas including chemicals considered under the analyses, product composition and lifetime, etc.

Comment theme 1. The conclusions in draft report are damaging to LEED's leadership position and to the momentum behind market transformation.

TSAC takes the report's potential influence on this issue *very* seriously, and concurs that the draft report ran this risk by focusing exclusively on the issue of whether or not avoiding a certain material can be justified. TSAC addressed this issue directly in its re-analysis and re-write, in a number of ways. Principal among them is the addition of suggestions for ways that comprehensive and quantitative information may be used to *support and encourage* the use of preferable materials, instead of simply avoiding those perceived to be less desirable.

A number of comments suggested that because (in the opinion of the commenter) there are known to be preferable alternatives to PVC for any given application, LEED should include a credit for avoiding the use of PVC. TSAC's position is that, even if one assumes that there are better alternatives, simply crediting the avoidance of PVC does not provide an incentive for using those better alternatives as opposed to other alternatives that might not be better. If the next available material is associated with higher exposure of harmful toxicants the substitution away from PVC can potentially pose higher overall health and ecological impacts. It follows that if PVC is not consistently among the worst option for its common applications, such a credit could readily become an incentive to use something worse, which would not represent positive market transformation.

Therefore, TSAC has undertaken to determine whether or not PVC (or any other material) is consistently among the worst options, and recommends incentivizing the use of preferable materials, in the interest of providing an effective leadership position.

Comment theme 2. The validity and wisdom of using this "new" method combining LCA and risk assessment is questionable.

TSAC's approach to its charge includes the use of a variety of analytical tools, principal among them being life cycle assessment and risk assessment.

Stakeholders raised a wide variety of concerns about the use of these two methods. These concerns included:

- The scope of impacts that this pair of methodologies is able to address. For example, a stakeholder questioned whether the methods could address the impact of "lead and other PVC additives, air and groundwater contamination around PVC manufacturing facilities, dioxin generation from landfill fires, and recyclability."
- Transparency and complexity of methodology;
- Transparency of data and data sources;
- The general validity of the analysis in light of its novelty; and
- The need for peer review of the method and report.

In place of TSAC's central methodology, several stakeholders joined in recommending that TSAC apply hazard analysis, screening methods based on principles such as sustainability, and precautionary approaches.

TSAC took this full and broad set of concerns very seriously, and addressed the full scope of this input in a number of ways.

First, we returned to basic principles and goals, we questioned our use of LCA and risk assessment, and we considered roles for hazard assessment, screening methods, and precautionary approaches. In a nutshell, we concluded that the following two principles needed to be at the center of our analysis: comprehensiveness and quantitative methods.

Comprehensiveness means that we could not in good conscience ignore any stages of the life cycle for any alternatives. We have strived to rectify omissions that stakeholders noted in the first report, including any emissions of pollutants for which emission factor data was initially not present in the available LCA databases (e.g. emissions of mercury and dioxins from some processes) and impacts of end-of-life fates of the building materials studied (routine as well as accidental, formal and informal). Comprehensiveness means that we cannot ignore exposures of workers in manufacturing facilities, exposures of people in communities living near manufacturing facilities, and exposures of the wider public from releases to the ambient environment.

Comprehensiveness means that we cannot exclude any categories of pollutant release. Comprehensiveness also means that we cannot exclude environmental or human health impact categories or pathways that have been deemed to be worth considering by the broad and historical community of stakeholders who have been considering comprehensive environmental and health assessments of products during the past three decades.

Why do we also insist on applying quantitative methods? For example, why not use the kind of hazard assessment which identifies the existence of a chemical release or a potential hazard, without quantifying it? Such analyses, for example, come to conclusions such as "The manufacture of product X is associated with chemical Y." This kind of approach is also the basis

for some popular screening methods. If a priority chemical is present in a product, the product fails to pass the screen.

TSAC reconsidered hazard assessment and this type of screening method. We affirmed the need to include, but go beyond, the considerations of these methods, as follows:

- The presence of a hazardous chemical in a product is of concern, but so are emissions of chemicals used in the production of the product, even if the chemicals or pollutants do not end up in the final product. Indeed, ultimately what is important is the potential for an adverse impact to the environment or human health.
- The exposure (actual or potential) to chemicals caused by the manufacture of a product is of concern, but so are exposures (actual or potential) from other processes in the product life cycle. These include processes in the supply chain, such as the production of additives, basic materials, the extraction and refining of fuels, the generation of electricity, transportation of heavy materials over long distances, and so on.

Now, once a comprehensive approach has widened our focus to include all chemicals of concern to all stakeholder groups over the full life cycle, we find that a simple hazard assessment, without quantitative thresholds, becomes rather pointless. Every product's life cycle is "associated with" POP/PBT emissions, either directly or indirectly through its supply chain. Even though we may all agree that any amount of POP emission (or climate change, or ozone depletion, etc.) is morally unacceptable, we are stuck with the need to ask "how much?" Because everything is connected in the economy as well as the environment, we cannot switch immediately to products with no negative impacts; rather, a leadership position would be to find and promote products whose life cycle impacts are much better than those of other products, and stimulate the shift towards products with net positive impacts. Such promotion requires a quantitative approach. It requires analyses which do their best not only to identify connections, but to go further, asking and doing their best to answer the question: "how much?"

A precautionary stance in relation to environmental and health impacts means that we must consider not only known impacts, but potential impacts as well: risks. And a quantitative approach to risk brings us back to risk assessment.

And so we arrive at the next set of stakeholder concerns that were raised about TSAC's risk-plus-life-cycle approach: transparency and complexity. TSAC is in total agreement that the analysis, and its data sources, must all be open for scrutiny and cross-examination, cross-evaluation, by the widest possible stakeholder community, including academic researchers, industry representatives, non-governmental organizations, and any other interested representatives of stakeholder interests.

There are two separate points here, both important: (1) making the transparency achievable, and (2) making the complex analysis widely accessible.

For transparency, TSAC has publicly released all spreadsheets used in (risk) analyses. In addition, the life cycle assessment modeling and data are being made available for download in the form of an interactive and no-cost software tool that will enable interested users to examine the assumptions and data in detail. To make the complex analyses more accessible, we have modified the body of the report to make it shorter and more readable, while moving many of the technical details to appendices.

Finally, there are the questions of novelty and the need for peer review. TSAC welcomed the call for peer review. The TG actively sought out peer review of the revised draft report in an effort to verify validity of methodology and to gain constructive criticism of the approaches used.

Sections of the report have undergone a peer-review process by recognized experts in the appropriate fields. The sections on phthalate toxicity and exposure were reviewed by Dr. Russ Hauser from Harvard School of Public Health, and Dr. Thør Larsen from National Institute of Occupational Health, Denmark. The section on fenceline risk was reviewed by Dr. Jon Levy from Harvard School of Public Health. Analyses estimating dioxin emissions were reviewed by Dr. Ulrich Quass from Müller-BBM GmbH, Germany. Mercury exposure was reviewed by Dr. Glenn Rice from Harvard School of Public Health. In some cases, comments from the peer review resulted in additional analyses; these are discussed in the appropriate appendices.

On the question of novelty, TSAC concurs that it is trying something new. Comprehensiveness leads to a life cycle-based approach, but as many stakeholders pointed out, conventional LCA alone neglects some kinds of risk, from such potential sources as accidents, fires, and spills; it also tends to neglect site-specific impacts of manufacturing facilities and fence line exposures. To address these gaps in a quantitative way we apply risk assessment where possible. As mentioned before, the combination of tools does not ignore, but rather includes and is forced to go beyond, hazard assessment and screening methods, because of its comprehensiveness. It is far from perfect, as are all analyses. TSAC has worked to make it as valuable and accessible as possible.

Comment theme 3. "Data gap" issues, that are not included in the quantitative analysis don't appear to be accounted for in the conclusions and recommendations?

The draft report included discussions and analyses of a number of important issues that could not be incorporated into the integrated LCA/risk assessment quantitative analysis for lack of appropriate data. As noted above, the approach of combining risk assessment and LCA was developed with the goal of generating quantitative results that account for as broad a range of issues as possible. However, usable data sets are not available for all issues of concern, so there are a number of issues that could not be factored into the quantitative analysis.

TSAC addressed this concern in several ways. First, additional data were collected and analyzed so that some issues, such as emissions from landfill fires and mercury releases, are better represented in the integrated quantitative analysis, thereby reducing the number of data gaps. Second, the narratives in the report that discuss these data gaps were revised based on numerous specific stakeholder comments, so that they do a better job of characterizing the issues that cannot be factored into the quantitative results.

Comment theme 4. The need to address Stockholm convention, POPs, and PBTs.

An example of this thread of input is the following quote of stakeholder input:
The draft report "… ignores the international consensus to end the production of the worst of these particularly potent chemicals [PBTs], including dioxin and three others which are uniquely associated with the entire life cycle of PVC."

TSAC addressed this concern in several different ways, which fall under two main categories. The first was to address issues around data (especially data blind spots), and the second was to be sure that a comprehensive and explicit analysis of persistent organic pollutants (POPs) and persistent bioaccumulative toxins (PBTs) was conducted for each alternative.

A few comments specifically noted the lack of reference to the Stockholm Convention in the draft report. This convention targets 12 POP compounds or groups of compounds, classified as "intentionally produced" and "unintentionally produced" chemicals. Ten of these are intentionally produced—nine pesticides and one industrial chemical. The two unintentionally produced classes of compounds, dioxins and furans, were addressed extensively in TSAC's draft

report, and researched further for this final report. The general approach represented in the Stockholm Convention, of targeting certain compounds for elimination, applies readily to intentionally produced compounds, but none of the materials considered in the TG's report are directly associated with the ten Stockholm chemicals in that category. We therefore focused our efforts in the final report on addressing the Stockholm compounds that are widely produced unintentionally.

Taking the data issue first, we recognized that quantitative analyses can be no better than their data basis, and we further recognized that the publicly available life cycle assessment databases which are inputs to our analysis needed to be checked and compared with other independent sources to address data gaps. We therefore investigated and summarized alternative sources (including those suggested by stakeholders) of data on emissions of the full set of Stockholm-relevant POPs and PBTs, and augmented our LCA databases with the additional information wherever there is evidence to support doing so. This was done for all stages of the life cycle, from raw materials acquisition through primary and secondary manufacturing, use, disposal, and end-of-product-life—including the potential influence of fires (discussed separately below). And we have attempted to address the data gap issue on POPs and PBTs comprehensively across the full set of product alternatives and life cycle stages. The development of emissions factors for POPs and PBTs is discussed in detail in Appendix D: Life Cycle Assessment Emission Factors.

As in the draft report, where data gaps remain, we have attempted to flag these prominently, and to consider anew the influence of these gaps on conclusions.

The second way that TSAC attempted to address the topic of POPs and PBTs head-on was by creating and using an explicit set of impact categories for these chemicals. That is, once we had expanded the data sources and augmented the data basis of the analysis accordingly, we next created two additional impact categories (one for human cancer, and one for ecotoxicity) that address and total the impacts of strictly the chemicals named by the Stockholm convention. These impact methods take into account the often major differences in potency of health impact and expected probability of exposure among the Stockholm chemicals, from factors such as persistence and bio-accumulation. The total mortality risk from Stockholm chemicals via cancer can then be separately compared as well as aggregated with total human morality risks over the product life cycles, and ecotoxicity impacts of Stockholm chemicals can also be compared with total ecotoxity impacts. The impact assessment methods used in our analysis are described in Appendix E: Life Cycle Impact Assessment Characterization of Human Health Impacts.

Comment theme 5. Recyclability and the end-of-life fates of PVC were not addressed.

Many stakeholders pointed out that the draft report ignored the disposal phase in the life cycle of PVC and alternative materials. TSAC agreed with this concern and has added an evaluation of impacts from the disposal phase for each material. Previously, we depended on LCA databases that focused only on the upstream, manufacturing impacts. For our new analysis we have researched databases and literature that include emissions and impacts during disposal.

As for the modes of disposal, some stakeholders pointed out that TSAC did not address the challenges of recycling and the fact that many industries are moving away from PVC because of that concern. They also argued that PVC is an inherently unsustainable material because non-recycled PVC waste is landfilled, sent to an incinerator, or burned in backyard barrels. On the other hand, stakeholders from the PVC industry argued that all vinyl products can be recycled and made into new products, and called into question if resilient flooring and vinyl siding is burned in backyard burning barrels. TSAC sought information on actual end-of-life fates for the

product groups and materials studied in this report. Some data and some anecdotal information was provided in response to a follow-up solicitation to stakeholders using a specially arranged, online "Outreach Forum". TSAC has incorporated the information into the LCA modeling, differentiating the impacts of the disposal practices based on the proportion of PVC and alternative products that are either sent to landfill, incinerated or recycled. In the same manner, TSAC sought information regarding the amount of waste that is burned in the backyard, and included that data in the revised analysis.

Comment theme 6. Fenceline exposures and risks to people living near manufacturing facilities were not adequately considered.

TSAC received an extensive set of comments on the fenceline exposure section of the draft report. We reexamined air monitoring data from Kentucky and Louisiana, as well as tumor registry data. For the revised report, we have updated the Kentucky monitoring data to include the most recent analyses; further, we have included cancer risk estimates based on recent monitoring data for vinyl chloride that are available for both Kentucky and Louisiana. Corresponding analyses for the other building materials were not performed due to lack of exposure data.

Regarding the tumor registry data, we agree that parish-level data may not capture locally elevated cancer incidences. We undertook additional research to determine what additional data may be available that can be incorporated into the risk assessment portion of our analysis. The results of this additional research are presented in Appendix I.

Comment theme 7. The role of fires, including building fires and other accidental fires in the life cycles of building materials, was not adequately considered.

Numerous comments argued that the draft report failed to adequately address risks from accidental fires, including fires in occupied buildings (with risks to occupants and firefighters), fires in landfills, and fires at PVC manufacturing and distribution facilities. TSAC extended its research significantly, following up on citations provided by stakeholders and on other sources to better incorporate those issues into this revised analysis. The development of emission factor estimates for these processes is described in Appendix D.

Comment theme 8. Several assumptions about compounds used in vinyl, the lifetime of alternative building materials (e.g., cork flooring) and types of building materials (e.g., Swiss window) should be changed based on provided information.

TSAC reviewed information provided by stakeholders to revise previous estimates, both in the LCA and risk assessment sections of the report. Prior to using that information, however, we worked to confirm the accuracy of the suggestions made by the stakeholders and to find more appropriate information in order to derive more accurate results to the analyses. In other instances, additional data on toxicity of compounds were provided or citations were recommended for inclusion in the report. TSAC reviewed those suggestions and made appropriate changes to the report. For example, TSAC re-did the entire window analysis, replacing its use of data on European windows with data for North American windows.

2.6 Report Structure

This report is intended to support the LEED Steering Committee in its effort to make a decision about "the inclusion of a PVC-related credit in the LEED rating system." The Task Group has performed an extensive literature search, sought and received stakeholder input, organized the literature and culled values when possible to present the Steering Committee with information to

inform its decisions. The report is organized in six main sections, supplemented by appendices and an online database.

The rest of the report is organized as follows:

Section 2 defines report scope. It includes an explanation of issues that are relevant to this report but fall outside the scope based on various limitations and the need to focus on support for a USGBC decision.

Section 3 defines methodology.

Section 4 presents findings from the integrated analysis of life cycle and risk related evidence. This section shows the results of life cycle assessments of each of the product alternatives based on several databases, and integrates these results with the results of occupational risk assessments for manufacturing, installation, and end-of-life phase of the product alternatives.

Section 5 presents findings from additional analyses that TSAC was not able to integrate into the combined LCA and risk assessment.

Section 6 contains discussion of data gaps that remain.

Section 7 is a summary of findings, conclusions and recommendations based on the evidence presented.

The topics of the appendices are as follows:

Appendix A: Acronyms

Appendix B: Database and Sources

Appendix C: Life cycle assessment assumptions

Appendix D: Life cycle assessment emission factors

Appendix E: Life cycle impact assessment characterization of human health impacts

Appendix F: Human Health Risk Assessment

Appendix G: Non-cancer and cancer risk estimates normalized per functional unit of building material

Appendix H: Additional Analyses: Risk Assessment of Phthalate Exposure

Appendix I: Additional Analyses: Air Monitoring Data and Fenceline Analysis

Appendix J: Peer Review

Appendix K: Sensitivity Analysis of Aluminum Windows

Appendix L: Task Group Biographies

Appendix M: Sources Cited

3 Methodology

The core elements of the approach taken by the Task Group for this work are outlined in the following sections. These items are described in further detail in corresponding appendices. This chapter describes:

- The ***database*** created for the study
- Data ***sources***
- The ***life cycle assessment*** framework
- ***Risk assessment***
- Identification of ***data gaps***
- ***Normalization***
- Definition of ***materials*** to be studied

3.1 Database

The Task Group created a relational database that organizes the available research literature in order to assess the quantity and quality of relevant evidence. The database was envisioned as a knowledge-mapping tool. It contains a list of documents or "Sources", each of which is linked to one or more cells in a matrix that represents the universe of information that may be needed to make a decision regarding a PVC-related credit. The database was used to develop the first draft of this report, but was not updated for this final report.

3.2 Sources

All sources were received as either stakeholder submissions or through an exhaustive literature search. Additional detail on data sources is presented in Appendix B.

3.2.1 Literature searches

The literature search was conducted using Medline, Pollution Abstracts, Online Computer Library Center (OCLC) and the Internet, in addition to the researchers' personal collections of reports. In the literature searches keywords were used to identify the materials of interest and some of their primary precursors. Searches were limited to English language articles.

The types of the selected articles fall into one the following categories:

- Risk Assessment – exposure and risk-related articles.
- Toxicological assessment – epidemiological studies, animal studies and discussion papers and comments based on toxicological studies.
- Emission study – those related to emissions from manufacturing sites as well as general air pollution study (i.e., ambient or indoor concentrations) related to the materials of our interest.
- Position paper – those that take one side of an argument.
- Communication/letter – letters and reports addressing various issues, presentation materials, newspaper and newsletter articles, as well as letters from the stakeholders.
- Comparative life cycle analysis – LCA studies of one or more materials of our interest.
- Life cycle inventory (LCI) data – Reports or databases containing life cycle inventory of products.
- Others – articles that do not fall into one of the above categories (e.g., an overview of the toxicological effects of chemicals), or those cover more than one category.

All sources that were referenced in the development of this report are listed in Appendix M.

3.2.2 Stakeholder submissions

The sources include a rich contribution of literature generously provided by stakeholders. In total, there are 892 stakeholder submissions, encompassing the full range of source types. For those submissions addressing a building material, the sources are overwhelmingly focused on PVC (more than 90%).

3.3 Life Cycle Assessment (LCA) Framework

As noted in section 2.5.2 on stakeholder input, our approach attempts to be comprehensive in terms of life cycle stages and impact categories, and it attempts to provide quantitative comparisons of different product/material alternatives that provide equivalent function in a building application. This section describes the life cycle methods and data underlying this approach. More detail on the assumptions and LCA methodology are provided in Appendix C.

A conceptual illustration of the method is provided in the figure below.

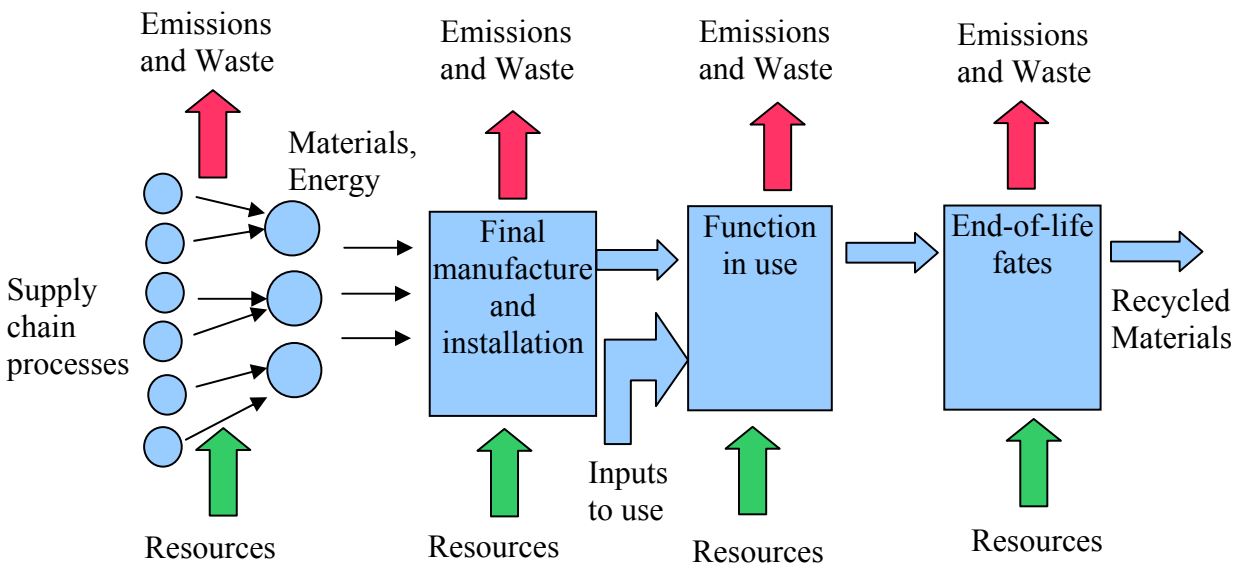

Figure 3-1: Scope of the life cycle studies in this report

3.3.1 Life cycle inventory analysis

We start (in the middle of the figure) by identifying the *function* provided by the product, such as covering 1 square foot of floor surface for 20 years. In this case, providing 20 square-foot-years of flooring would be what we refer to as the "functional unit." We then seek to compare different material/products that provide this same level of function.

During the use of the product, there may be natural resources consumed (such as water for washing) and emissions or wastes released (such as VOCs released from the flooring), Also, the use phase may require inputs of other produced materials and energy, such as cleaners, heating of water for washing, etc.

Now that the usage phase is described, we look back "upstream": what did it take to manufacture and install flooring that will cover 1 square foot for 20 years? If, for example, the flooring lasts an average of 10 years, then we will need 2 square feet of flooring to provide the functional unit, plus two applications of adhesive, etc. For data on the energy and material inputs to final

manufacturing and installation, data on durability, and data on emissions from final manufacturing and installation, our first choice was to use data published by NIST for its BEES model (NIST, 2002). The assumptions and data sources used to determine the functional unit, to model the use phase, and to model final manufacturing and installation, are described in Appendices C and D.

Next we need data for the whole supply chains that produce the material and energy inputs to final product manufacturing. This is the so-called "cradle-to-gate" portion of the life cycle model. In our study we rely on existing and publicly available databases to provide this information. Our preferred source of data for this step is the Swiss EcoInvent database (Ecoinvent Centre, 2005), due to its current data, public availability, extensive data quality assurance process, and its relatively high degree of thoroughness in relation to both the emissions and resource flows and the process types included. We also substitute data from Franklin Associates on electricity production in the U.S., wherever possible (Franklin Associates and Sylvatica, 1998).

The EcoInvent and Franklin databases are referred to as process-level life cycle inventory (LCI) databases. Process-level LCI databases contain models that describe production supply chains as an interconnected system of unit processes, linked by material or energy flows. In process-based LCI, these unit processes are generally at the level of individual engineering unit processes, such as manufacturing operations, fuel-specific power plants generating electricity, individual transportation legs by a specific mode, and so on. LCI databases contain information generally derived from samples of a number of similar manufacturing operations of a specific type and within a specific geographic region. For each unit process in the database, the data include quantities of each pollutant released to air, water, or land; inflows of raw materials and intermediate materials, and the resulting products.

The life cycle analysis tool SimaPro™, which contains the EcoInvent and Franklin databases mentioned above (as well as others), was used to calculate life cycle impacts of the upstream supply chains in our analysis (PRé Consultants, 2004).

We further augmented these publicly available LCI databases with emission factors for important pollutants that were not fully addressed in those databases, as described in detail in Appendix D. Life cycle inventory databases normally cover a wide range of emissions related to production of energy and materials associated with a final product. However, where some specific information about important processes or production and disposal practices in the U.S. was available, the existing life cycle inventories were modified or replaced based on our own analysis. For the production phase, emission factors were developed for:

- dioxin emissions in the production of PVC, aluminum, cast iron, and fiber cement,
- mercury emissions in the production of PVC, cast iron, and fiber cement, and
- VCM and EDC emissions in the production of PVC.

Next we add, within our integrated life cycle and risk assessment method, modeling of potential exposures of workers to pollutants within selected key processes in the supply chain as well as during final product manufacturing and (where relevant and possible) product installation. This step is described in more detail in section 3.4.2 and in Appendix F.

Finally, we turn to the product end-of-life phase. Modeling this step requires data on the end-of-life fates of the materials (how much is typically sent to landfill versus recycling versus incineration), and then we require data on the input requirements and emissions and resource flows associated with the end-of-life processes. For landfilling, recycling, and municipal

incineration, the LCI databases described above contain data on the input requirements and the emissions and wastes from these processes. Information on emissions for accidental landfill fires and backyard burning is very limited, however, since such accidental sources are traditionally omitted in conventional life cycle analyses, and the LCI databases mentioned above do not contain data for them. Given the lack of information in literature, but the potential importance of this life cycle phase in the total results, we developed methods to estimate these end-of-life impacts using available scientific knowledge in literature. In order to address the magnitude of uncertainties associated with the available scientific knowledge, we conducted an analysis to estimate the upper and lower boundaries for dioxin emission factors per unit of material disposal. The results are presented for a high model (upper boundary), low model (lower boundary), and average model.

In addition to dioxin emissions, we estimated emission factors for a number of pollutants such as particulates, oxides of nitrogen (NOx), volatile organic compounds (VOCs) and polycyclic aromatic hydrocarbons (PAHs) that are emitted during combustion. We estimate emission factors for those combustion-related pollutants from accidental landfill fires and backyard burning using emission factors for the domestic combustion of wood, open burning of plastic film, and uncontrolled combustion of refuse, which are available in EPA's AP-42 database (U.S. EPA, 1995). For PVC and ABS, we take into account additional pollutants that are not available in AP-42, such as benzene, toluene and HCl.

Details on methods used to derive these emission factors and develop end-of-life estimates are provided in Appendix D.

3.3.2 Life cycle impact assessment

The end result from the life cycle inventory models described in the previous section is a table of estimated total life cycle releases (in kg) for each of hundreds of different pollutants to air, and of others to water, and still others to land. This is a sum, over the entire system, of the red arrows depicted in Figure 3-1. The life cycle inventory results also include estimates of the total mass of each resource extracted from the environment – a sum of the green arrows in Figure 3-1. These inventory flows are estimated because of the different impacts that they can cause on human health and the environment. As a next step in the life cycle assessment, the methods of life cycle impact assessment (LCIA) are used to characterize each of these hundreds of different flows in terms of their relative and potential cumulative influence on impacts of concern. The impact categories chosen for this study include those listed in EPA's Tool for the Reduction and Assessment of Chemical and Other Environmental Impacts (TRACI) method (Bare et al., 2003). To these impact categories we added specially-developed Stockholm-related impact categories for cancer and for ecotoxicity. We also added modeling of estimated neurotoxicological impacts of mercury releases and exposures. The impact categories and their units are:

Environment/Resource

- Acidification – damage to forest and freshwater ecosystems from "acid rain" due to emissions of air pollutants (+H millimoles)
- Ecotoxicity – damage to ecosystems from toxic pollutants (g 2,4-D-equivalent)
- Eutrophication – changes to aquatic ecosystems due to introduction of excess nutrients (g N-equivalent)
- Fossil Fuel depletion – depletion of resources of coat, petroleum, and natural gas (Surplus MJ)

Combined human and ecosystem health

- Ozone Depletion – damage to the protective ozone layer high in the stratosphere (g CFC-11 equivalent)
- Photochemical Smog – formation of ozone in the air that people (and plants) breathe (g NOx equivalent)

Strictly human health[7]

- Human Health: Particulates – HHPM (microDALY)
- Human Health: Cancer – HHC (microDALY)
- Human Health: Non-Cancer – HHNC (hazard index)
- Human Health: Global Climate Change – HHGW (microDALY)
- Human Health: Mercury – HHHg (microDALY)

Notice that the units of the environment/resource impact categories, and the human/ecosystem health impact categories, are each different. This means that if we wanted to assess somehow the overall environmental impacts of a product based on its life cycle inventory and impact assessment, we would need to do some aggregating of different types and units of impact; this in turn requires subjective weighting. If we had used the existing TRACI impact assessment methods for cancer-related human health impacts, we would have the same problem of different units and an inability to estimate the overall human health impacts without using arbitrary and subjective weighting. For this reason, in order to enable more objective and integrated aggregation of human mortality risks, we used a modified form of the TRACI cancer impact assessment model which provides mortality risk estimates. For the same reason we used the EcoIndicator 99 impact assessment model for global climate change impacts (Goedkoop and Spriensma, 2001). Characterization factors are used to estimate the population impacts associated with life cycle emissions. Characterization factors were developed for:

- Particulate matter exposure and potency
- Cancer exposure and potency
- Non-cancer exposure and potency
- Global climate change
- Metals
- Mercury

Details on methods used to develop these characterization factors are presented in Appendix E.

[7] As discussed later, our method to integrate occupational risk and use-phase risk results (both for cancer and non-cancer impacts) with the pollution-driven population risk results (which are standard impact categories in LCA) has delivered some noteworthy conclusions for the LCAs and risk studies reported here: the occupational impacts did not appear to be at all trivial by comparison with the population risk impacts in these life cycles. This is partly due to the conservative method we have used to estimate occupational exposures, but nevertheless, it indicates that occupational exposures should not be automatically ignored in LCAs in the future.

3.4 Risk Assessment

3.4.1 Overview of risk assessment

In this report, the Task Group has developed risk estimates for occupational exposures to compounds used in the manufacture of select vinyl building materials and their alternatives, as well as estimates for installers and end users of flooring. The methodology used is established by the National Academy of Sciences and is standard risk assessment practice guided by the U.S. EPA. It involves the following four steps:

1. Identification of compounds that may be hazardous to the individuals of interest (Hazard Identification),
2. Assessment of the exposure of the individuals to the compounds (Exposure Assessment, e.g., how much VCM might a person inhale?),
3. Identification of the range of doses that are anticipated to cause harm in humans (Dose-Response Assessment, e.g., how much of this compound can a person be exposed to without any resultant adverse effects?), and
4. Characterization of the risk (Risk Characterization, e.g., quantify the likelihood that someone might develop cancer or some non-cancerous disease as a result of exposure).

A thorough explanation of the steps taken by the Task Group in the development of the occupational and end-user risk values is given in Appendix F, Human Health Risk Assessment.

Identification of Relevant Compounds

The compounds evaluated in the risk assessment are the following:

Pipe	Window Frames and Siding	Flooring
VCM	VCM	acetaldehyde
EDC	barium	acetone
Acrylonitrile	dibutyltin stabilizers	benzyl alcohol
1,3-butadiene	coke oven emissions	2-butoxyethanol
Styrene	silica	ethylene glycol
iron oxide fume	pyrene	naphthalene
coke oven emissions	benzo(a)pyrene	propionaldehyde
Limestone	aluminum	toluene
Silica	fluoride	trimethylsilanol
Manganese		phenol
Pyrene		formaldehyde
benzo(a)pyrene		furfural
		VCM
		DEHP

Exposure data were obtained from published literature, industry estimates, or by using OSHA PEL values (to provide an upper bound risk estimate). Average daily doses of a particular compound (over the working tenure of the occupational worker or over the appropriate timeframe of end use) were generated using default inhalation and activity values published by the U.S. EPA. These doses were then compared to regulatory limits (Reference Doses for non-cancer effects and Cancer Slope Factors for cancer endpoints) to estimate risk.

3.4.2 Estimates of non-cancer health hazards and cancer risks

Non-cancer Hazard Index values and Integrated Lifetime Cancer Risk values were obtained by summing the risk values generated for each compound to which individual workers may be exposed. Our goal is to arrive at estimates of the overall human health mortality risk from all of the releases from all of the processes in the life cycle, *per unit of function delivered by the product* (for example, per 20 square-foot-years of flooring). The occupational risk numbers initially reflect expected incremental mortality risk for a worker exposed routinely to the modeled concentrations of the pollutants of interest. In order to convert these results to a functional unit basis, we need estimates of the number of worker-hours required from the various unit processes in the life cycle, in order to produce the amount of product required to deliver the functional unit. In order to make this conversion, we used data on the amount of material produced per year for a process of interest in physical units, the average price of the material in order to convert production figures to dollars; and the input of manufacturing labor time to produce a unit of output (production worker hours per dollar) The resulting risk values, converted to a functional unit basis, are presented in a summary table in Appendix G.

3.5 Materials and Alternatives Selected for Study

In response to its charge, the PVC Task Group identified several building applications in which PVC-based materials have a significant market share. Then, to explore whether there was a basis for a PVC-related credit, the Task Group ***investigated whether for those applications the available evidence indicates that PVC-based materials are consistently among the worst of the materials studied in terms of environmental and health impacts***. Four applications representing a diversity of uses of PVC in buildings were chosen for study. For each application, several alternatives were selected to represent a broad range of commonly-used alternative products. Since the study was not designed to determine which alternative is “best” in any application, only whether PVC is consistently among the worst alternatives, it was not necessary to include all potential alternatives. The applications and alternative materials for each application that were studied are described briefly below; descriptions of each of the materials and the assumptions used in the analyses are included in Appendix C.

3.5.1 Siding

Four types of siding—vinyl, aluminum, wood, and fiber-cement—were analyzed for this report. For aluminum, wood and fiber-cement, a painting cycle of 6 years was assumed. The paint was assumed to consist of water-based 71% of the time and solvent-based 29% of the time.

- Vinyl siding was assumed to have a useful life 40 years. Galvanized nails for installation were included in the model.

- Aluminum siding is assumed to have a useful life of 80 years. It was assumed to have a recycled content of 21.4%. Aluminum nails needed for installation were also included in the models.
- Wood siding was modeled as beveled cedar siding (clapboards) with galvanized steel nails. The siding was assumed to have been installed over battens for durability, giving it a 40-year useful life.
- Fiber cement siding was modeled with a 50-year useful life. Galvanized steel nails were assumed for the installation, with a 5% installation waste.

3.5.2 Drain/waste/vent pipe

The three most common materials used to make drain/waste/vent (DWV) pipe for building applications were selected for study: PVC, ABS (acrylonitrile butadiene styrene), and cast iron. The life span of all the pipe products has been assumed to be 50 years.

- ABS is assumed to contain only the ABS polymer.
- PVC is assumed to contain only PVC polymer.
- Cast iron is assumed to be made entirely from scrap iron and steel (100% recycled content), although separate analyses were performed with 90% recycled content to test this assumption.

3.5.3 Resilient flooring

Two PVC-based resilient flooring products—sheet vinyl and vinyl composition tile (VCT)—and two non-PVC products—linoleum and cork—were selected for study.

- Sheet vinyl was modeled using several different compositions to reflect the variation in the marketplace. Styrene butadiene flooring adhesive was also included in the models. he presumed life span of sheet vinyl is 15 years.
- Vinyl composition tile was modeled with a useful life of 18 years. Styrene butadiene flooring adhesive was included in the models.
- Linoleum was assumed to have an 18-year useful life. Styrene butadiene flooring adhesive was included in the model.
- Cork flooring was assumed to have a 50-year useful life. A water-based contact adhesive was modeled for the installation, and transportation to North America from Europe was also included.

3.5.4 Window Frames

We compare windows that have frame materials made entirely from PVC, aluminum, or wood, with the associated ancillary materials needed for their assembly and installation. The impact of the window frames on energy use while a building is occupied was included in the analysis.

- Vinyl window frames were modeled with the following framing components: PVC (85%), fiberglass (6%), galvanized steel (5%), and EPDM (5%).

- Aluminum window frames used in the model contained no thermal breaks, and therefore have much higher use-phase energy use than the others.[8] The framing components include aluminum with 21.4% recycled content (86%), fiberglass (6%), galvanized steel (4%), and EPDM (4%). A painting cycle of 7 years for the interior surface and 9 years for the exterior surface was assumed.
- Wood window framing components include wood (86%), aluminum (4%), fiberglass (2%), galvanized steel (2%), and EPDM (2%). A painting cycle of 7 years for the interior surface and 9 years for the exterior surface was assumed.

[8] According to Lawrence Berkeley National Laboratory, in 2003, in the residential market, aluminum frames were 10% of the total and approximately 60% of these frames had no thermal break. In the nonresidential market, aluminum was 87% of the total (including curtain wall, storefront and manufactured windows) and approximately 26% had no thermal break.

4 Findings of the Integrated Life Cycle Assessment and Risk Assessment

4.1 Format for Presentation of Findings

This chapter presents the findings of the integrated Life Cycle Assessment and Risk Assessment for window frames, piping, siding, and flooring.

Following the integrated LCA-RA results, findings on mortality associated with vinyl chloride, phthalates, and mercury are presented. These findings could not be integrated into the LCA-Risk Assessment findings.

As described earlier in this report, our analysis of the impacts of PVC-based building components versus other material alternatives in various building uses has been addressed using an analysis framework which is driven by the goal of achieving comprehensiveness in two dimensions: (1) the full life cycle and (2) as full as possible a set of human health and environmental impact categories.

The presentation of results in this section takes into account the fact that different stakeholder groups may be concerned with different impact categories. For example, some may focus strictly on impacts to human health associated with releases of those chemicals named by the Stockholm Convention on persistent pollutants. Others may only focus on environmental impacts. For this reason, we summarize results and conclusions relative to our charge (whether or not the evidence supports a credit related to the use of PVC) with respect to a variety of different impact scopes:

- **Human cancer impacts** from releases of chemicals in the life cycle inventories. These impacts include the potential influence of approximately 150 chemicals released from processes to air, water, soil, and exposures within occupational environments, including eight carcinogens listed under the Stockholm Convention.;
- **"Combined" human health impacts**, including cancer as well as possible human health consequences of global climate change, exposures to particulates and mercury, and cancer from environmental releases of metals.
- **Environmental impacts**, including acidification, eutrophication, eco-toxicity, smog formation, ozone depletion potential, global climate change, and fossil fuel depletion.

Results and conclusions are also summarized based on different degrees of comprehensiveness across the life cycle:

- **Cradle-through-use** life cycle assessment results, augmented by our best estimates for emission factors for pollutants lacking data in the available LCA databases or which were believed (in part due to stakeholder input) to be systematic under-estimates. The emission factor estimates are supported by sensitivity tests with lower- and upper-bound estimates to reflect the inherent uncertainties.
- Adding **end-of-life** disposal to the scope, including (motivated by stakeholder input) our best estimates (with upper- and lower-bound sensitivities) of possible emissions due to open burning and accidental landfill fires;
- Adding health impacts per functional unit based on risk assessments for **occupational exposures** in product manufacturing, installation and use;

The presentation of results at different levels of life cycle comprehensiveness is done to clarify separately the influence on results and conclusions of the highly uncertain modeling of end-of-life emissions, and of the occupational exposure risks that are traditionally not captured in standard life cycle assessments. This clarification and comparison may help to

highlight why, in some cases, our results and conclusions may differ from those of prior LCA-based comparisons.

For each material group, the findings include:

- A summary that ranks all material alternatives, including PVC, according to environmental and human health impacts, for cradle-through-use, with the addition of end-of-life impacts, and with the addition of occupational risks.
- Environmental impacts, for cradle-through-use and with the addition of end-of-life impacts
- Human health impacts, for cradle-through-use, with the addition of end-of-life impacts, and with the addition of occupational risks.
- A more detailed analyses of the contribution of each category of human health impact and the range of uncertainties associated with the estimates
- Human mortality results in perspective
- Sensitivity analysis of the uncertainties in global climate change and particulate impacts
- Hazard index results for other toxic risks not included in the life cycle assessment and risk assessment results
- Notes on data gaps and uncertainties.

The results include uncertainties based on assumptions and emission factors. Therefore, on the summary tables, there are “high”, “low”, and “average” results presented for each material alternative. The variable assumptions are summarized below.

- **Mercury**. Our “low” assumption on mercury impacts is limited in two ways. First, it is based strictly on neurotoxicity impacts and second, it does not assume that all the “missing” mercury from chloralkali processes was released to the environment; thus, our ”high” assumption on mercury adds additional health impacts and also assumes all missing mercury was released to the environment.
- **Global Climate Change.** Health impacts from global climate change are very uncertain. Our baseline assumption on global climate change uses the standard impact factors from EcoIndicator 99. However, that publication also cited a geometric standard deviation for the uncertainty in this factor, and as a low-end estimate we use the mean divided by the square of the standard deviation.
- **Particulates.** Health impacts for particulates are much more certain than those for global climate change. Still, there are some possible ways in which our estimated impacts could be over-estimated; we test cases in which both our estimated intake fractions (exposures to the emitted pollution) are lower than baseline, and the expected health sensitivity is lower than baseline, using a published lower-bound concentration-response slope.
- **Cancer: End-of-Life: Dioxin Emissions:** Health impacts from exposure to dioxin emissions from accidental landfill fires and barrel burning of PVC and other materials are highly uncertain for a variety of reasons. The first is the uncertainty about the health impacts of a given (known) exposure; this uncertainty is not modeled quantitatively in our analysis. Two uncertainties we do treat quantitatively using scenarios; the first is uncertainty about the mass fraction of each discarded building material which ends up being burned in a barrel or in a landfill fire. The second is the uncertainty in the dioxin emissions per quantity of material burned in either a barrel or a landfill fire. We are

confident the actual emissions lie somewhere above our lower bound for dioxin emissions from end-of-life burning. Our confidence that the actual end-of-life dioxin emissions are below our upper bound is difficult to estimate because of the very high uncertainties; it is our best attempt at an upper bound which is conservatively high yet plausible.

- **Cancer: Occupational Exposures:** Health impacts from occupational exposures are very uncertain, partly because data on actual levels of occupational exposure to chemicals are lacking for most processes and most chemicals. Our upper bound estimates of exposure generally reflect OSHA Permissible Exposure Limits (PELs) while the lower bound estimates are either zero or a low empirical estimate found in the literature.

4.2 Window Frames

4.2.1 Summary

Environmental impacts: Aluminum window frames, with or without thermal breaks[9], are consistently the worst material relative to environmental impact categories among the alternatives studied.

Human health impacts: Aluminum frames, with or without thermal breaks, are worst for cradle-through-use portion of the life cycle among the alternatives studied. With the additional releases from end-of-life including accidental burning, aluminum remains worst relative to overall human health impacts among alternatives studied, though PVC is worst if focusing strictly on cancer-related impacts to health. Adding the impacts of occupational exposures within manufacturing and installation, either PVC or aluminum is worst among alternatives studied, depending upon assumptions.

[9] We also analyzed aluminum windows with thermal breaks, using data on usage phase thermal performance from Lawrence Berkeley National Laboratory's RESFEN database, provided at http://www.efficientwindows.org. These data show that for an equivalent glazing type, thermally broken aluminum frame windows have a thermal efficiency that is intermediate between those of unbroken aluminum and vinyl frames. The results of life cycle comparisons indicate that the relative performance of wood, aluminum and vinyl window frames do not change when thermal breaks are added to the aluminum windows; their life cycle health and environmental impacts remain worse than those for PVC frames. A comparison of the results with and without thermal breaks is provided in Appendix K.

Table 4-1: Ranking of Materials by Scale of Impacts – Window Frames Listed from Greatest Impact or "Worst" (1)

Rankings are considered "tied" when the difference from the next ranked material is less than 20%.			
Impact Category Environment	**Cradle-through-use, environmental pathways**	**Plus end-of-life with accidental burning**	
Environment*	1. Aluminum high (1,1,1,1,1,1,1) Aluminum avg (2,2,2,2,2,2,2) Aluminum low (3,3,3,3,3,3,3) 4. Wood high (5,4,5,4,4,7,5) Wood avg (7,5,7,5,8,8,7) ***PVC high*** (4,7,4,7,5,4,4) ***PVC avg*** (6,8,6,8,6,5,6) Wood low (9,6,9,6,9,9,9) ***PVC low*** (8,9,8,9,7,6,8)	1. Aluminum high (1,1,1,1,1,1,1) Aluminum avg (2,2,2,2,2,2,2) Aluminum low (3.3.3.3.3.3.3) 4. Wood high (6,4,4,4,4,4,7) Wood avg (7,6,5,6,5,8,8) ***PVC high*** (4,5,7,5,7,5,5) Wood low (9,8,6,8,6,9,9) ***PVC avg*** (5,7,8,7,8,6,5) ***PVC low*** (8,9,9,9,9,7,6)	
Impact Category Human Health	**Cradle-through-use, environmental pathways**	**Plus end-of-life with accidental burning**	**Plus occupational impacts including installation**
Cancer **	1. Aluminum high Aluminum avg Aluminum low ***PVC high*** 5. Wood high Wood avg Wood low ***PVC avg*** ***PVC low***	1. ***PVC high*** 2. ***PVC avg*** 3. Aluminum high Aluminum avg ***PVC low*** 6. Aluminum low 7. Wood high Wood avg Wood low	1. ***PVC high*** 2. ***PVC avg*** 3. Aluminum high ***PVC low*** 5. Aluminum avg 6. Aluminum low 7. Wood high Wood avg Wood low
Total human health***	1. Aluminum high Aluminum avg Aluminum low 4. ***PVC high*** 5. ***PVC avg*** Wood high ***PVC low*** Wood avg Wood low	1. Aluminum high Aluminum avg Aluminum low 4. ***PVC high*** ***PVC avg*** ***PVC low*** Wood high Wood avg Wood low	1. ***PVC high*** Aluminum high Aluminum avg Aluminum low 5. ***PVC avg*** 6. ***PVC low*** 7. Wood high Wood avg Wood low

* Note: rankings with respect to each separate environmental impact category are presented in parenthesis, with the following impact category order: acidification, eutrophication, smog, ozone depletion, global climate change, fossil fuel depletion, ecotoxicity). The materials appear in this table in the order of their average normalized performance across the seven impact categories. The order of appearance does *not* indicate an overall environmental score, since such an overall score would require value-based weighting across the impact categories.
** Includes cancers from exposure to chemicals and metals
***Includes cancer from row above plus effects of global climate change, particulates, mercury

4.2.2 Environmental impacts

Environmental impact results are summarized in the two figures on this page. Because the results for each impact category are measured in different units, the figure presents results that have been normalized by dividing the results in each impact category by the results for the highest-impact material and emissions estimate.

The two figures below show the aluminum window frames to be worst overall among alternatives studied; this is due to their lower energy efficiency.

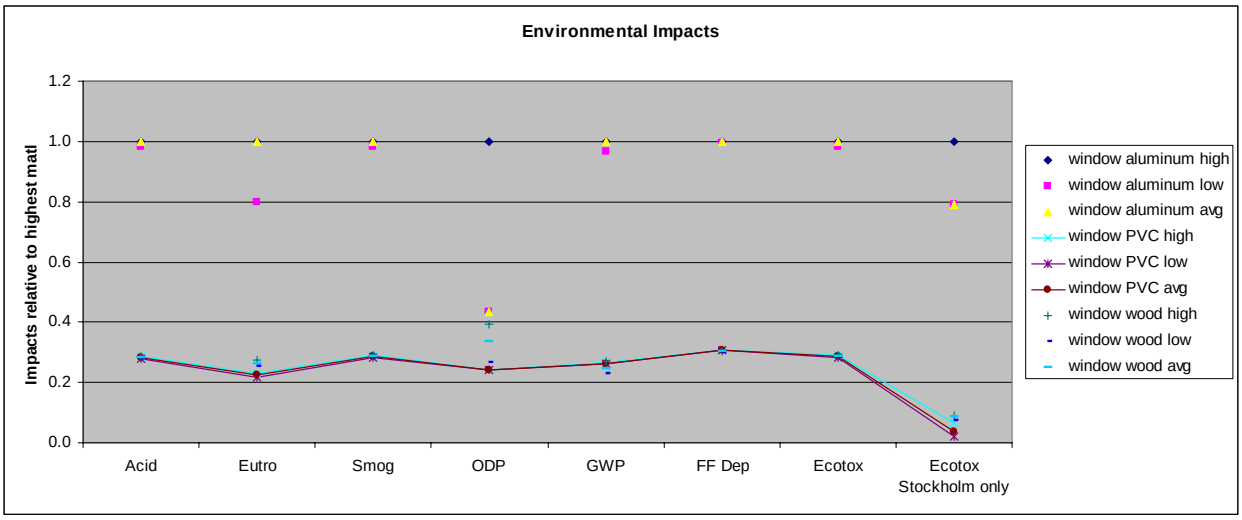

Figure 4-1: Environmental impacts of window frames, normalized to highest impact per category, cradle-through-use, without end-of-life

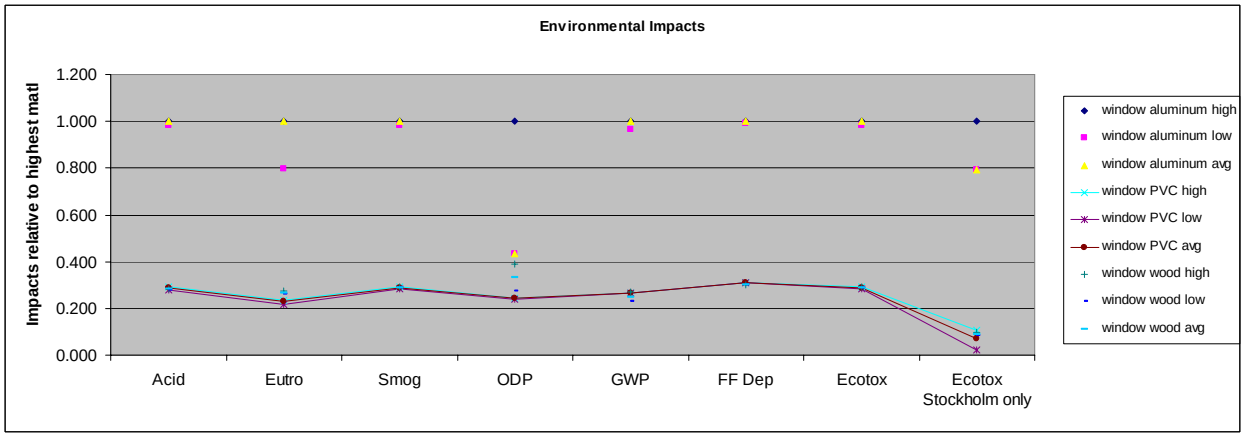

Figure 4-2: Environmental impacts of window frames, normalized to highest impact per category, cradle-through-use with end-of-life

4.2.3 Human health impacts

Human Health Impacts – Cradle through use

The scale of the graph below is logarithmic, in order to allow clear visualization of results which vary by several orders of magnitude. The results in the figure below lead to the following conclusions: aluminum window frames are consistently worst relative to total human health impacts among alternatives studied, and tend to be worst relative to cancer impacts, although the maximum PVC case is worse than the minimum aluminum case. Also, human health impacts of global climate change ("HH GW") are the most significant contributor to total human health impacts as a group, followed by particulate impacts ("HH PM").

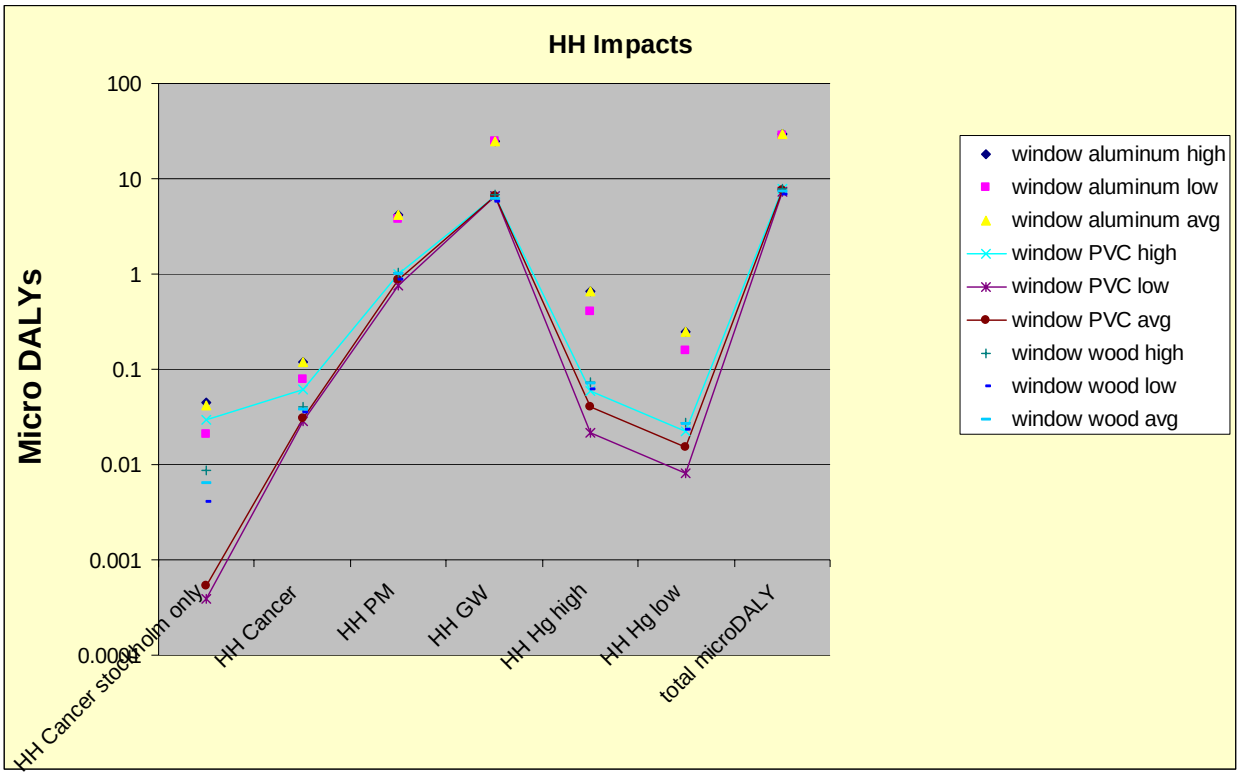

Figure 4-3: LCA results for window frames, cradle-through- use, without end-of-life or occupational effects

Notes on how to read the Human Health Impact Figures on this and subsequent Figures:
For each material alternative there are three sets of results, corresponding to combinations of assumptions on uncertainties in emission factors in the life cycle modeling
"HH" means Human Health; the units of all of these impacts are "micro-DALYs", or millionths of a DALY, where one DALY is one disability-adjusted life-year. In our analysis these DALYs represent expected life years lost due to premature death; the morbidity impacts of particulate exposures are the only non-mortality impact, and are a small share of the DALYs due to particulate exposures.
Cancer impacts reflect population cancer risk associated with exposure to chemical and metal carcinogens. The first category of cancer impacts (on the far left) are the total cancer risks from strictly the set of chemicals addressed by the Stockholm convention; they are a subset of the total cancer impacts displayed as "HH Cancer."
"PM" (Particulate matter) impacts are associated with releases of primary particulates and releases of nitrogen oxides and sulfur oxides which react in the atmosphere to form particulates. "GW" (Global climate change) impacts are estimates of potential human health consequences of global climate change due to releases of greenhouse gasses
"Hg high and HH Hg low": potential neuro-toxicological impacts of mercury emissions under high and low assumptions related to uncertainties in magnitude of release quantities and impact pathways.
Total Micro DALY: The sum of the cancer, particulates, global climate change, and mercury-related impacts.

Human Health Impacts – Cradle through Use Plus End-of-Life

The figure below shows the results after adding the emissions from end-of-life processing, including the risk of emissions from accidental landfill fires and from backyard burning. These results can be displayed using a linear scale on the vertical axis, because the cancer-related impacts (from end-of-life emissions) are no longer orders of magnitude lower than the expected impacts from particulates and global climate change. These results lead to the following conclusions: PVC frames are worse for cancer-related impacts, while aluminum frames are worse for human health over all among the alternatives studied.

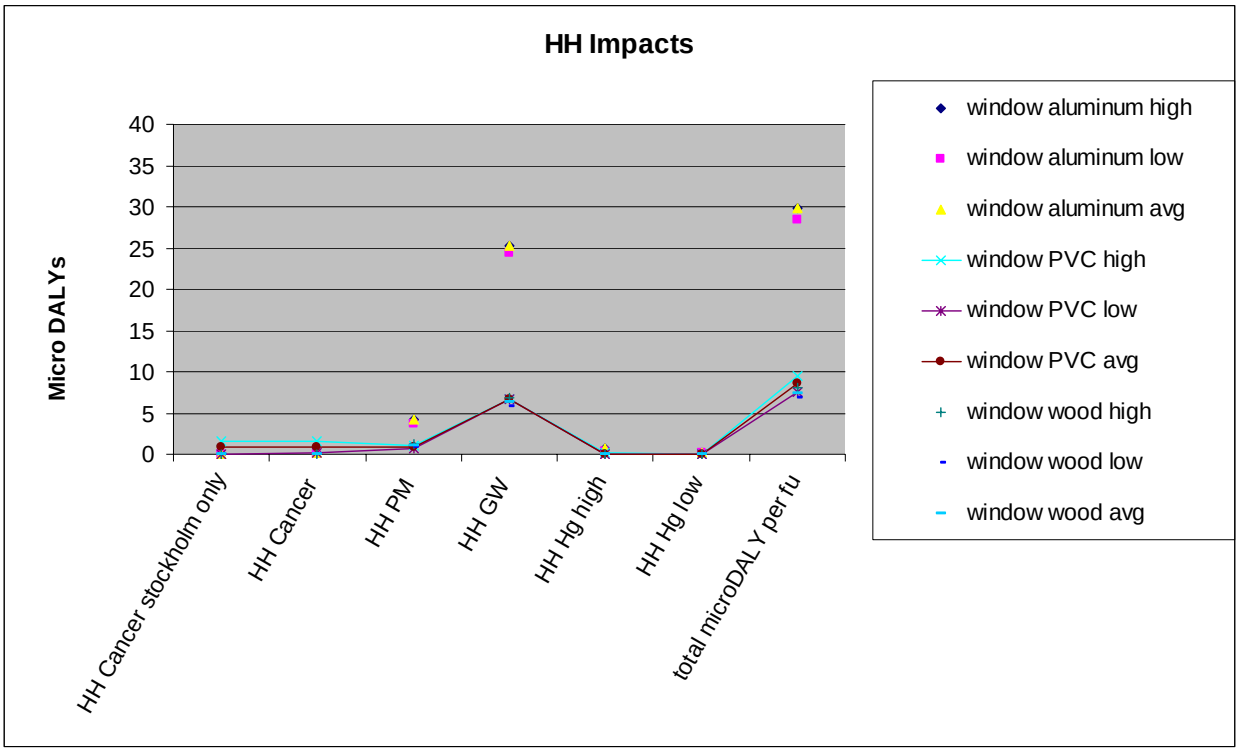

Figure 4-4: LCA results for window frames, cradle-through-use with end-of-life but without occupational effects

For notes on how to read this figure, see Figure 4-3.

Human Health Impacts – Cradle through Use Plus End-of-Life and Occupational Effects

Adding the impacts to health from exposures of workers in the production and installation of the products leads to the results shown in the figure below. These are the most comprehensive human health results in terms of the scope of the life cycle model. The resulting conclusions are that for overall human health impacts, either aluminum or PVC is the worst of the frame materials studied; the results are influenced by the uncertainties about emission factors in the life cycle and risk modeling of the studied systems. The worst material overall among alternatives studied is PVC under high-end assumptions; next come the three results for aluminum frames, and the remaining PVC frame results.

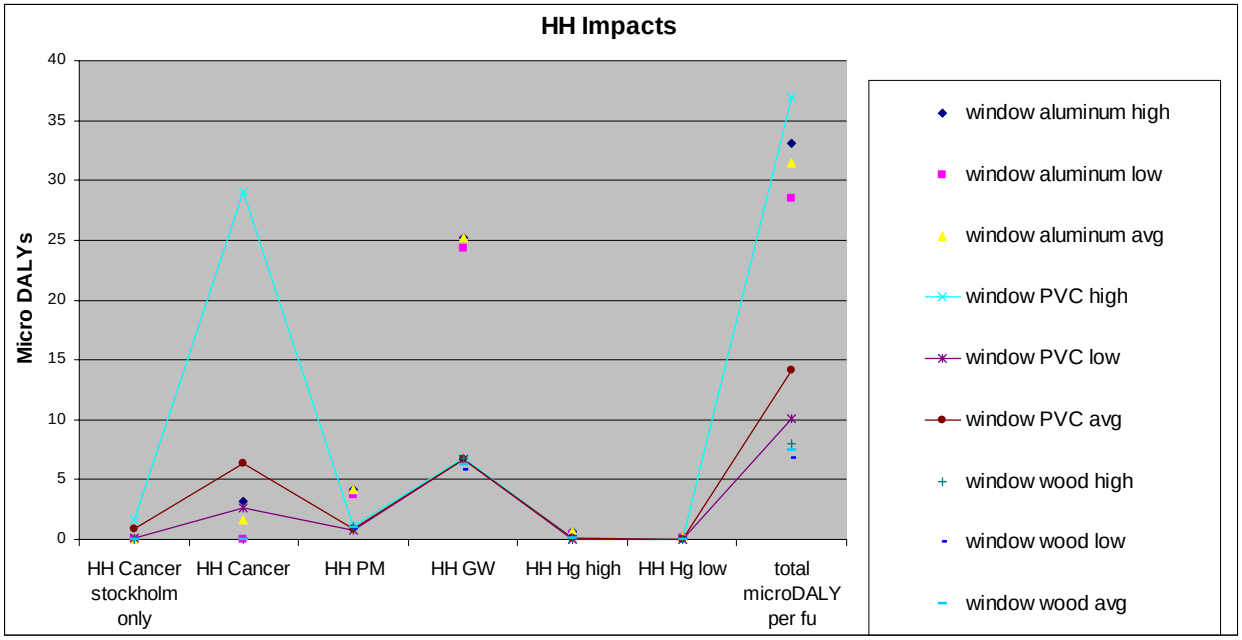

Figure 4-5: LCA results for window frames, cradle-through-use with end-of-life and occupational effects

For notes on how to read this figure, see Figure 4-3.

Human Health Impacts – Mortality Results in Perspective

The figure below helps summarize the findings on human health mortality impacts, not only the comparisons among the materials, but also the ways conclusions change as we move from low-end to high-end estimates on uncertain emissions and exposures, *and* the relative influence of occupational cancer risks, end-of-life impact, cradle-through-use cancer and cradle-through-use energy-related emissions (global climate change, particulate, and mercury impacts).

Interpretive conclusions which the figure helps clarify include:

- Wood window frames appear reliably better than the other two options in relation to life cycle mortality risks;
- Cradle-through-use cancer mortality risks are negligible compared to the other risks in the life cycle (they are invisible in this figure);
- Energy-related emission impacts play the largest role by far for wood and aluminum window frames, and for PVC as well except for the high-end scenarios for occupational exposures;
- End-of-life emission risks do not play a large role in the total risks, even for PVC window frames in the high-end emission/exposure scenario.

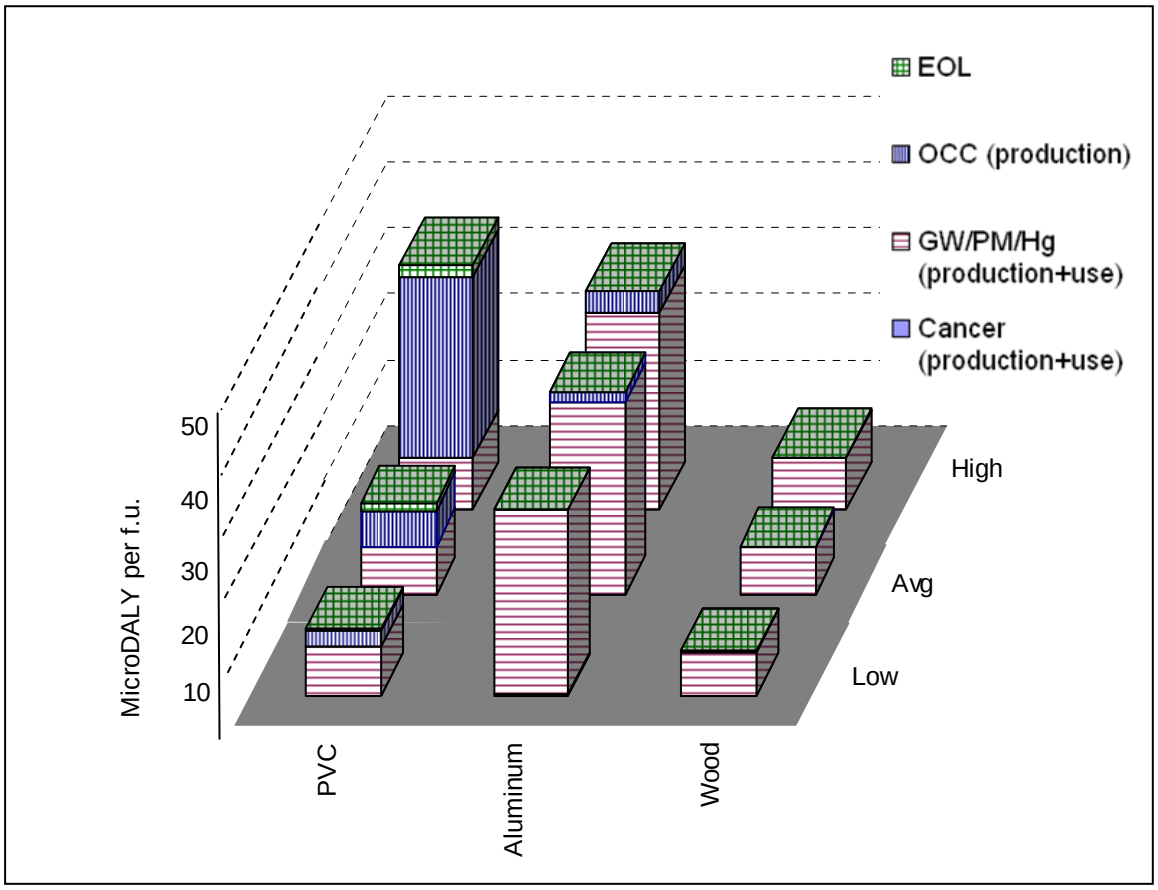

Figure 4-6: High, average and low estimates of human health impacts (microDALY per functional unit) by life cycle stage – Window frames

Human Health Impacts – Sensitivity Analysis of Uncertainties in Global Climate Change and Particulate Impacts

The results and conclusions are tested for their sensitivity to uncertainties in the impact modeling for global climate change and particulates, in the figure below. Under low-end estimates for the expected health impacts of pollution in both of these impact categories, the conclusion remains that average aluminum frames are worse than average PVC frames, and both of them are considerably worse than wood frames.

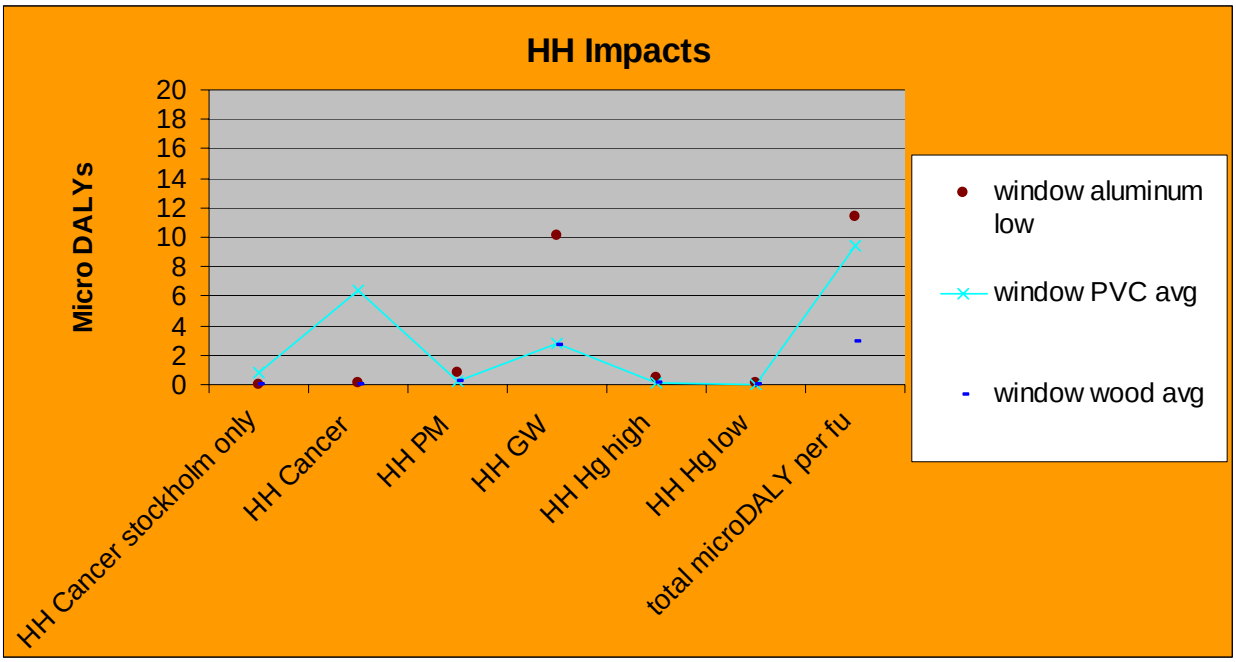

Figure 4-7: Sensitivity analysis for window frames using average emission levels and low-end assumptions for impacts of global climate change and particulates, cradle-through-use with end-of-life and occupational effects

Human Health Impacts – Hazard Index Results for Human Health Morbidity Risks

The Hazard Index (HI) is another metric for human health impacts. The HI is a ratio of a person's actual exposure to a compound divided by the concentration of that compound that represents a recommended upper limit on a daily basis (by the U.S. EPA or other regulatory body). Therefore, HI values of greater than 1 mean that people are exposed to concentrations of compounds that exceed that limit. HI values >1 do not necessarily mean that a person will develop an adverse health effect (such as liver toxicity or asthma). However, they do indicate a greater chance for such an effect if concentrations stay constant and exposures are not limited.

Sums of the total Hazard Index related to human health toxicity impacts for non-carcinogenic effects are summarized in the figure below. For these pollutants and impact pathways, aluminum frames are worst among the alternatives studied. The relative rankings of the building materials with respect to HI are generally consistent with those for the mortality-related summaries of human health impacts.

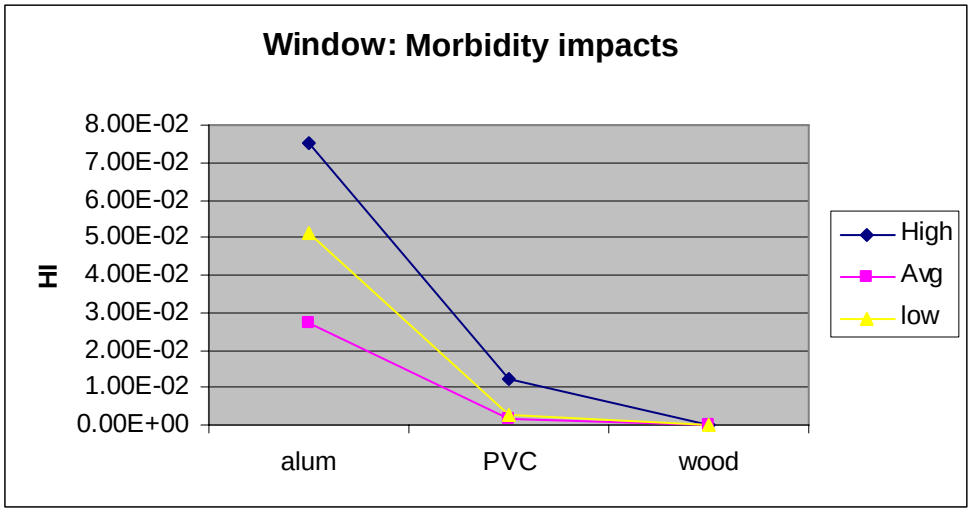

Figure 4-8: Morbidity Hazard Index impacts for window frames, cradle-through-use with end-of-life and occupational effects

Notes on how to read the Human Health Morbidity Figures:
How does the hazard index ("HI") shown in the figure above differ from the DALY metric used in all other human health results figures? The DALY results reflect mortality risk in the case of cancer and global climate change and mercury exposures, and the bulk of the particulates impacts; total DALY results for particulates include the additional (small) contributions of non-fatal respiratory illness. The HI results, on the other hand, reflect a kind of aggregate measure of toxicity responses to chemical exposures. The reason that the HI results have not been converted to DALYs is that the exposure thresholds are generally based on levels at which the first adverse health effect is observed. But these adverse health effects vary widely in severity, starting with minor irritations. And there may be other potential health effects that occur at higher exposures for some chemicals, and not others. So not only do the most sensitive health effects vary from chemical to chemical, but the total possible adverse health affect risk cannot be captured in the HI. In our judgment, the majority of the total expected health impacts are captured in the DALY results, to which the additional impacts captured in the HI results represent a minor adjustment. However, the HI results are presented as well, for purposes of completeness.

Notes on Uncertainties in Window Frames Findings

The risk estimates for aluminum window frames reflect uncertainty about controls of exposures to coke oven emissions and PAHs. Risk estimates for vinyl window frames reflect exposure to vinyl compounds from the manufacture of the resin, and metals, coke oven emissions, and PAHs in the manufacture of steel that is also present in the windows.

It was not possible to develop risk estimates for wood window frames at this time as these building materials lack dose-response data for adverse effects in occupational workers. It is well documented that exposure to wood dust in various industries is linked to the incidence of nasal cancers (Goldsmith and Shy, 1988; Teschke et al., 1999), but data were unavailable that might indicate the increased relative risk for those exposed to wood dust in the manufacture of wood window frames and siding.

For PVC building materials, we find the manufacture of resin to be overwhelmingly the largest contributor. This is because vinyl chloride monomer (VCM) and ethylene dichloride (EDC), the compound used to make it, are both carcinogens. Further, exposure to both compounds is assumed in the manufacture of PVC resin, even though it is possible any one worker may be exposed to only one compound. While exposure data were available from the Vinyl Institute with regard to VCM, these data were not available for EDC; therefore, exposures were assumed limited by the OSHA PEL (permissible exposure limit) for EDC.

Individuals involved in the manufacture of plastic resins are potentially exposed to many other compounds including solvents used to clean the resins, such as methyl ethyl ketone, tetrahydrofuran, acetone, and cyclohexanone. These solvents, although known central nervous system depressants, are not as toxic as the individual resin monomers; in other words, their OSHA PEL values are much higher than those for VCM or acrylonitrile, butadiene, and styrene. Actual exposure data for occupational workers involved in cleaning resins were not available. Therefore, the contribution of potential risks from these compounds would have to be identical in the occupational risk analysis. To avoid the large uncertainty that might be introduced by assuming OSHA PEL exposure values for these compounds, with no corresponding actual data to provide a “reality check”, they were omitted from the risk analyses. It is acknowledged that this omission may result in artificially low risk estimates, which would cause an unfair comparison to alternatives. However, Health Hazard Evaluations in the NIOSH database were queried for PVC manufacture. None were identified that involved CNS or respiratory effects in workers that could be attributable to these solvents. Epidemiological studies of vinyl building products have focused predominantly on exposures to VCM and EDC and not solvent cleaners. Therefore, this omission is not thought to under-represent the relative risk of PVC compared to the alternatives.

4.3 Pipe

4.3.1 Summary

Environmental Impacts: Cast iron pipe is generally the worst material relative to environmental impact categories among the alternatives studied.

Human Health Impacts: PVC is worst for cancer-related impacts among alternatives studied, while cast iron or PVC is worse for overall health impacts depending on the assumptions when both end-of-life and occupational exposures are included. For the cradle-through-use, cast iron is worst overall among alternatives studied from a human health point of view.

Table 4-2: Ranking of Materials by Scale of Impacts – Pipe Listed from Greatest Impact or "Worst" (1)

Rankings are considered "tied" when the difference from the next ranked material is less than 20%.

Impact Category Environment	Cradle-through-use, environmental pathways	Plus end-of-life with accidental burning
Environment*	1. Cast iron high (1,1,1,1,1,1,1) Cast iron avg, (2,2,2,2,2,2,2) Cast iron low (3,3,3,3,3,3,3) 4. ***PVC high,*** (4,7,4,7,4,7,4) ABS high, (7,4,7,4,6,4,5) ABS avg, (8,5,8,5,7,5,6) ABS low, (9,6,9,6,8,6,7) ***PVC avg,*** (5,8,5,8,5,8,8) ***PVC low*** (6,9,6,9,9,9,9)	1. Cast iron high, (1,3,1,1,1,1,1) Cast iron avg, (2,4,2,2,2,2,2) Cast iron low (3,4,3,3,5,2,3) 4. ***PVC high,***(4,6,4,4,3,6,4) ***PVC avg,*** (5,8,5,5,4,7,5) ABS high, (7,1,7,6,7,3,7) ABS avg, (8,2,8,8,7,4,8) ABS low, (9,7,9,9,8,5,9) ***PVC low*** (6,9,6,6,9,8,6)

Impact Category Human Health	Cradle-through-use, environmental pathways	Plus end-of-life with accidental burning	Plus occupational impacts including installation
Cancer **	1. Cast iron high, ***PVC high*** 3. Cast iron avg 4. Cast iron low 5. ***PVC pipe avg*** 6. ***PVC pipe low,*** ABS high, ABS avg, ABS low	1. ***PVC high*** 2. ***PVC avg*** 3. ***PVC low*** 4. Cast iron high 5. Cast iron avg 6. Cast iron low 7. ABS high ABS avg, ABS low	1. ***PVC high*** 2. ***PVC avg*** 3. Cast iron high 4. ***PVC low*** 5. Cast iron avg, ABS high 7. ABS avg 8. Cast iron low 9. ABS low
Total human health***	1. Cast iron high 2. Cast iron avg 3. Cast iron low, ***PVC high*** 5. ***PVC avg,*** ABS high, ABS avg, ABS low 9. ***PVC low***	1. Cast iron high, ***PVC high*** 3. Cast iron avg, ***PVC avg*** 5. C.iron low, ABS high, ABS avg, ABS low, ***PVC low***	1. ***PVC high,*** Cast iron high 3. Cast iron avg, ***PVC avg*** 5. Cast iron low, ABS high, ABS avg 8. ABS low, ***PVC low***

* Note: rankings with respect to each separate environmental impact category are presented in parenthesis, with the following impact category order: acidification, eutrophication, smog, ozone depletion, global climate change, fossil fuel depletion, ecotoxicity). The materials appear in this table in the order of their average normalized performance across the seven impact categories. The order of appearance does *not* indicate an overall environmental score, since such an overall score would require value-based weighting across the impact categories.

** Includes cancers from exposure to chemicals and metals

***Includes cancer from row above plus effects of global climate change, particulates, mercury

4.3.2 Environmental impacts

Environmental impact results are summarized in the two figures on this page. Because the results for each impact category are measured in different units, the figures present results that have been normalized by dividing the results in each impact category by the results for the highest-impact material and emissions estimate. The results show that cast iron pipe is consistently the worst material among alternatives studied except for eutrophication, for which ABS is the worst when end-of-life emissions are considered.

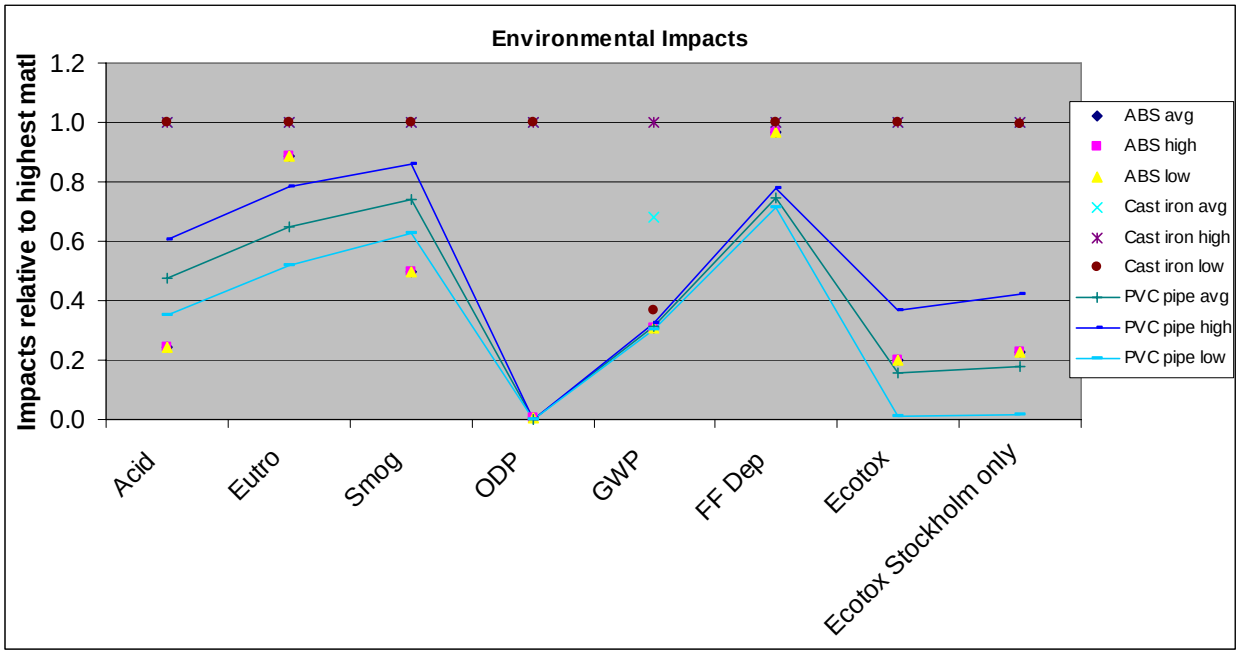

Figure 4-9: Pipe environmental impacts, normalized to highest impact per category, no EOL

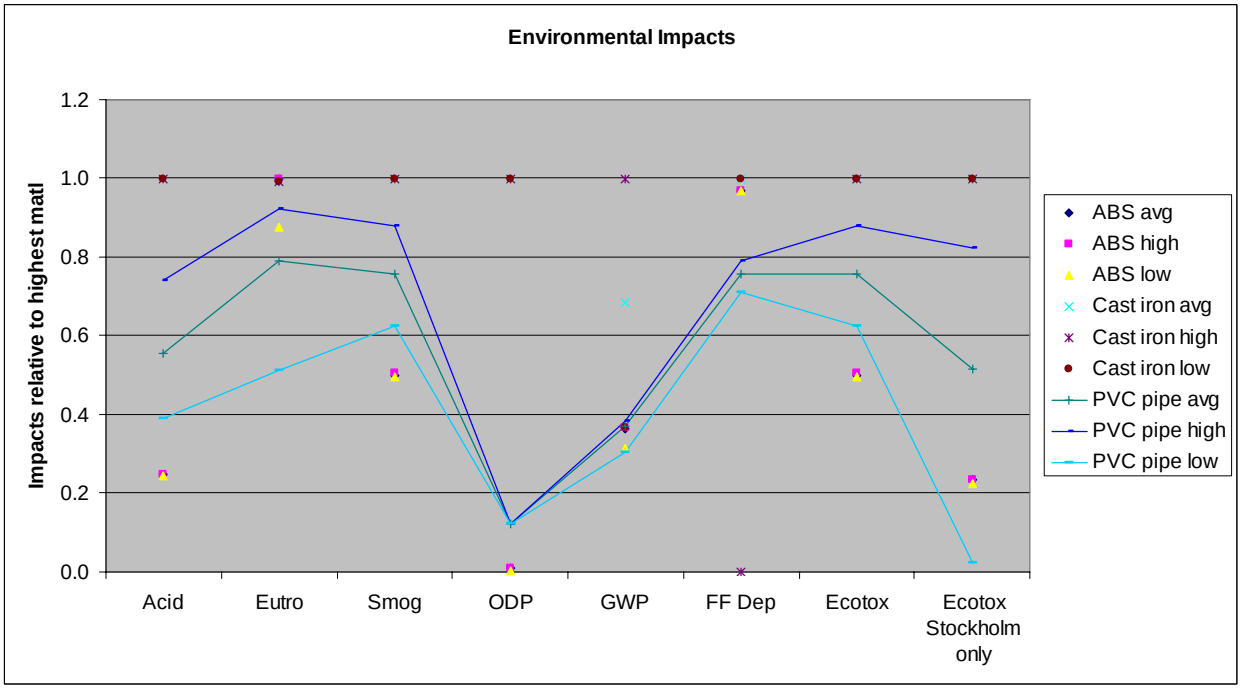

Figure 4-10: Pipe environmental impacts, normalized to highest impact per category, with EOL

4.3.3 Human health impacts

Human Health Impacts – Cradle-through-Use

The results for each material are shown relative to each of the separate human health impact categories in the figure below. The results in the figure below lead to the conclusions that cast iron pipe is worse than alternatives studied relative to total human health impacts for production and use, and tends to be worst relative to cancer impacts, although the maximum PVC case is very close to the maximum cast iron cases.

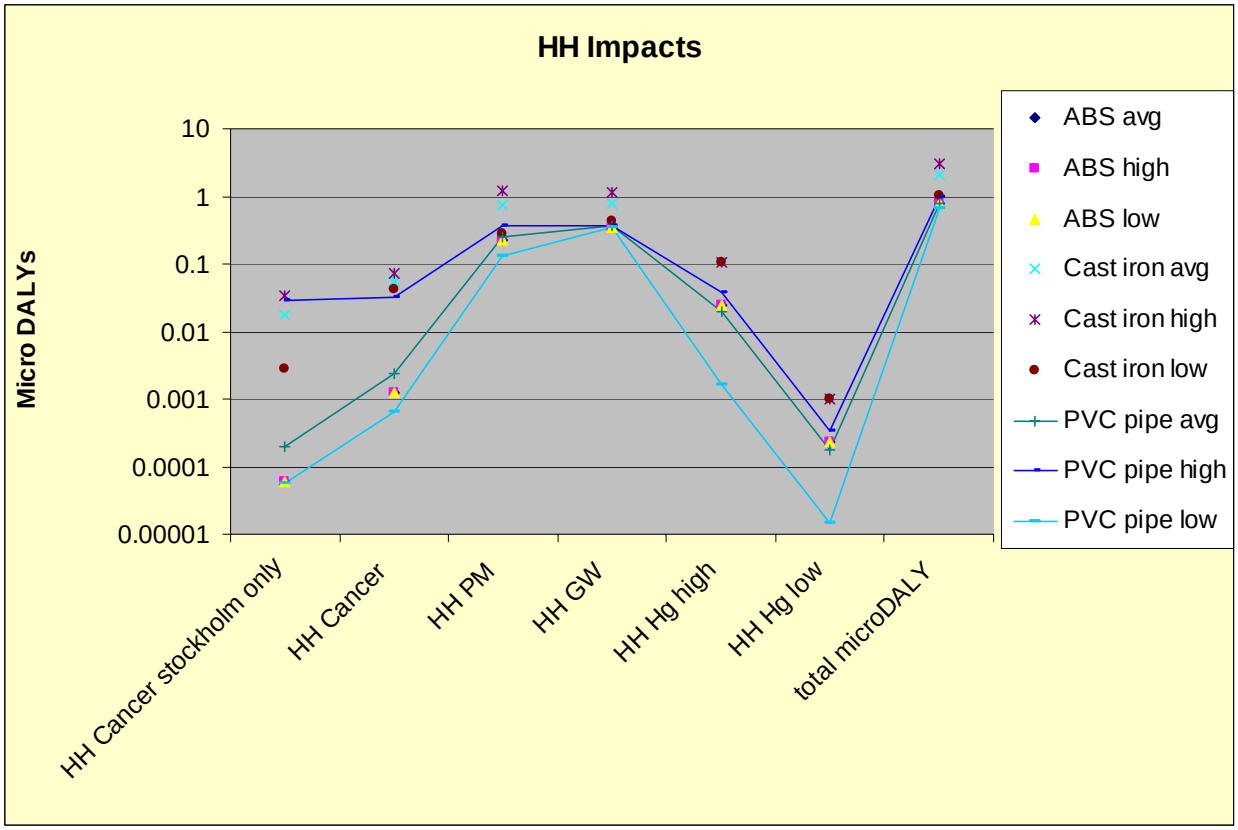

Figure 4-11: Pipe LCA results cradle-through use, no EOL, no Occupational

Notes on how to read the Human Health Impact Figures:
For each material alternative there are three sets of results, corresponding to combinations of assumptions on uncertainties in emission factors in the life cycle modeling
"HH" means Human Health; the units of all of these impacts are "micro-DALYs", or millionths of a DALY, where one DALY is one disability-adjusted life-year. In our analysis these DALYs represent expected life years lost due to premature death; the morbidity impacts of particulate exposures are the only non-mortality impact, and are a small share of the DALYs due to particulate exposures.
Cancer impacts reflect population cancer risk associated with exposure to chemical and metal carcinogens. The first category of cancer impacts (on the far left) are the total cancer risks from strictly the set of chemicals addressed by the Stockholm convention; they are a subset of the total cancer impacts displayed as "HH Cancer."
"PM" (Particulate matter) impacts are associated with releases of primary particulates and releases of nitrogen oxides and sulfur oxides which react in the atmosphere to form particulates. "GW" (Global climate change) impacts are estimates of potential human health consequences of global climate change due to releases of greenhouse gasses
"Hg high and HH Hg low": potential neuro-toxicological impacts of mercury emissions under high and low assumptions related to uncertainties in magnitude of release quantities and impact pathways.
Total Micro DALY: The sum of the cancer, particulates, global climate change, and mercury-related impacts.

Human Health Impacts – Cradle-through-Use Plus End-Of-Life

The figure below shows the results after adding the emissions from end-of-life processing, including the risk of emissions from accidental landfill fires and from backyard burning. These results can be displayed using a linear scale on the vertical axis, as the cancer-related impacts (from end-of-life emissions) are no longer orders of magnitude lower than the expected impacts from particulates and global climate change. These results lead to the conclusions that PVC pipe is worse than alternatives studied for cancer-related impacts, while cast iron pipe tends to be worse than PVC for the maximum and average cases.

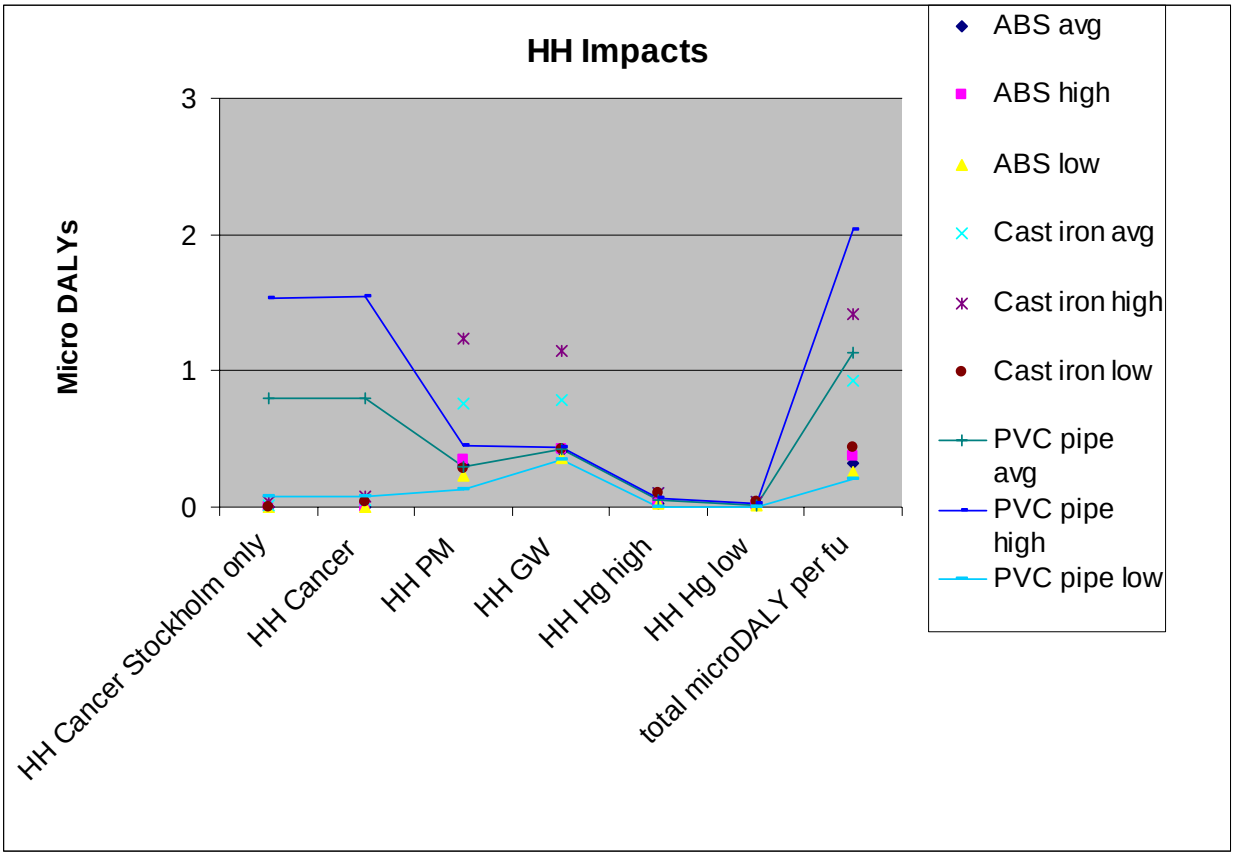

Figure 4-12: Pipe LCA results with EOL but no Occupational

For notes on how to read this figure, see Figure 4-11.

Human Health Impacts – Cradle through Use Plus End-of-Life and Occupational Effects

Adding the impacts to health from exposures of workers in the production and installation of the products leads to the results shown in the figure below. These are the most comprehensive human health results in terms of the scope of the life cycle model. The resulting conclusions are that for overall human health impacts, cast iron pipe is worse than PVC for the maximum and average cases. The determination of the worst material among alternatives studied is sensitive to the uncertainties about emissions factors in the life cycle and risk modeling of the studied systems. For cancer-related impacts, the maximum and average PVC models are clearly worse than the other materials.

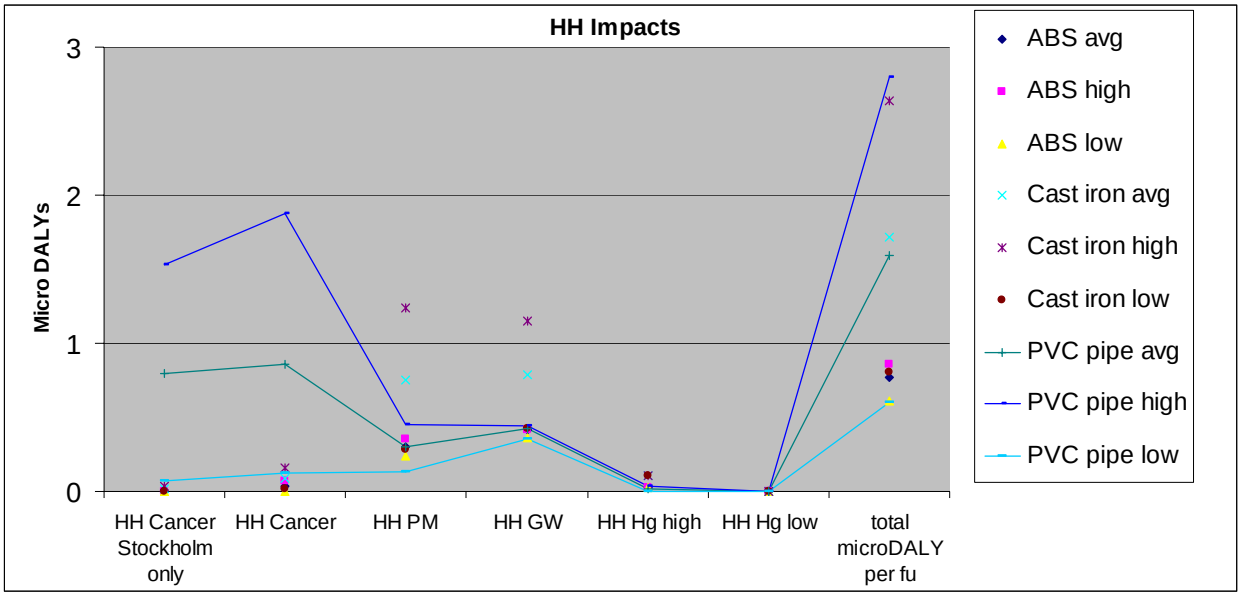

Figure 4-13: Pipe LCA results with EOL and Occupational

For notes on how to read this figure, see Figure 4-11.

Human Health Impacts – Human Mortality Results in Perspective

The figure below helps summarize the findings on the human health mortality impacts of pipe life cycles, related to differences among the materials, among low, middle and high-end estimates on uncertain emissions and exposures, *and* between occupational cancer risks, end-of-life impact, cradle-through-use cancer and cradle-through-use energy-related emissions (global climate change, particulate, and mercury impacts).

Interpretive conclusions which the figure helps clarify include:

- ABS pipe appears better than the other two options in relation to life cycle mortality risks for the middle and high-end exposure/emission scenarios, while the three materials are quite close in expected impacts for the low-end exposure/emission scenarios.
- Cradle-through-use cancer mortality risks are negligible compared to the other risks in the life cycle (they are almost invisible in this figure)
- Energy-related emission impacts are the most important category of mortaltiy impacts for cast iron and ABS pipe across the emission/exposure scenarios.
- End-of-life impacts are the most important for PVC under the middle and high emission/exposure scenarios, while they account for a small share of the life cycle mortality risk for ABS and cast iron.

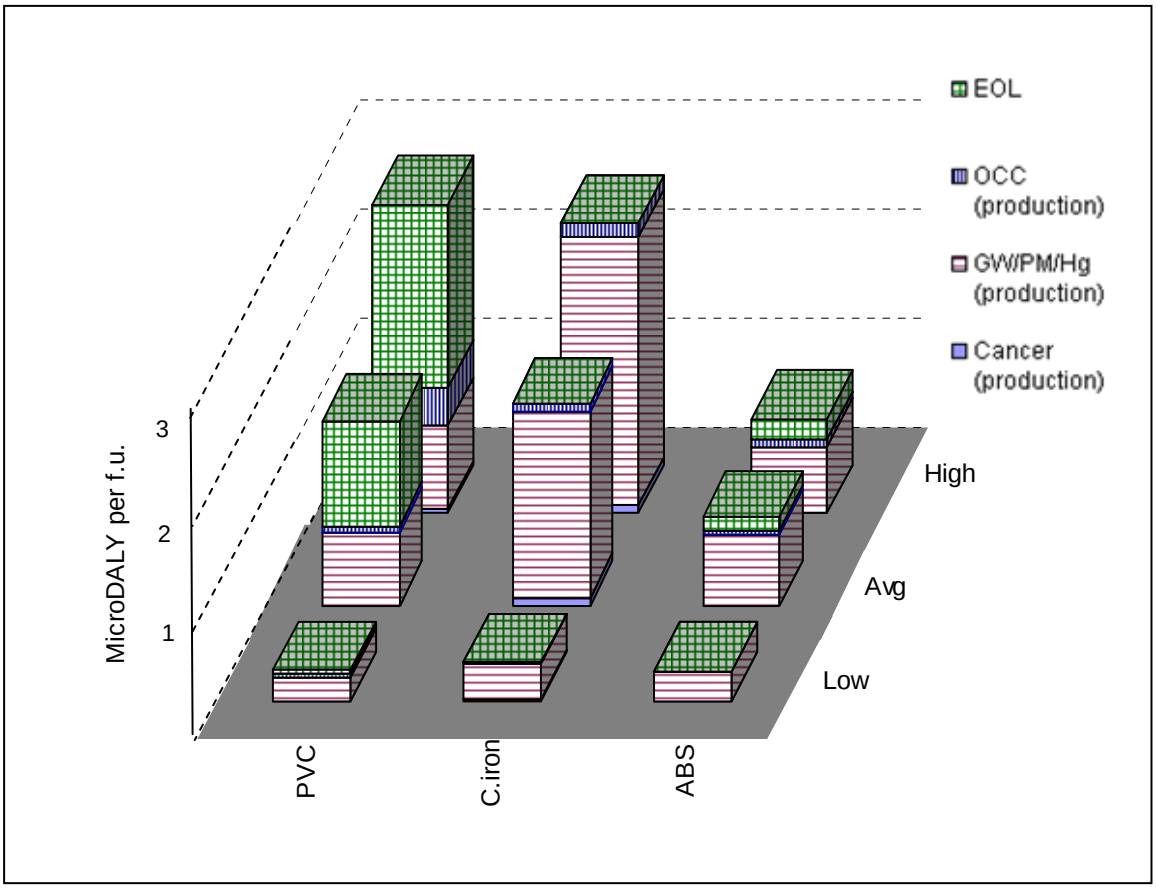

Figure 4-14: High, average and low estimates of human health impacts (microDALY per functional unit) by life cycle stage – Pipe

Human Health Impacts – Sensitivity Analysis of Uncertainties in Global Climate Change and Particulate Impacts

The results and conclusions are tested for their sensitivity to uncertainties in the impact modeling for global climate change and particulates in the figure below. Under low-end estimates for the expected health impacts of pollution in both of these impact categories, PVC average and cast iron average are very close; ABS average remains clearly superior to both of them.

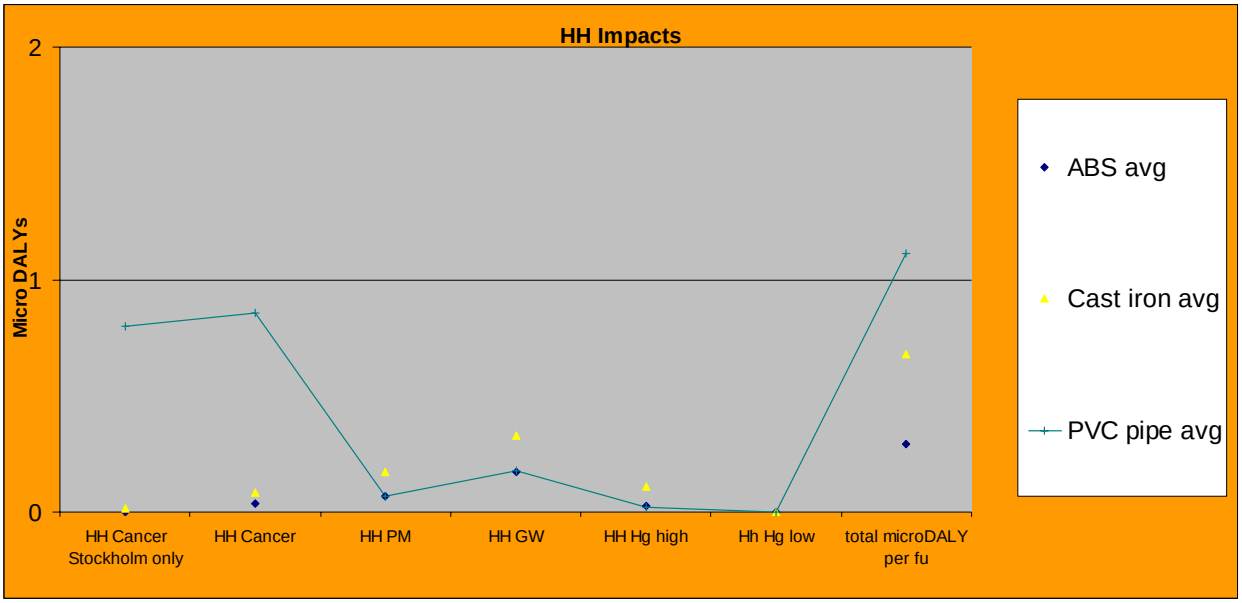

Figure 4-15: Sensitivity analysis for pipe using average emission levels and low-end assumptions for impacts of global climate change and particulates, with EOL and Occupational

Human Health Impacts – Hazard Index Results for Human Health Morbidity Risks

Sums of the total "hazard index" related to human health toxicity impacts for non-carcinogenic effects are summarized in the figure below. For these pollutants and impact pathways, cast iron is worst among the alternatives studied, followed by ABS, with PVC the best.

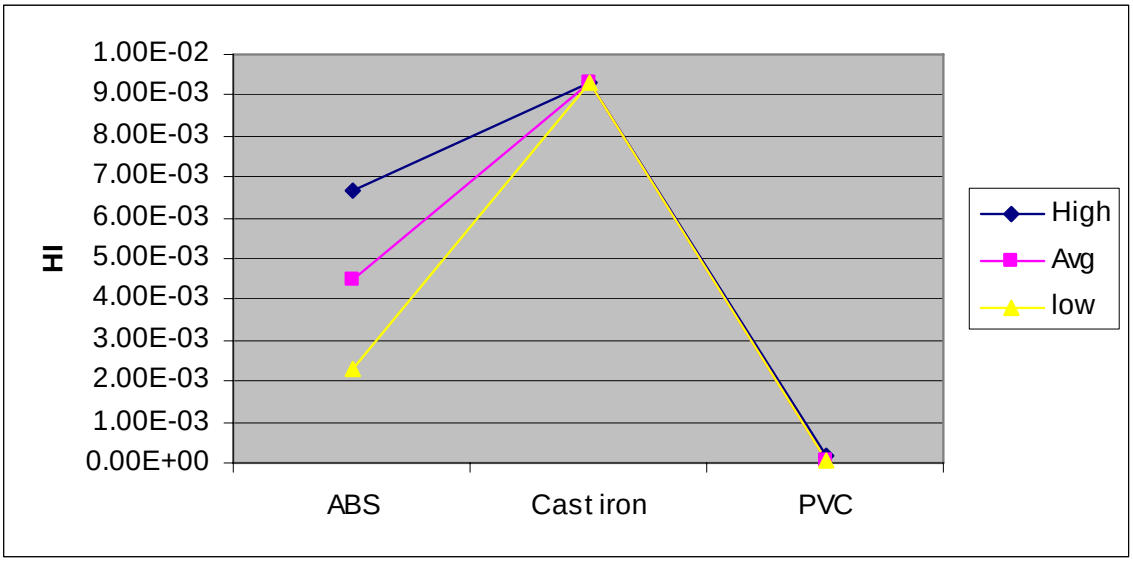

Figure 4-16: Pipe Morbidity "Hazard Index" impacts with EOL and Occupational

Notes on how to read the Human Health Morbidity Figures:

How does the hazard index ("HI") shown in the figure above differ from the DALY metric used in all other human health results figures? The DALY results reflect mortality risk in the case of cancer and global climate change and mercury exposures, and the bulk of the particulates impacts; total DALY results for particulates include the additional (small) contributions of non-fatal respiratory illness. The HI results, on the other hand, reflect a kind of aggregate measure of toxicity responses to chemical exposures. The reason that the HI results have not been converted to DALYs is that the exposure thresholds are generally based on levels at which the first adverse health effect is observed. But these adverse health effects vary widely in severity, starting with minor irritations. And there may be other potential health effects that occur at higher exposures for some chemicals, and not others. So not only do the most sensitive health effects vary from chemical to chemical, but the total possible adverse health affect risk cannot be captured in the HI. In our judgment, the majority of the total expected health impacts are captured in the DALY results, to which the additional impacts captured in the HI results represent a minor adjustment. However, the HI results are presented as well, for purposes of completeness.

Notes on Uncertainty for Pipe Findings

Cancer risks for PVC and cast iron pipe generally varied by one order of magnitude between high and low estimates. Risks for ABS pipe differed by two orders of magnitude. The non-cancer HI for cast iron pipe was affected primarily by cadmium exposure and did not change between the low- and high-estimate. The difference in the two cancer risk estimates was due to the assumption that exposure to coke oven emissions was controlled in the low estimate, but not in the high estimate. Risk estimates for both PVC and ABS pipe were driven by the manufacture of the resin. The same assumptions for exposure to thermal degradation products made for vinyl siding were made for PVC pipe. It was assumed that exposure to barium heat stabilizers would occur in the manufacture of PVC pipe; further, exposure to dialkyl tins was assumed, but there are no current dose-response data that could be used to estimate potential health risk from these tin compounds.

Forrest et al. (1995) reported data that also indicated low emissions from thermal processing of ABS resin; therefore, the low C2S risk for ABS pipe was based on only exposures during resin manufacture. For the high C2S risk for ABS pipe, emissions data from Contos et al. (1995) on ABS pipe manufacture were used. This difference in exposure to thermoprocessing emissions underlies the two orders of magnitude difference in estimated risk for ABS pipe. Nevertheless, the non-cancer and cancer risks per functional unit are found to be comparable across the different piping materials.

For PVC building materials, we find the manufacture of resin to be overwhelmingly the largest contributor. This is because vinyl chloride monomer (VCM) and ethylene dichloride (EDC), the compound used to make it, are both carcinogens. Further, exposure to both compounds is assumed in the manufacture of PVC resin, even though it is possible any one worker may be exposed to only one compound. While exposure data were available from the Vinyl Institute with regard to VCM, these data were not available for EDC; therefore, exposures were assumed limited by the OSHA PEL (permissible exposure limit) for EDC.

As mentioned previously for window frames, assessment of potential occupational risks from solvents used in the manufacture of vinyl or ABS resin were not evaluated for pipe. It is acknowledged that this omission may result in artificially low risk estimates, which would cause an unfair comparison to cast iron. However, Health Hazard Evaluations in the NIOSH database were queried for PVC and ABS manufacture. None were identified that involved CNS or respiratory effects in workers that could be attributable to these solvents. Epidemiological studies of vinyl building products have focused predominantly on exposures to VCM and EDC and not solvent cleaners. Therefore, this omission is not thought to under-represent the relative risk of PVC and ABS pipe compared to cast iron pipe, or in PVC building materials compared to the alternatives.

4.4 Siding

4.4.1 Summary

Environmental Impacts. Aluminum siding is often the worst material among alternatives studied relative to environmental impact categories.

Human Health Impacts: For cancer impacts alone, PVC is clearly the worst material among alternatives studied, especially when the system scope includes end of life impacts. For human health as a whole, results depend on emission factor uncertainties. For the most comprehensive models, PVC and aluminum vie for worst among alternatives studied, depending on emission and lifetime uncertainties.

Table 4-3: Ranking of Materials by Scale of Impacts – Siding Listed from Greatest Impact or "Worst" (1)

Rankings are considered "tied" when the difference from the next ranked material is less than 20%.

Impact Category Environment	Cradle-through-use, environmental pathways	Plus end-of-life with accidental burning
Environment*	1. Aluminum high (1,1,1,1,1,3,1) 2. Aluminum avg (2,4,10,3,2,7,2) Wood high (6,2,5,2,6,5,4) FiberCement high (3,6,2,6,3,1,7) FiberCement avg (5,7,3,7,4,2,8) FiberCement low (7,8,4,8,5,4,9) 7. Wood avg (9,3,7,4,11,6,5) Aluminum low (11,9,12,9,7,12,3) ***PVC high*** (4,10,6,10,8,8,10) ***PVC avg*** (8,11,9,11,9,10,11) Wood low (12,5,8,5,12,9,6) ***PVC low*** (10,12,11,12,10,11,12)	1. Aluminum high (1,1,2,1,1,2,1) 2. Aluminum avg (3,4,10,3,2,7,2) Wood high(7,2,1,2,6,5,4) FiberCement high(4,6,3,6,3,1,8) FiberCement avg, (5,7,4,7,4,3,9) FiberCement low(6,8,6,8,5,4,10) Wood avg(9,3,5,4,11,6,5) ***PVC high***(2,10,7,10,8,8,7) Aluminum low (11,9,12,9,7,12,3) ***PVC avg***(8,11,9,11,9,10,11) Wood low (12,5,8,5,12,9,6) ***PVC low*** (10,12,11,12,10,11,12)

Impact Category Human Health	Cradle-through-use, environmental pathways	Plus end-of-life with accidental burning	Plus occupational impacts including installation
Cancer **	1. Aluminum high, ***PVC high,*** Aluminum avg 2. FiberCement high, Wood high, Aluminum low 7. FiberCement avg, Wood avg 9. Wood low, FiberCement low *11.**PVC avg*** ***12. PVC low***	1. ***PVC high*** 2. ***PVC avg*** 3. ***PVC low,*** Aluminum high, Wood high, Aluminum avg 7. Wood avg 8. FiberCement high 9. Aluminum low, FiberCement 11. Wood low 12. FiberCement low	1. ***PVC high*** 2. ***PVC avg*** 3. ***PVC low*** 4. Aluminum high 5. Aluminum avg 6. Wood high 7. Wood avg 8. FiberCement high 9. Aluminum low, FiberCement avg 11. Wood low 12. FiberCement low
Total human health***	1. Aluminum high 2. Aluminum avg, FiberCement high, FiberCement avg, Wood high, FiberCement low, Aluminum low 8. ***PVC high,*** ***PVC avg,*** ***PVC low,*** 11. Wood avg 12. Wood low	1. Aluminum high, ***PVC high*** 3. Aluminum avg, Wood high, ***PVC avg,*** F.Cement high, F.Cement avg, F. Cement low, Aluminum low, 10. ***PVC low,*** Wood avg, 12. Wood low	1. ***PVC high,*** Aluminum high 3. Aluminum avg, ***PVC avg,*** Wood high, F.Cement high, F.Cement avg, F.Cement low, Aluminum low 10. ***PVC low*** 11. Wood avg, 12.Wood low

* Note: rankings with respect to each separate environmental impact category are presented in parenthesis, with the following impact category order: acidification, eutrophication, smog, ozone depletion, global climate change, fossil fuel depletion, ecotoxicity). The materials appear in this table in the order of their average normalized performance across the seven impact categories. The order of appearance does *not* indicate an overall environmental score, since such an overall score would require value-based weighting across the impact categories.

** Includes cancers from exposure to chemicals and metals

***Includes cancer from row above plus effects of global climate change, particulates, mercury

4.4.2 Environmental impacts

Environmental impact results are summarized in the two figures on this page. Because the results for each impact category are measured in different units, the figures present results that have been normalized by dividing the results in each impact category by the results for the highest-impact material and emissions estimate. The results show the aluminum siding to be the worst material among alternatives studied for many categories, while fiber cement is the worst for fossil fuel depletion. When end-of-life is included, wood is the worst among alternatives studied for the smog effect.

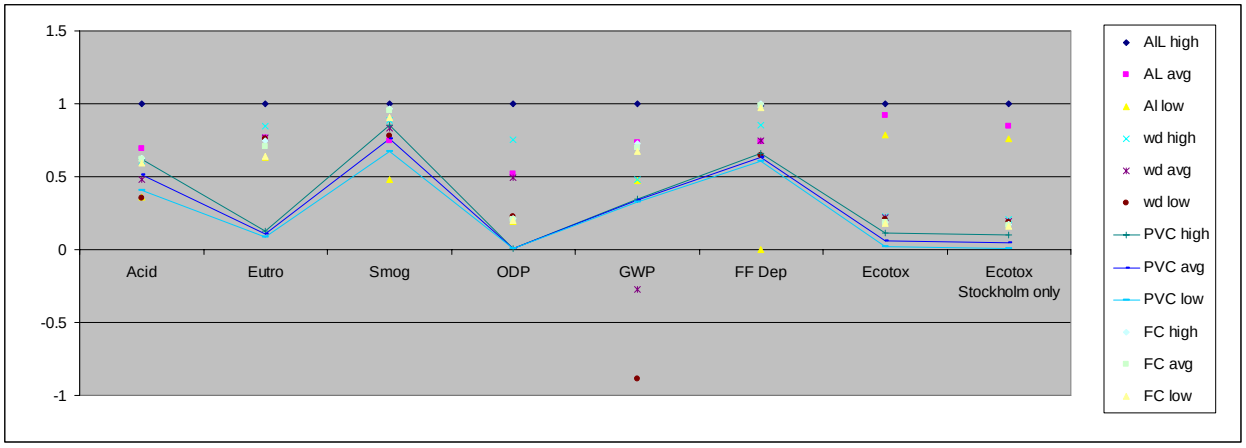

Figure 4-17: Siding environmental impacts, normalized to highest impact per category, no EOL

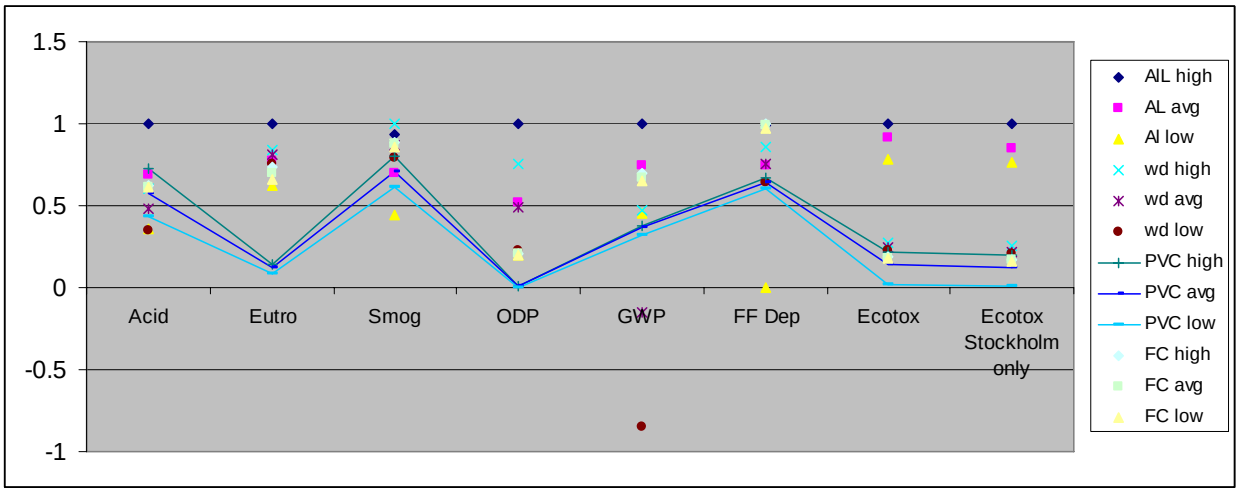

Figure 4-18: Siding environmental impacts, normalized to highest impact per category, with EOL

4.4.3 Human health impacts

Human Health Impacts – Cradle through use

The results in the figure below lead to the conclusions that aluminum and PVC are worst for cancer, while aluminum and fiber cement are worst among alternatives studied for overall health impacts.

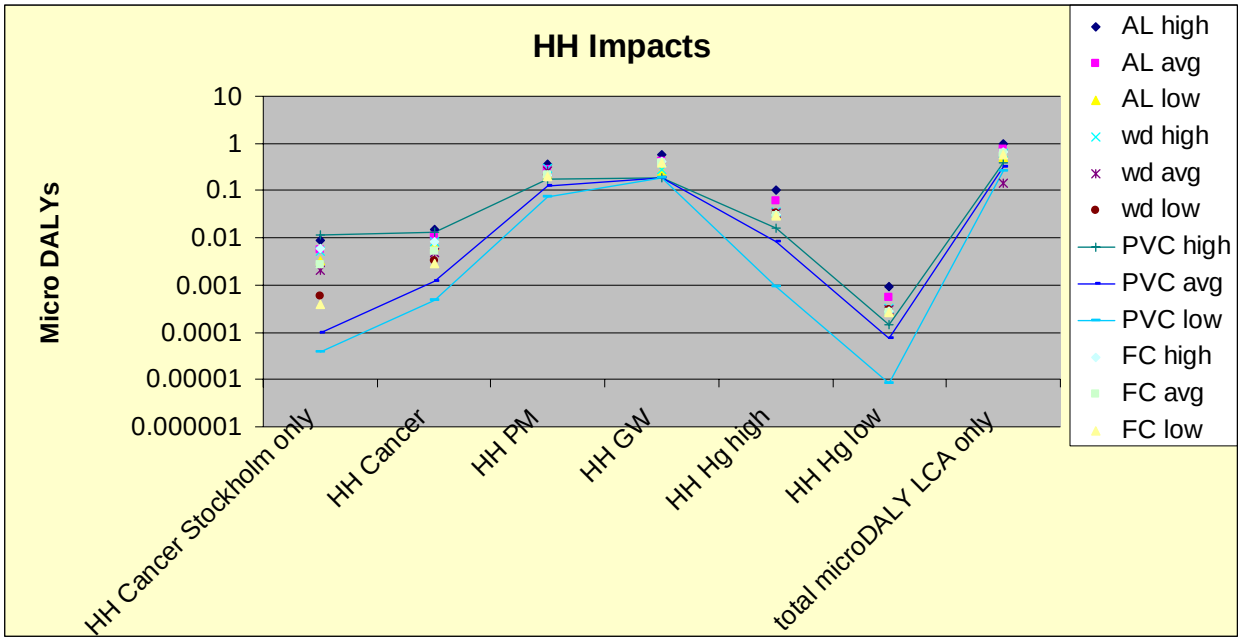

Figure 4-19: Siding LCA results cradle-through use, no EOL, no Occupational

Notes on how to read the Human Health Impact Figures:
For each material alternative there are three sets of results, corresponding to combinations of assumptions on uncertainties in emission factors in the life cycle modeling
"HH" means Human Health; the units of all of these impacts are "micro-DALYs", or millionths of a DALY, where one DALY is one disability-adjusted life-year. In our analysis these DALYs represent expected life years lost due to premature death; the morbidity impacts of particulate exposures are the only non-mortality impact, and are a small share of the DALYs due to particulate exposures.
Cancer impacts reflect population cancer risk associated with exposure to chemical and metal carcinogens. The first category of cancer impacts (on the far left) are the total cancer risks from strictly the set of chemicals addressed by the Stockholm convention; they are a subset of the total cancer impacts displayed as "HH Cancer."
"PM" (Particulate matter) impacts are associated with releases of primary particulates and releases of nitrogen oxides and sulfur oxides which react in the atmosphere to form particulates. "GW" (Global climate change) impacts are estimates of potential human health consequences of global climate change due to releases of greenhouse gasses
"Hg high and HH Hg low": potential neuro-toxicological impacts of mercury emissions under high and low assumptions related to uncertainties in magnitude of release quantities and impact pathways.
Total Micro DALY: The sum of the cancer, particulates, global climate change, and mercury-related impacts.

Human Health Impacts – Cradle through Use Plus End-of-Life

The figure below shows the results after adding the emissions from end-of-life processing, including the risk of emissions from accidental landfill fires and from backyard burning. These results can be displayed using a linear scale on the vertical axis, because the cancer-related impacts (from end-of-life emissions) are no longer orders of magnitude lower than the expected impacts from particulates and global climate change. These results lead to the conclusions that PVC pipe is worse than alternatives studied for cancer-related impacts, while aluminum high and PVC high are tied for last place for human health over all; aluminum average is worse than wood and PVC average, but the differences are under one order of magnitude.

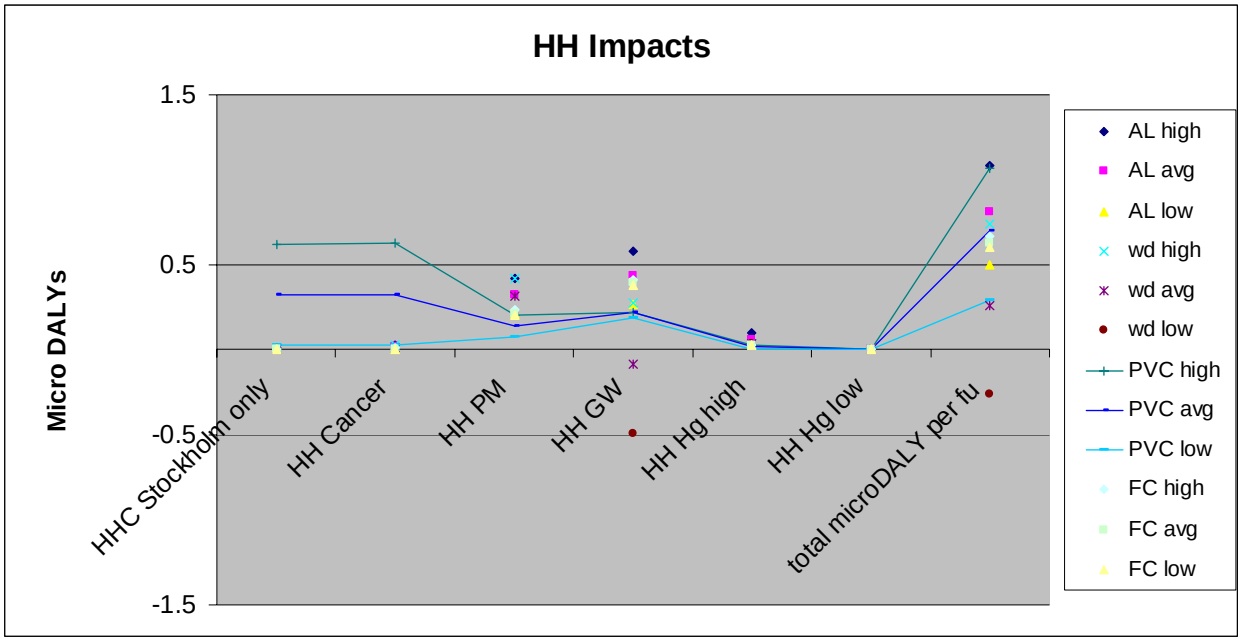

Figure 4-20: Siding LCA results with EOL but no Occupational

For notes on how to read this figure, see Figure 4-19.

Human Health Impacts – Cradle through Use Plus End-of-Life and Occupational Effects

Adding the impacts to health from exposures of workers in the production and installation of the products leads to the results shown in the figure below. These are the most comprehensive human health results in terms of the scope of the life cycle model. The resulting conclusions are that for overall human health impacts, PVC high is worst among alternatives studied, followed by the aluminum high and average results; thus, the determination of the worst material depends on the uncertainties about emissions factors in the life cycle and risk modeling of the studied systems. For cancer-related impacts, all three PVC models are worse than all of the models of the other materials studied.

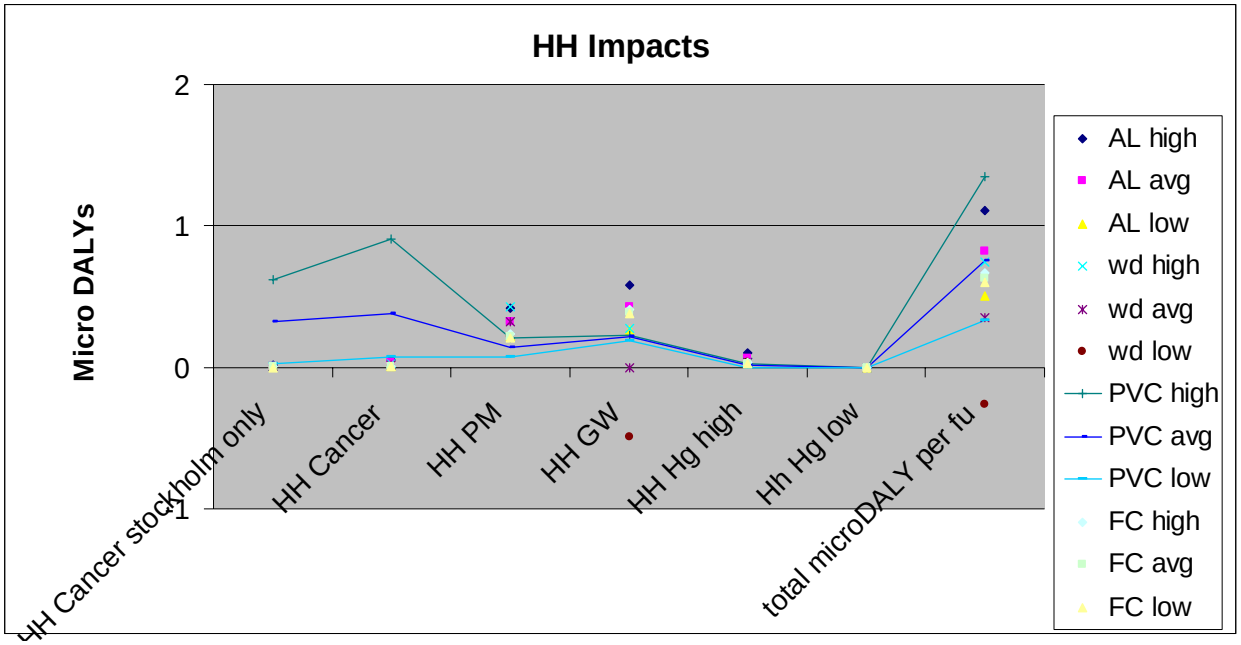

Figure 4-21: Siding LCA results with EOL and Occupational

For notes on how to read this figure, see Figure 4-19.

Human Health Impacts – Human Mortality Results in Perspective

The figure below helps summarize the findings on the human health mortality impacts of siding life cycles, related to differences among the materials, among low, middle and high-end estimates on uncertain emissions and exposures, *and* between occupational cancer risks, end-of-life impact, cradle-through-use cancer and cradle-through-use energy-related emissions (global climate change, particulate, and mercury impacts).

Interpretive conclusions which the figure helps clarify include:

- No single material performs consistently best among alternatives studied in relation to mortality risk across the exposure/emission scenarios, Wood is best for the low and average scenarios, while fiber cement is roughly equal with wood under the high-end emissions/esposure scenario.
- Cradle-through-use cancer mortality risks are negligible compared to the other risks in the life cycle (they are almost invisible in this figure)
- Energy-related emission impacts are the most important category of mortality impacts for wood, aluminum, and fiber cement siding, regardless of the emission/exposure scenario.
- End-of-life impacts are the most important for PVC under the middle and high emission/exposure scenarios, while they account for a small share of the life cycle mortality risk for most other materials for most scenarios (the only exception being the average scenario for wood.
- Occupational risks account for roughly a fourth of the total mortality risk for PVC siding in the high-exposure scenario.

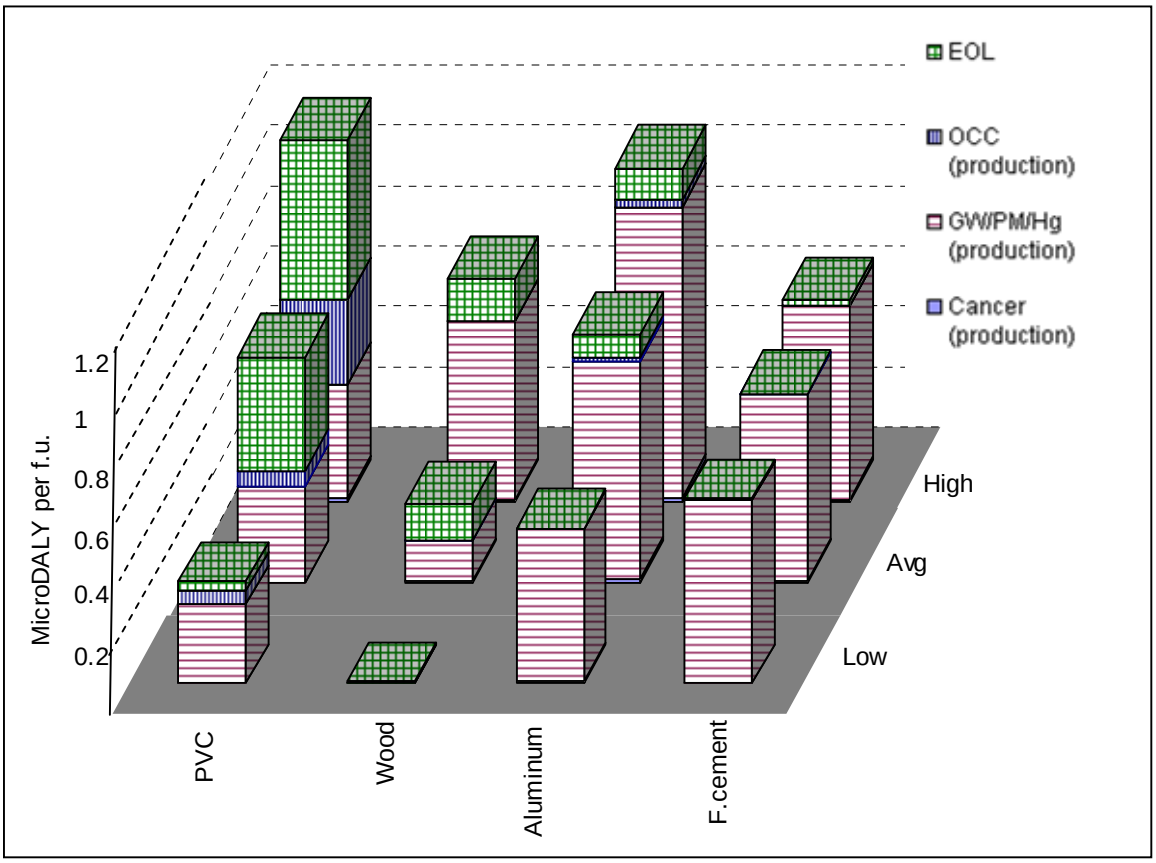

Figure 4-22: High, average and low estimates of human health impacts (microDALY per functional unit) by life cycle stage – Siding

Human Health Impacts – Sensitivity Analysis of Uncertainties in Global Climate Change and Particulate Impacts

The results and conclusions are tested for sensitivity to uncertainties in the modeling for global climate change and particulates in the figure below. Under low estimates for expected health impacts of pollution in these impact categories, PVC average is worse than cast aluminum average.

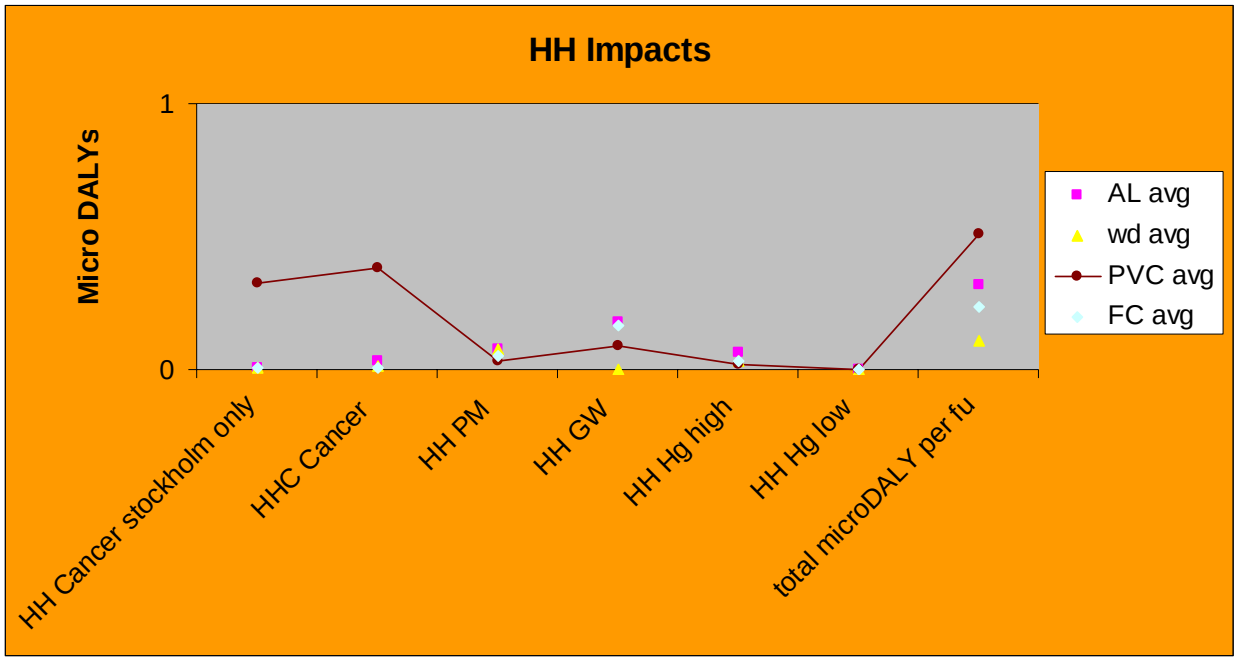

Figure 4-23: Sensitivity analysis for siding using average emission levels and low-end assumptions for impacts of global climate change and particulates with EOL and Occupational

The results in the following figure test sensitivity of the conclusions to the differences in estimated durability of the materials. In place of the assumptions about lifetimes (80 for Aluminum, 50 for Fiber Cement, 40 for Vinyl, and 35 for Wood) all lifetimes are set to 50 years. Under these assumptions, aluminum high and average are worse than PVC high; aluminum low is worse than PVC average, and PVC high and low dominate cancer-related impacts.

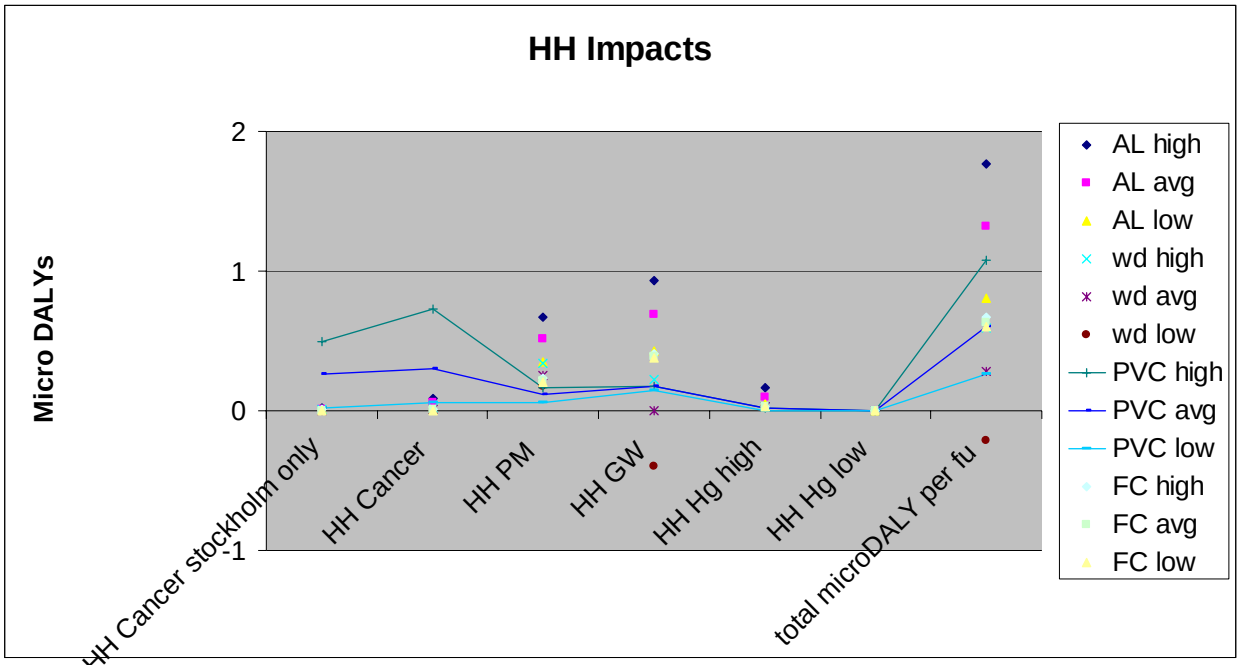

Figure 4-24: Siding LCA results with EOL and Occupational, with all lifetimes equal to 50 years, instead of 80 for Aluminum, 40 for Vinyl, and 35 for Wood

Human Health Impacts – Hazard Index Results for Human Health Morbidity Risks

Sums of the total "hazard index" related to human health toxicity impacts for non-carcinogenic effects are summarized in the figure below. For these pollutants and impact pathways, aluminum tends to be worst among alternatives studied, followed by PVC, with the conclusions depending somewhat on the emission factor uncertainties.

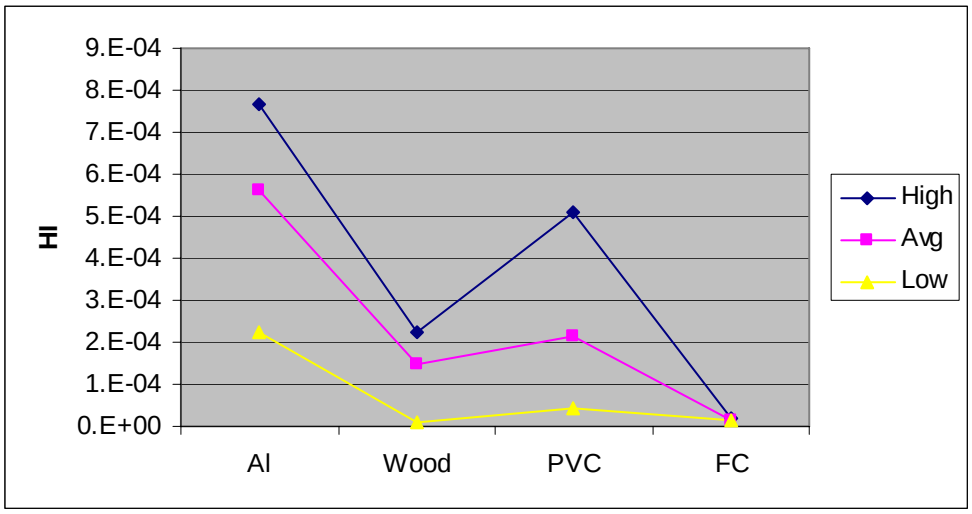

Figure 4-25: Siding Morbidity "Hazard Index" impacts with EOL and Occupational

Notes on how to read the Human Health Morbidity Figures:

How does the hazard index ("HI") shown in the figure above differ from the DALY metric used in all other human health results figures? The DALY results reflect mortality risk in the case of cancer and global climate change and mercury exposures, and the bulk of the particulates impacts; total DALY results for particulates include the additional (small) contributions of non-fatal respiratory illness. The HI results, on the other hand, reflect a kind of aggregate measure of toxicity responses to chemical exposures. The reason that the HI results have not been converted to DALYs is that the exposure thresholds are generally based on levels at which the first adverse health effect is observed. But these adverse health effects vary widely in severity, starting with minor irritations. And there may be other potential health effects that occur at higher exposures for some chemicals, and not others. So not only do the most sensitive health effects vary from chemical to chemical, but the total possible adverse health affect risk cannot be captured in the HI. In our judgment, the majority of the total expected health impacts are captured in the DALY results, to which the additional impacts captured in the HI results represent a minor adjustment. However, the HI results are presented as well, for purposes of completeness.

Notes on Uncertainty for Siding Findings

Risks for vinyl siding per functional unit varied by one order of magnitude between the low-end and high-end estimates. Differences in the risk estimates occurred based on assumptions made regarding exposure to VCM (assumed below detection limit for low cradle to site (C2S) exposure, at the exposure level quoted by Vinyl Institute [2004], and at the OSHA PEL level—these are identified in the resulting graphs as "C2S low," "C2S likely," and "C2S high," respectively). Both non-cancer and cancer risk estimates were dominated by the contribution of resin manufacture, as exposures to toxicants were considered well-controlled during the thermal processing phase. Based on data from Forrest et al. (1994), compounds emitted from thermal processing of vinyl resin were low and represented negligible risks.

Risk estimates for manufacture of aluminum siding differed at the low and high exposure levels based on assumptions of exposure to coke oven emissions and polycyclic aromatic hydrocarbons (PAHs). At the low level, it was assumed that exposure to coke oven emissions were controlled through appropriate respirators, although exposure to aluminum and fluorine was still assumed to occur. At the high level, exposure to coke oven emissions at the OSHA PEL level was assumed in the absence of appropriate exposure data, and PAH values were modeled using total PAH data from Healy et al. (2001). Both non-cancer HI values and cancer risk estimates per functional unit were found to be comparable between vinyl and aluminum siding.

It was not possible to develop risk estimates for wood siding at this time as these building materials lack dose-response data for adverse effects in occupational workers. It is well documented that exposure to wood dust in various industries is linked to the incidence of nasal cancers (Goldsmith and Shy, 1988; Teschke et al., 1999), but data were unavailable that might indicate the increased relative risk for those exposed to wood dust in the manufacture of wood siding.

With regard to fiber cement siding, exposure to components of Portland cement (e.g., silica dusts) is associated with increased incidences of bronchitis and other respiratory complaints (Alvear-Galindo et al., 1999), as well as lung cancer and potentially stomach or colon cancer (Jakobsson et al., 1994; Smailyte et al., 2004). As with wood dust, however, dose-response data were lacking, both for those involved in manufacture of, and installation of fiber-cement siding.

Latex paint is recommended for use on both wood and fiber-cement siding. Manufacture of latex paint involves exposure to several different toxic compounds (including the manufacture of the acrylic resin that forms the basis of the paint). However, exposure to toxic compounds in both the manufacture and use of latex paint was not assessed in this report.

For PVC building materials, we find the manufacture of resin to be overwhelmingly the largest contributor. This is because vinyl chloride monomer (VCM) and ethylene dichloride (EDC), the compound used to make it, are both carcinogens. Further, exposure to both compounds is assumed in the manufacture of PVC resin, even though it is possible any one worker may be exposed to only one compound. While exposure data were available from the Vinyl Institute with regard to VCM, these data were not available for EDC; therefore, exposures were assumed limited by the OSHA PEL (permissible exposure limit) for EDC.

As with the previous two building materials, risks from exposure to solvents used in the manufacture of vinyl resin were not assessed.

4.5 Resilient Flooring

4.5.1 Summary

Environmental Impacts: Regarding environmental impacts, sheet vinyl (not VCT) performs worst among alternatives studied on all impact categories except eutrophication, for which linoleum is worst.

Human Health Impacts: VCT and sheet vinyl vie for worst among alternatives studied on human health, but they are both worst on human health in most cases compared to the alternative materials, especially cork.

Table 4-4: Ranking of Materials by Scale of Impacts – Flooring Listed from Greatest Impact or "Worst" (1)

Rankings are considered "tied" when the difference from the next ranked material is less than 20%.

Impact Category Environment	Cradle-through-use, environmental pathways	Plus end-of-life with accidental burning
Environment*	1. ***Sheet Vinyl high,*** (1,4,1,1,1,2,1) ***Sheet Vinyl avg,*** (2,5,2,2,2,3,2) ***VCT high,*** (5,8,5,3,7,6,5) Linoleum high (6,1,4,6,6,1,4) 5. ***Sheet Vinyl low,***(5,8,5,3,7,6,5) Linoleum avg, (7,2,7,8,8,4,7) ***VCT avg*** (4,7,6,5,4,12,6) 8. ***VCT low*** (8,9,9,10,5,7,9) 9. Linoleum low (9,3,8,12,9,8,8) 10. Cork high (10,10,10,7,10,9,10) 11. Cork avg (11,11,11,9,11,10,11) 12.Cork low (12,12,12,11,12,11,12)	1. ***Sheet Vinyl high,*** (1,4,1,1,1,2,2) ***Sheet Vinyl avg,*** (2,5,2,2,2,3,3) ***VCT high,*** (3,6,3,4,3,5,1) Linoleum high, (6,1,4,6,5,1,7) ***Sheet Vinyl low*** (5,9,5,3,8,6,5) 6. Linoleum avg, (7,2,7,8,7,4,8) ***VCT avg*** (2,5,2,2,2,3,3) 8. ***VCT low*** (5,9,5,3,8,6,5) 9. Linoleum low (9,3,8,12,9,11,11) 10. Cork high (10,8,10,7,10,7,9) 11. Cork avg (11,11,11,9,11,8,10) 12.Cork low (12,12,12,11,12,12,12)

Impact Category Human Health	Cradle-through-use, environmental pathways	Plus end-of-life with accidental burning	Plus occupational impacts including installation
Cancer **	1. ***Sheet Vinyl high*** 2. ***VCT high*** 3. ***Sheet Vinyl avg, VCT avg*** 5. Linoleum high, ***Sheet Vinyl low***, Cork high ***8. VCT low*** 9. Linoleum avg 10. Cork avg 11. Linoleum low 12. Cork low	1. ***Sheet Vinyl high*** 2. ***Sheet Vinyl avg, VCT high*** 4. ***VCT avg*** ***5. Sheet Vinyl low,*** ***6. VCT low*** 7. Linoleum high 8. Linoleum avg 9. Cork high 10. Cork avg 11. Linoleum low 12. Cork low	1. ***Sheet Vinyl high*** 2. ***VCT high, Sheet Vinyl avg*** 4. ***VCT avg*** 5. ***Sheet Vinyl low*** 6. ***VCT low*** 7. Linoleum high, Cork high 9. Cork avg, Linoleum avg 11. Linoleum low 12. Cork low
Total human health***	1. ***VCT high*** 2. ***VCT avg,*** Linoleum high, ***Sheet Vinyl high*** 5. ***Sheet Vinyl avg,*** Linoleum avg 7. ***VCT low*** 8. ***Sheet Vinyl low,*** Linoleum low 10. Cork high 11. Cork avg 12.Cork low	1. ***VCT high*** 2. ***Sheet Vinyl high*** 3. ***VCT avg, Sheet Vinyl avg*** 5. Linoleum high 6. Linoleum avg 7. ***VCT low*** 8. ***Sheet Vinyl low*** 9. Linoleum low 10. Cork high 11. Cork avg 12. Cork low	1. ***VCT high*** 2. ***Sheet Vinyl high*** 3. ***VCT avg, Sheet Vinyl avg*** 5. Linoleum high 6. Linoleum avg 7. ***VCT low, Sheet Vinyl low*** 9. Linoleum low 10. Cork high 11. Cork avg 12. Cork low

* Note: rankings with respect to each separate environmental impact category are presented in parenthesis, with the following impact category order: acidification, eutrophication, smog, ozone depletion, global climate change, fossil fuel depletion, ecotoxicity). The materials appear in this table in the order of their average normalized performance across the seven impact categories. The order of appearance does *not* indicate an overall environmental score, since such an overall score would require value-based weighting across the impact categories.

** Includes cancers from exposure to chemicals and metals

***Includes cancer from row above plus effects of global climate change, particulates, mercury

4.5.2 Environmental impacts

Environmental impact results are summarized in the two figures on this page. Because the results for each impact category are measured in different units, the figure presents results that have been normalized by dividing the results in each impact category by the results for the highest-impact material and emissions estimate. Sheet Vinyl and VCT tend to be worst among alternatives studied, with the exception of eutrophication and fossil fuel depletion, for which linoleum is the worst.

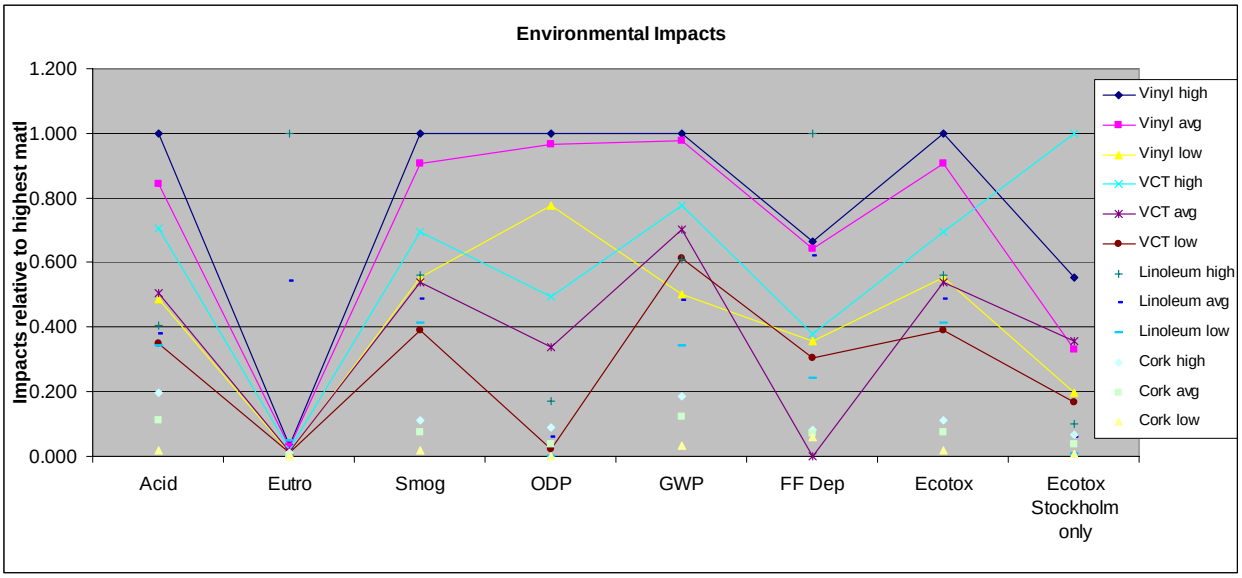

Figure 4-26: Flooring environmental impacts, normalized to highest impact per category, no EOL

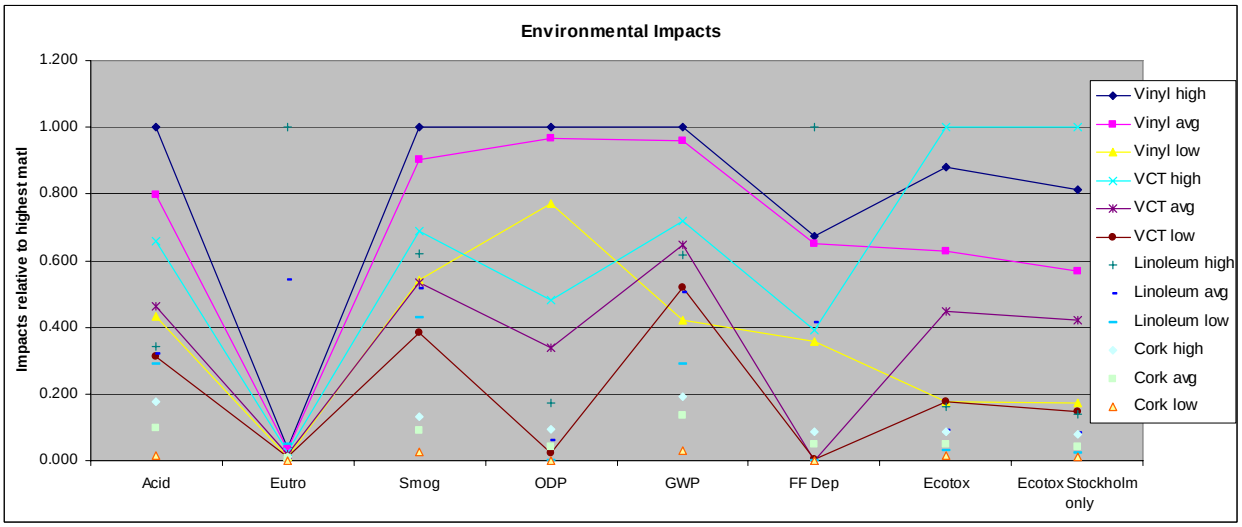

Figure 4-27: Flooring environmental impacts, normalized to highest impact per category, with EOL

4.5.3 Human health impacts

Human Health Impacts – Cradle through use

The results in the figure below lead to the conclusions that VCT and sheet vinyl are worst among alternatives studied relative to human health impacts overall, and relative to the subset of human health impacts via cancer.

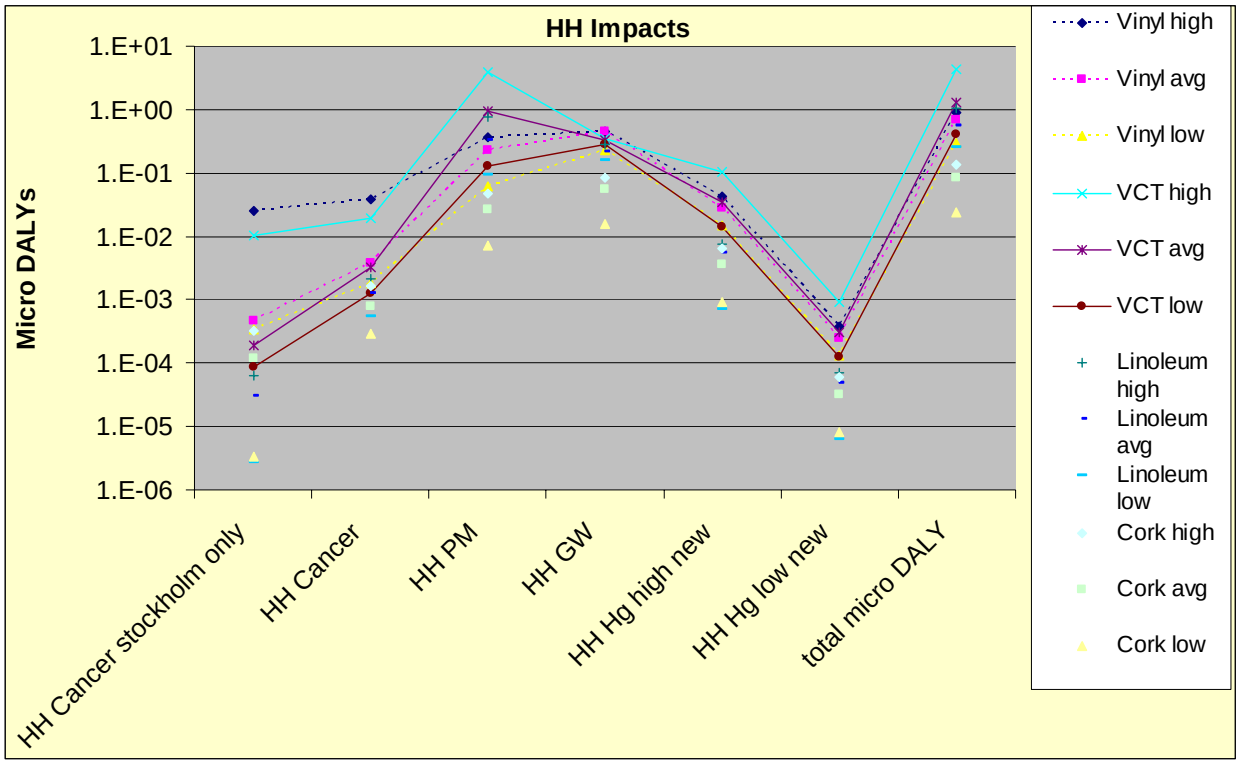

Figure 4-28: Flooring LCA results cradle-through use, no EOL, no Occupational

Notes on how to read the Human Health Impact Figures:
For each material alternative there are three sets of results, corresponding to combinations of assumptions on uncertainties in emission factors in the life cycle modeling
"HH" means Human Health; the units of all of these impacts are "micro-DALYs", or millionths of a DALY, where one DALY is one disability-adjusted life-year. In our analysis these DALYs represent expected life years lost due to premature death; the morbidity impacts of particulate exposures are the only non-mortality impact, and are a small share of the DALYs due to particulate exposures.
Cancer impacts reflect population cancer risk associated with exposure to chemical and metal carcinogens. The first category of cancer impacts (on the far left) are the total cancer risks from strictly the set of chemicals addressed by the Stockholm convention; they are a subset of the total cancer impacts displayed as "HH Cancer."
"PM" (Particulate matter) impacts are associated with releases of primary particulates and releases of nitrogen oxides and sulfur oxides which react in the atmosphere to form particulates. "GW" (Global climate change) impacts are estimates of potential human health consequences of global climate change due to releases of greenhouse gasses
"Hg high and HH Hg low": potential neuro-toxicological impacts of mercury emissions under high and low assumptions related to uncertainties in magnitude of release quantities and impact pathways.
Total Micro DALY: The sum of the cancer, particulates, global climate change, and mercury-related impacts.

Human Health Impacts – Cradle through Use Plus End-of-Life

The figure below shows the results after adding the emissions from end-of-life processing, including the risk of emissions from accidental landfill fires and from backyard burning. These results can be displayed using a linear scale on the vertical axis, as the cancer-related impacts (from end-of-life emissions) are no longer orders of magnitude lower than the expected impacts from particulates and global climate change. These results lead to the conclusions that VCT is worst for health overall among alternatives studied, while sheet vinyl is worst for cancer-related health impacts.

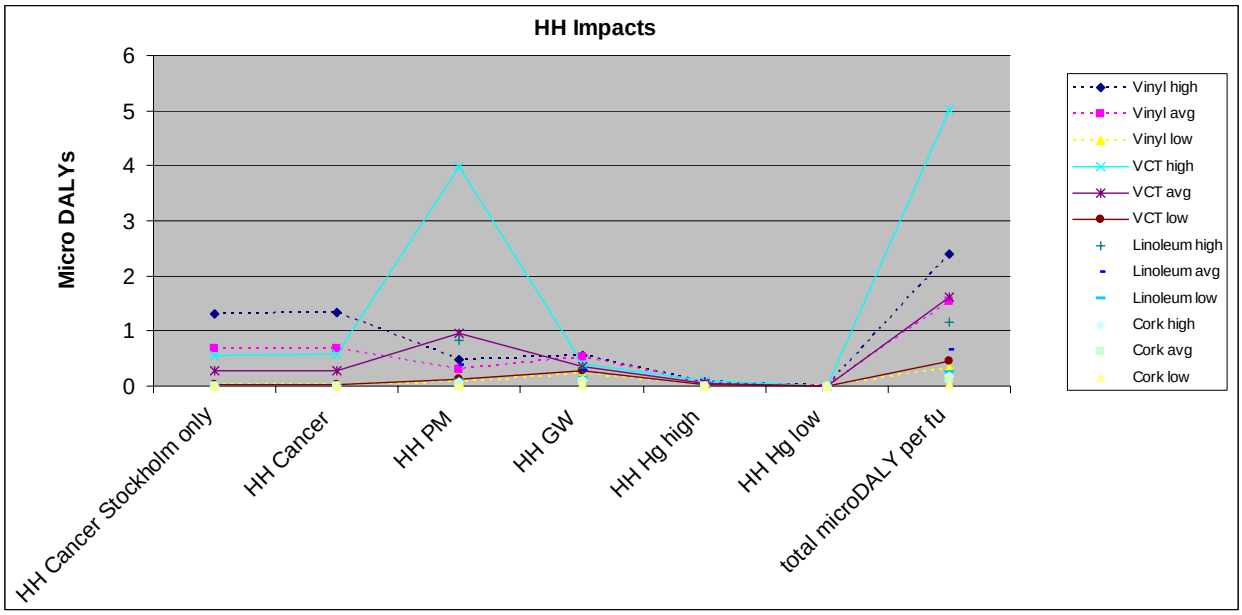

Figure 4-29: Flooring LCA results with EOL but no Occupational
For notes on how to read this figure, see Figure 4-28.

Human Health Impacts – Cradle through Use Plus End-of-Life and Occupational Effects

Adding the impacts to health from exposures of workers in the production and installation of the products leads to the results shown in the figure below. These are the most comprehensive human health results in terms of the scope of the life cycle model. The resulting conclusions are that VCT and sheet vinyl are worst for human health and for the cancer subset of human health impacts among alternatives studied.

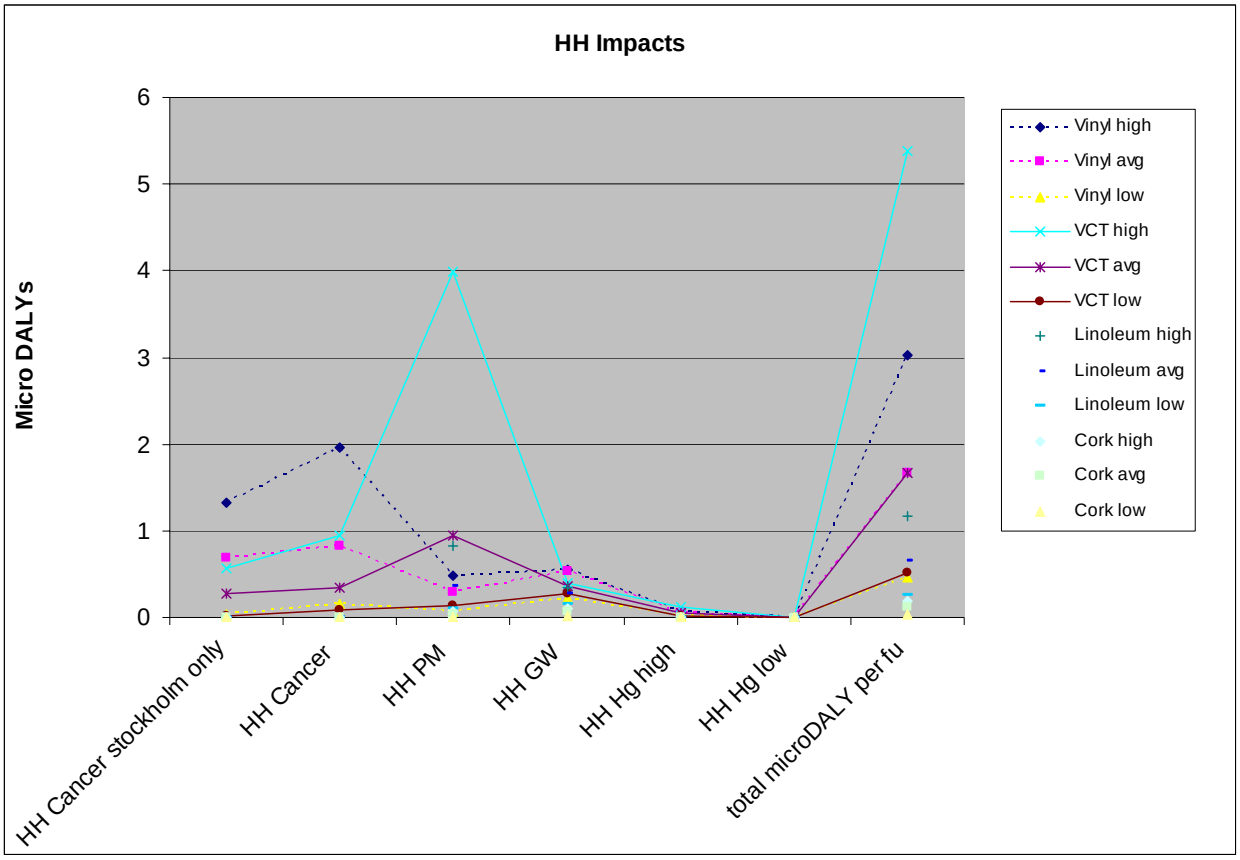

Figure 4-30: Flooring LCA results with EOL and Occupational

For notes on how to read this figure, see Figure 4-28.

Human Health Impacts – Human Mortality Results in Perspective

The figure below helps summarize the findings on the human health mortality impacts of flooring life cycles, related to differences among the materials, among low, middle and high-end estimates on uncertain emissions and exposures, *and* between occupational cancer risks, end-of-life impact, cradle-through-use cancer and cradle-through-use energy-related emissions (climate change, particulate, and mercury impacts).

Interpretive conclusions which the figure helps clarify include:

- Cork performs consistently best in relation to mortality risk across the exposure/emission scenarios and linoleum performs consistently second-best for mortality risk across the exposure/emission scenarios among the alternatives studied.
- Cradle-through-use cancer mortality risks are negligible compared to the other risks in the life cycle (they are invisible in this figure)
- Energy-related emission impacts are the most important category of mortaltiy impacts for linoleum and vinyl composition tile (VCT), regardless of the emission/exposure scenario.
- End-of-life impacts are the most important for sheet vinyl under the middle and high emission/exposure scenarios, while they account for a small share of the life cycle mortality risk for the other materials.

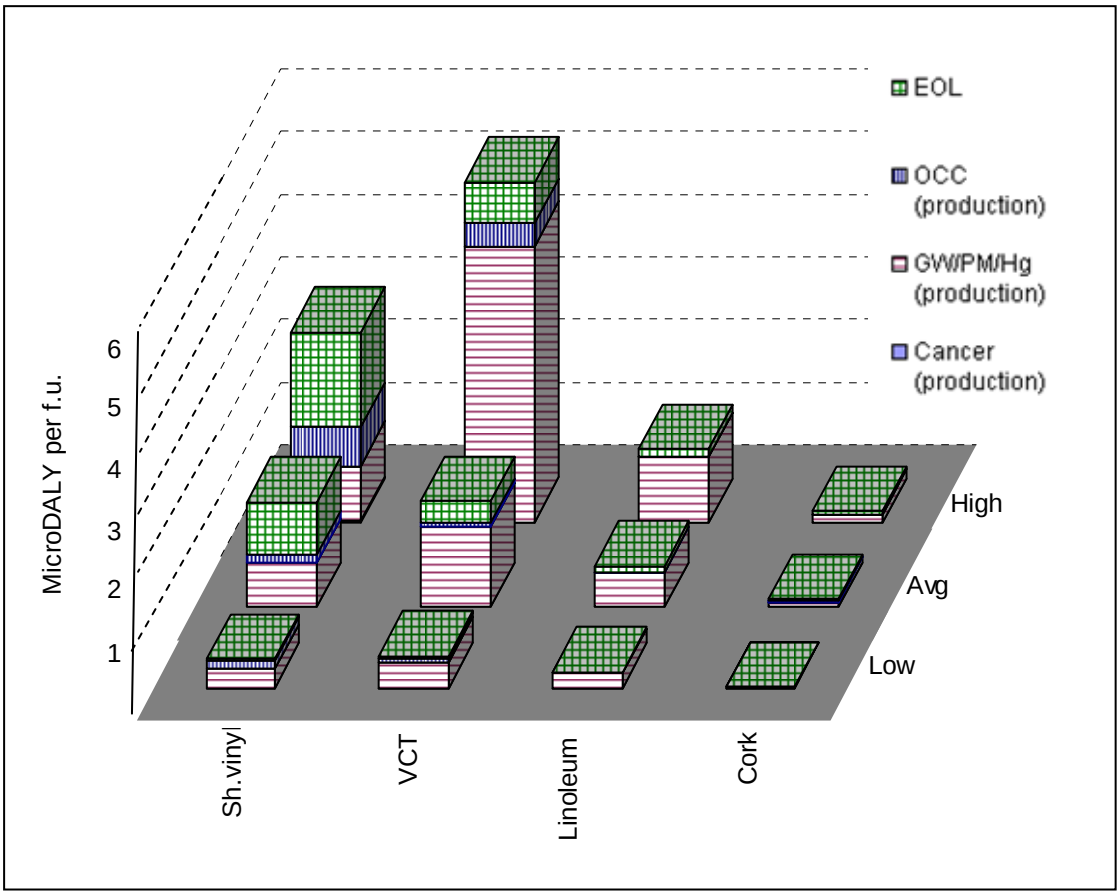

Figure 4-31: High, average and low estimates of human health impacts (microDALY per functional unit) by life cycle stage -- Flooring

Human Health Impacts – Sensitivity Analysis of Uncertainties in Global Climate Change and Particulate Impacts

The results and conclusions are tested for their sensitivity to uncertainties in the impact modeling for global climate change and particulates, in the figure below. Under low estimates for expected health impacts in both of these categories, sheet vinyl and VCT are worse than the other alternatives.

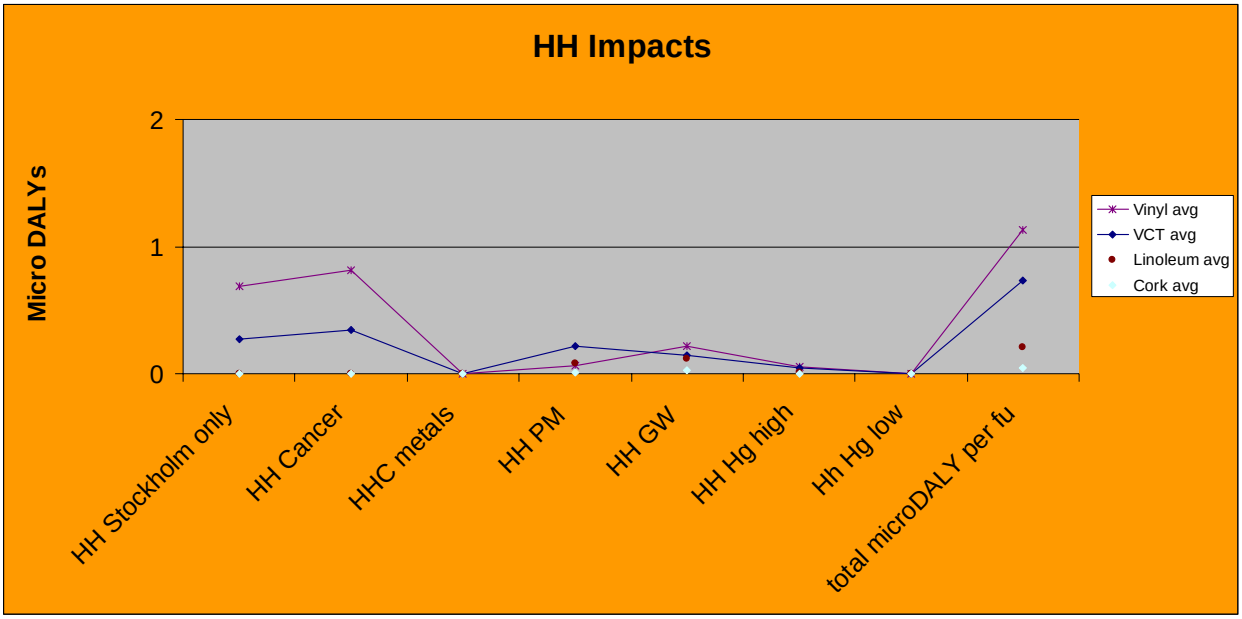

Figure 4-32: Sensitivity analysis for flooring using average emission levels and low-end assumptions for impacts of global climate change and particulates with EOL and Occupational

The results in the figure below test sensitivity of the conclusions about human health to differences in estimated durability of materials. In place of assumptions about lifetimes (15 for vinyl, 18 for linoleum and VCT, 50 for cork) all material lifetimes are set to 18 years. Under these assumptions, VCT is worse than sheet vinyl, and both are worse than alternatives.

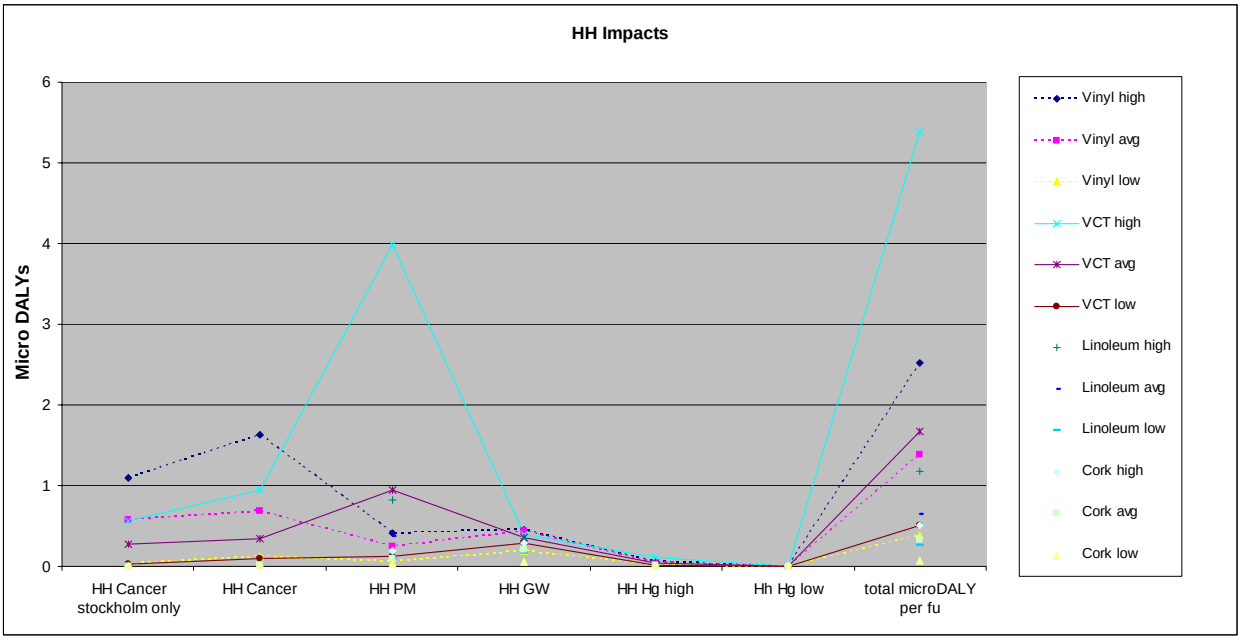

Figure 4-33: Flooring LCA Results with EOL and Occupational, with all lifetimes set to 18 years, rather than 15 for vinyl and 50 for cork

Human Health Impacts – Hazard Index Results for Human Health Morbidity Risks

Sums of the total "hazard index" related to human health toxicity impacts for non-carcinogenic effects are summarized in the figure below. For these pollutants and impact pathways, no material consistently dominates. Vinyl is worst for the low-end emissions estimates, while cork is worst for the medium and high-end estimates among alternatives studied.

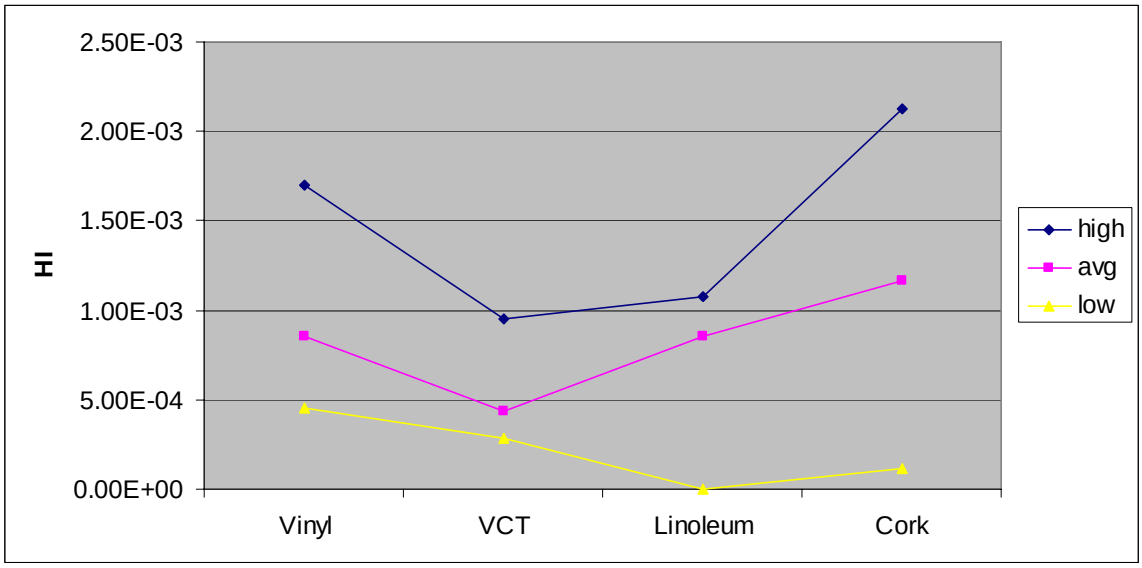

Figure 4-34: Flooring Morbidity "Hazard Index" impacts with EOL and Occupational

Notes on how to read the Human Health Morbidity Figures:
How does the hazard index ("HI") shown in the figure above differ from the DALY metric used in all other human health results figures? The DALY results reflect mortality risk in the case of cancer and global climate change and mercury exposures, and the bulk of the particulates impacts; total DALY results for particulates include the additional (small) contributions of non-fatal respiratory illness. The HI results, on the other hand, reflect a kind of aggregate measure of toxicity responses to chemical exposures. The reason that the HI results have not been converted to DALYs is that the exposure thresholds are generally based on levels at which the first adverse health effect is observed. But these adverse health effects vary widely in severity, starting with minor irritations. And there may be other potential health effects that occur at higher exposures for some chemicals, and not others. So not only do the most sensitive health effects vary from chemical to chemical, but the total possible adverse health affect risk cannot be captured in the HI. In our judgment, the majority of the total expected health impacts are captured in the DALY results, to which the additional impacts captured in the HI results represent a minor adjustment. However, the HI results are presented as well, for purposes of completeness.

Notes on Uncertainties in Flooring Findings

Risk estimates for vinyl flooring and vinyl composition tiles (VCTs) are very comparable due to the similarity of their core compounds, VCM and phthalate plasticizers. The difference in the risk estimates for manufacture of the resin was due to the addition of vinyl acetate to the resin used to manufacture VCTs. Risk estimates for manufacture of the actual flooring were the same for the two flooring materials because it was assumed that these workers would be exposed to the same amounts of VCM and DEHP, in the absence of better exposure data. The primary component of VCT by mass is calcium carbonate, or limestone; risk from exposure to this compound was not assessed for this material, dose-response toxicity data were unavailable. Exposures to this compound are limited based on nuisance dust regulations, which are designed to keep dust in occupational environments to below 10-15 mg/m3. Normalized risk estimates for the different types of vinyl flooring take into account the decreased amount of PVC resin in VCT tiles compared to vinyl sheet.

Risk estimates for linoleum and cork flooring were much lower per functional unit, particularly those for cork. Cradle to site occupational risk estimates for linoleum could not be developed due to lack of exposure data, and the fact that the compounds used to make linoleum are not very toxic (e.g., they lack dose-response data). The construction and end use risk estimates were generated using the CIWMB (2003) emissions study classroom exposure estimates. Cradle to site risk estimates for cork were modeled using the emissions data from Horn et al. (1998) and assuming a standard fabrication plant size and air change rate.

For PVC building materials, we find the manufacture of resin to be overwhelmingly the largest contributor. This is because vinyl chloride monomer (VCM) and ethylene dichloride (EDC), the compound used to make it, are both carcinogens. Further, exposure to both compounds is assumed in the manufacture of PVC resin, even though it is possible any one worker may be exposed to only one compound. While exposure data were available from the Vinyl Institute with regard to VCM, these data were not available for EDC; therefore, exposures were assumed limited by the OSHA PEL (permissible exposure limit) for EDC.

As with the previous three building materials, risks from exposure to solvents used in the manufacture of vinyl resin were not assessed.

5 Findings of Additional Analyses

Sources in the database include a huge amount of information that, for various reasons, could not be integrated into the LCA-risk assessment framework. This includes:

- PVC fires
- Air monitoring and fenceline analyses
- Non-cancer risk from exposure to phthalates

5.1 PVC Fires

Concern has been raised over the potential exposures of the general population and firefighters to the toxic components of gases and smoke arising from combustion and/or pyrolysis of PVC-containing plastics in house and/or commercial building fires. In particular, hydrochloric acid (HCl) can be generated from burning PVC materials, and from those materials before they have ignited. There are accounts in the literature attributing long-term health-effects to exposure HCl (Moisan, 1991; Dyer & Esch, 1976; Colardyn et al, 1976; Markowitz, 1989; Markowitz et al., 1989), as well as descriptions of situations in which people were incapacitated by these gases, reducing their ability to escape from the fire. Several experimental studies have shown that HCl adsorbs onto soot particles (Stone et al., 1973) and water droplets (Stone, 1975), which may increase the penetration and retention of HCl in the lungs.

Highly toxic phosgene gas can also be released from burning PVC, and some hypotheses suggest that as the culprit in some otherwise unexplained health effects observed in firefighters (Brown & Birky, 1980).

None of the scientific literature that we found that included analysis of air samples from building fires showed problematic levels of either HCl or phosgene, however (Treitman et. al, 1980; Bolstan-Johnson, 2000). It remains possible that HCl has the effects described in certain specific situations, but those do not appear to be typical. Phosgene does not persist in the air, so it is possible that it is created under certain conditions and then decomposes prior to being detected. But theoretical models in the literature indicate that even under optimal conditions the concentration of phosgene would not be high enough to cause adverse reactions (Bjerre, 1984).

The risk remains that firefighters may be exposed to highly toxic gases from PVC under certain conditions, but structural fires have many sources of toxic gases so any firefighter not using breathing apparatus would be taking on unnecessary risk, regardless of the specific materials present. Studies suggestion long-term health consequences for firefighters conflict with two others regarding firefighters potentially exposed to PVC (Tashkin et al., 1977; Musk et al., 1982). And at least one large cohort mortality found no significant increase in overall mortality among municipal firefighters compared with a comparable civilian population (Baris et al., 2001).

The literature also includes case reports regarding warehouse fires involving large quantities of PVC (Upshur et al., 2001). These data indicate that a large PVC fire can result in reported symptoms in a locally-exposed community, but immediate symptoms tend not to be serious, do not require medical attention, and are not long-lasting. Nevertheless, long-term effects were not reported, or even looked for, and it is not possible to rule out long-term illness as a result of exposure.

Such fires release dioxins and other compounds that may have long-term effects on human health and the environment, but local impacts do not appear to be long-lasting. One such fire in Hamilton, Ontario, was monitored extensively (Ministry of Environment and Energy, Canada, 1997). While initial levels of dioxin and HCl were predictably high, they dropped off quickly.

A sampling of 20 residential properties performed one month after the fire indicated no measurable residual dioxin contamination of soil and vegetation, which included backyard garden produce. Measured concentrations of dioxin in nearby water returned to normal reported ranges rapidly, although the report did not indicate dioxin levels in the sediments either prior to or after the fire. It is likely that the dioxin did not remain in the water, but adsorbed onto sediments at the bottom of the harbor. Lack of sediment data represents a data gap in the overall estimation of local ecological impacts from this fire, and the fact that the dioxins dispersed doesn't reduce the potential impact of their contribution to exposures elsewhere.

Compared with other plastics, and other combustible materials, PVC may have a beneficial role in reducing injuries in structural fires, as it may reduce the chances of a fire igniting or spreading due to its relatively high ignition temperature. While the toxic gases released from PVC as it heats up prior to ignition represent a potential hazard, in many situations they not concentrated enough to disable or injure people, whereas a fully involved fire would be more hazardous.

In summary, there is evidence that PVC contributes to hazardous conditions in building fires, but insufficient evidence to determine how widespread or consistent a risk that represents, how it relates to the possible fire retarding effect of such materials, and how these factors compare to the other human health and environmental impacts studied in this report.

5.2 Air Monitoring Data and Fenceline Analysis

When assessing potential human health risks resulting from the manufacture of PVC and alternate materials, it is necessary to try to quantify the risks to those living in the vicinity of the manufacturing facilities. The TG evaluated recent ambient monitoring data taken in the area of West Jefferson County, Kentucky, and in Baton Rouge and Calcasieu Parish, LA to determine potential non-cancer and cancer risks from inhalation exposure to vinyl chloride monomer. The monitoring data were obtained from ambient air monitoring stations in the locations discussed above. Appendix I discusses in more detail the locations of the monitoring stations, the proximity of some to local vinyl facilities, the time frame of sample collection, the compounds analyzed, the compounds of interest, and the resultant risk estimates.

5.2.1 Risks from recent air monitoring in Louisville, Kentucky

Data from air sampling rounds taken from January 2003 to November 2005 were used to estimate current potential non-cancer and cancer risks from exposure to vinyl chloride monomer. Data were from five monitoring stations that have been maintained in West Jefferson County: Louisville Firearms Training Center; Ralph Avenue/Campground Rd; Farnsley Middle School; Cane Run Elementary School, and Chickasaw Park. Mean concentrations of VCM and the resultant non-cancer (Hazard Index) and cancer (Integrated Lifetime Cancer Risk) risk estimates are shown in the table below. Both median and mean concentrations are shown; risk estimates are based on the mean concentrations because they were the larger of the two.

Table 5-1: Risk Estimates for Exposure to Airborne Vinyl Chloride Monomer, KY, 2003-2005

	Louisville Firearms	Ralph Ave.	Farnsley MS	Chickasaw Park	Cane Run Elem. S.	Risk Limits Kentucky State
VCM (median), ppb_v	0.05					
VCM (mean), ppb_v	0.25	0.06	0.06	0.10	0.17	
VCM (mean), mg/m3	0.0006	1.7E-4	1.6E-04	2.6E-04	4.3E-04	
Hazard Index[a]	6.29E-03	1.64E-03	1.57E-03	2.61E-03	4.35E-03	1.0
ILCR[b]	5.53E-06	1.44E-06	1.38E-06	2.29E-06	3.82E-06	1.0E-06

[a]HI obtained by dividing the mean concentration of VCM (mg/m3) by the Reference Concentration of 0.1 mg/m^3 (U.S. EPA, 2004a)

b ILCR obtained by multiplying the mean concentration of VCM by the Unit Risk Estimate of 8E-03 $(mg/m^3)^{-1}$

–

The data above indicate that mean concentrations measured over the sampling period (January 2003-November 2005) do not result in a non-cancer hazard index that exceeds 1.0 (state and federal limit); but all concentrations result in cancer risk estimates that exceed the limit of 1.0E-06.

5.2.2 Air monitoring data - Louisiana

The TG used air monitoring data (ca 1999-2003) from the Louisiana Ambient Air Monitoring Program for Vinyl Chloride (Sage, 2004). Although the report contains data from 14 monitoring stations, the TG focused primarily on those which measured at least one exceedance of Louisiana's Ambient Air Standard for vinyl chloride. For a full list of the monitoring stations, please see Appendix I. Average concentrations of VCM over the years listed were generated using tabulated data provided in the report. These concentrations were then used to estimate non-cancer and cancer risks in the same manner as with the Kentucky data discussed above. The table below provides the average VCM concentrations and the resulting risk estimates. In addition, concentrations of VCM associated with certain cancer risk levels are presented to assist the reader in understanding the risk levels that are associated with certain exposure concentrations of VCM; these risk estimates are based on continuous exposure to VCM at the specified concentration for a lifetime starting at birth (obtained from U.S. EPA's IRIS database).

Table 5-2: Mean Concentrations of VCM and Cancer Risk Estimates Based on Air Monitoring Data in Louisiana

	Calcasieu Parish				E. Baton Rouge Parish				Iberville Parish		Ascension Parish	
	Lighthouse		Westlake		S. Scotlandville		Southern		B. Plaquemine		Dutchtown	
	ppbv	mg/m3	ppbv	mg/m3	ppbv	Mg/m3	ppbv	mg/m3	ppbv	mg/m3	ppbv	mg/m3
Average '99-'03	0.24	6.4E-04	0.10	2.6E-04	0.37	9.4E-04	0.11	2.7E-04	0.03	8.6E-05	0.05	1.4E-04
	HI[a]	ILCR	HI	ILCR	HI	ILCR	HI	ILCR	HI	ILCR	HI	ILCR
1999-2003	6.4E-03	**5.1E-06[b]**	2.6E-03	**2.1E-06**	9.4E-03	**7.5E-06**	2.7E-03	**2.2E-06**	8.6E-04	6.9E-07	1.7E-03	**1.1E-06**

Cancer Risk Estimates for Continuous Exposure to VCM Concentrations

Risk Estimates Using Unit Risk Value of 4.4E-03 $(mg/m^3)^{-1}$			Risk Estimates Using Unit Risk Value of 8.8E-03 $(mg/m^3)^{-1}$		
1E-04	0.023 mg/m^3		1E-04	0.012 mg/m^3	
1E-05	0.0023 mg/m^3		1E-05	0.0012 mg/m^3	
1E-06	0.00023 mg/m^3		1E-06	0.00012 mg/m^3	

A, HI values were generated by calculating the ratio of the mean VCM concentration and the RfC of 0.1 mg/m^3.
b, ILCR values were calculated by multiplying the mean VCM concentration by the Unit Risk (UR) value of 8.8E-03 per mg/m^3. ILCR values represent excess cancer risk over background. The value 5.1E-6 means there is a risk of 5.1 people in 1 million potentially contracting cancer due to exposure to vinyl chloride at the concentration measured at the Lighthouse Monitor over the time period indicated. Cancer and non-cancer risk values were generated by dividing the average concentration by the RfC (for HI estimation) or multiplying the concentration by the URE (for ILCR estimation) See Appendix I for a more thorough discussion of the RfC and UR values for VCM.

The risk estimates above indicate that average concentrations from all stations except for Bayou Plaquemine resulted in cancer risks (ILCR values) that exceeded a limit of one in one million; The risk estimates ranged from a low of 1.1 in one million (Dutchtown) to a maximum of 7.5 in one million (South Scotlandville). The HI values were all well below one, indicating that non-cancer systemic effects are not expected from these exposure concentrations.

5.2.3 Discussion of fence-line risk estimates: how do they compare and what do they mean?

In general, the risk estimates as modeled above indicate that airborne concentrations in West Jefferson County, KY, and in Baton Rouge and Calcasieu County, LA are greater than the usual risk levels considered acceptable by the USEPA. These cancer risk estimates cannot be directly compared to those generated for the occupational worker because those for occupational workers have been normalized to the total amount of building material manufactured per year. This normalization was necessary in order to be able to compare one type of building material to another with regard to risks during the cradle to gate period. However, when considering potential risks to residents who live near a manufacturing facility, it is not appropriate to normalize the risk values to some outside factor. To do so would run the risk of artificially decreasing the overall risk estimate or spreading it out amongst individuals who may live hundreds or thousands of miles away and who do not breathe the same air.

One limitation to this quantitative fence-line analysis is that it is only available for the vinyl industry. Nevertheless, the analysis shows that potential risks to fence-line residents are not negligible, and should be accounted for in future life-cycle analyses. For example, the cancer risks for Kentucky and Louisiana air monitoring stations were estimated at ~1-8 excess cancer deaths in one million exposed individuals. It is useful to compare occupational risks to fence-line risks on a functional unit basis. In order to do this, one needs to start with the risk estimates for occupational workers, and adjust those using the ratios of exposure concentrations and populations in each group. Therefore, for VCM, the ratio of the lowest concentration of VCM for occupational worker:resident is 827 (based on 0.13 mg/m^3 being the lowest [non-zero] concentration estimated for a VCM worker) and the ratio of the highest exposed worker (2.6 mg/m^3) to a resident is 16,538. These ratios then give us risks for the potential exposed resident when we adjust for differences in exposure duration (hours per day and days per week). The resulting values for potentially-exposed residents then range from 10^{-10} to 10^{-8}, on a functional unit basis. This range is comparable to the occupational risk estimates of workers for most of the building materials modeled in this report; the similarity of the risk estimates underscores the need to model both occupational and bystander risk in future life-cycle assessments.

Potential effects of fence-line exposures have been documented in case reports and government reports (LDHH, 2002; ATSDR, 2002). The risk values above, in combination with the case reports, indicate past and current exposures to vinyl-associated compounds are occurring and risk estimates in some cases exceed acceptable federal and state standards.

5.3 Non-Cancer Risk from Exposure to Phthalates

In the update to this report, the TG has sought to condense the discussion on exposure to phthalates and the resultant potential risks. Rather than do a risk assessment of all the phthalate compounds that may be used to plasticize vinyl flooring (sheet vinyl and VCT), the TG has chosen to focus only on DEHP as a screening compound based on its greater toxicity and historic use. Therefore, with regard to the discussion of DEHP-induced toxicity in humans and animal

models, we have relied heavily on the most recent update of the CERHR's report on DEHP (CERHR-DEHP, 2005) (see Appendix H for a more extensive discussion of the findings of this report).

In its updated report, CERHR's expert panel reviewed literature available after the publication of its initial report in 2000. The expert panel found that there was not sufficient data available from recent human studies to indicate that DEHP causes reproductive toxicity in either sex, but that available studies were sufficient to conclude that DEHP causes reproductive toxicity in female rats, female marmosets, male rats (when exposure occurred during gestation or in the early developmental period following birth), and in adult male mice. The expert panel found that there was insufficient data in humans to determine that prenatal or childhood exposure to DEHP results in developmental toxicity. However, the expert panel found sufficient evidence to conclude that DEHP in the diet results in developmental toxicity in rats when exposure occurs either during gestation or during the early postnatal period. The lowest effect levels for testicular or developmental effects in rodents following *in utero* or postnatal exposure were in the range of 14-23 mg/kg-day (CERHR-DEHP, 2005).

The TG has estimated doses of phthalates to individuals in the U.S. by doing dose reconstruction for four different phthalate compounds. Phthalate doses were reconstructed using urinary levels of metabolites published from the most recent NHANES exposure study (Silva et al., 2004). Creatinine-adjusted levels of the following metabolites, MEP, MEHP, MBP, and MBzP, for varying age groups were used to calculate exposure estimates based on the method used by Koch and coworkers (2003). Both geometric means and 95th percentile values for urinary metabolites were used to estimate potential phthalate exposures.

Table 5-3: Combined Intake of Phthalates Based on Urinary Metabolites in Varying Age Groups (NHANES IV Data, 1999-2000)[a]

Urinary Metabolite Levels, µg/g creatinine					Intake Values, µg/kg-day[b]			
Age Group	MBzP	MBP	MEP	MEHP	DI (BBzP)[c]	DI (DBP)	DI (DEP)	DI (DEHP)
Geometric mean urinary metabolite levels								
6-11	40	41.9	92.6	5.19	0.74	0.84	1.70	3.78
12-19	17.3	24.3	142	2.53	0.32	0.49	2.60	1.84
>20	11.8	20.4	179	3.03	0.40	0.74	5.97	4.01
20-39	12.5	20	178	3.3	0.42	0.73	5.93	4.37
>40	11.2	20.7	180	2.84	0.38	0.75	6.00	3.76
95th percentile urinary metabolite levels								
6-11	142	159	625	41.9	2.62	3.19	11.46	30.52
12-19	69.3	88.1	1550	12.1	1.28	1.77	28.42	8.81
>20	57.2	91	2170	17.5	1.92	3.32	72.35	23.18
20-39	61.1	81.4	2661	20.9	2.05	2.97	88.72	27.68
>40	56.8	97.7	2064	12.9	1.90	3.56	68.82	17.09
RfD (mg/kg-day)[d]					0.2	0.1	0.8	0.02
NOAEL[d]					159	125	750	19
MOE[e]					214,865-60,687	148,810-35,112	125,000-8,453	4,347-622

a NHANES data taken from Silva et al., 2004
b Intake values modeled using parameters and equations from Koch et al., 2003, including the following: Fue (adults) = 20 mg/kg-day and DI (µg/kg-day) = UE (µg/g) * CE (mg/kg-day)/Fue *1000 mg/kg * MW diester/MW monoester.
c Daily intake
d RfD and NOAEL values obtained from U.S. EPA IRIS website (www.usepa.gov/iris)
e Margin of Exposure, a ratio of the exposure value and the NOAEL; the values listed above are calculated based on the highest value listed for each metabolite level (geometric mean or 95th percentile)
UE, urinary excretion in µg/g; CE, creatinine excretion per day, mg/kg-day); Fue, molar conversion factor; MW, molecular weight, g/mole

The data above indicate that estimated exposure doses at the 95th percentile level for DEHP are the only ones that exceed the applicable exposure limit (RfD) for each phthalate. The margin of exposure in the last row is the ratio of the estimated exposure dose and the NOAEL for that compound; the U.S. EPA often uses this value when setting exposure limits for compounds such

as pesticides and allows a MOE of >100. As shown in the table, all the MOE values are greater than 100.

Risk values were estimated using concentrations of DEHP in air and dust obtained in the Silent Spring Household Exposure Study (Rudel et al., 2003). Median and maximum concentrations of DEHP in air and dust were used to estimate daily doses of the compound, then non-cancer HI values were calculated based on children, teens, and adults inhaling DEHP in air and incidentally ingesting the dust containing the compound. The risk estimates are provided in the table below.

Table 5-4: Non-Cancer Risk Estimates for Children, Teen and Adults Exposed to DEHP in Air and Dust from Cape Cod Homes

DEHP	Concentration	Child Hazard Index	Teen Hazard Index	Adult Hazard Index
Air Exposures				
Maximum	1E-03 mg/m3	0.038	0.022	0.0099
Median	7.7E-05 mg/m3	0.003	0.0017	0.00076
Dust Exposures				
Maximum	7700 µg/g dust	3.3	0.077	0.003
Median	340 µg/g dust	0.15	0.0034	0.00013

Exposure Parameters: children, 0-7, teens 8-17, adults, 18+; ingestion rate for dust: children, 135 mg/day; teens, 10 mg/day; adults, 0.56 mg/day. Inhalation rates: child, 0.6 m3/hr; teen, 1.4 m3/hr; adult, 0.9 m3/hr. Body weights: child, 15 kg; teen, 48 kg; adult, 70 kg. Duration in home (hrs): child, 20; teen and adult, 16. Exposure assumed 7 days/week, 50 weeks per year, for a 30-year period.

These risk estimates show that the HI for children from ingestion of the maximum detected concentration of DEHP in dust is the only value that exceeds federal risk limits of 1.0 for non-cancer effects. It is important to note that use of DEHP in this analysis is very conservative, as this compound is not primarily used as a flooring plasticizer, and its RfD is 5-40 fold lower (and thus gives higher risk estimates) than the RfDs for other phthalates. Further, the risk estimate is conservative because the concentration of DEHP in dust is not quantified by contribution from different materials within the home; therefore, the value represents DEHP loading from multiple sources, including flooring.

6 Data Gaps

6.1 Data Gaps in Human Health Risk Assessment

In order to prepare this analysis, the Task Group reviewed a significant percentage of the references listed in the on-line database, including stakeholder submissions, published studies on toxicology of the building materials, worker and ambient exposure, emissions studies and LCA assessments. These sources allowed the Task Group to estimate occupational and ambient risk and combine these values to generate an overall assessment of life cycle human health impacts. These values allow for the direct comparison of the building materials assessed. These values are informed by available exposure data from stakeholders or published literature. Although the literature reviewed was extensive, data gaps still exist. The results presented are our best estimates, using the data available. More current or accurate exposure or intake data would allow refinement of the LCA and risk estimates produced. We have addressed major uncertainties with sensitivity analyses.

One of the most significant data gaps in the estimation of human health non-cancer and cancer risk estimates is the lack of accurate occupational exposure data for a large number of processes in the supply chains. This lack of information is consistent across all the building materials and includes the lack of specific knowledge of the total number of compounds to which each production worker may be exposed and how exposures are affected by the use of personal protective equipment. The Vinyl Institute did provide some general information upon request for data on occupational exposures. These data did allow for a lower bound on risks from exposure to vinyl chloride monomer, but data on ethylene dichloride exposures were not available from the industry for the purposes of this report. Further, exposures data on other compounds involved in the manufacture of PVC and ABS, including solvent cleaners, additives, heat stabilizers and others were not available.

Lack of exposure data prompts the use of “Permissible Exposure Limits” (PELs) established by the U.S. Occupational Safety and Health Administration (OSHA). Using OSHA PELs as a means to estimate exposure levels is likely to generate estimates of risk that are upper bounds for actual risks. The reason for this is that OSHA PELs are regulatory *limits* on the amount or concenration of a substance in the air. OSHA PELs are developed not only in light of reviewed toxicological data on health effects, but also in light of economic and technological feasibility for attaining these exposure limits. Therefore, some of the PELs may be higher than would be set based on health-effects data alone; in fact, in many cases, they are higher than TLVs set by ACGIH. The latter agency's exposure limits have generally not been used in this analysis however, for two reasons: ACGIH is strongly opposed to the use of its exposure limits in a ranking exercise to determine risk, and the values have no regulatory bite. The exception to this is when no PEL value was available. Lack of exposure data also prevented the estimation of health risks to workers at both the construction phase (e.g., welding fumes, exposure to pipe primers and solvent cements) and at the end of life.

Lack of knowledge of occupational exposure limits used in foreign countries is also a source of uncertainty in the analysis. In the case of cork flooring manufacture, occupational exposures were estimated using expert judgment and published emission rates for volatile compounds that were measured as off-gassing from the prepared tiles. This was done because cork flooring is not manufactured in the U.S.

European data, particularly with regard to exposure to PAHs and other compounds in foundries, were used to model worker exposures when U.S. data were limited.

Significant data gaps exist in the dose-response of respiratory effects, both non-cancer and cancer, from exposure to wood dust and Portland cement, the latter being used to manufacture fiber-cement siding. Because of the lack of appropriate toxicity values, it was impossible at this time to estimate risk values for adverse health effects for occupational workers in these industries. The only exception to this is the estimation of life-years lost due to silicosis mortality based on the exposure to silica in the manufacture of Portland cement at the OSHA PEL level.

Research in the above areas should increase the accuracy of exposure doses and dose-response assessment for the hazardous compounds that are used in the manufacture of PVC and the non-PVC building materials.

6.2 Data Gaps in Life Cycle Inventory Emission Factors

For the life cycle inventory analysis, the existing life cycle inventories were revised based on more specific information whenever possible. The areas addressed included production-related emissions of dioxins, 1,2-dichloroethane, vinyl chloride and mercury and end-of-life-related emissions of dioxins, metals and other combustion byproducts. Although we employed the most updated documentation to derive realistic emission factors for these chemicals, a large magnitude of uncertainties is often associated with the emission estimates due to lack of accurate information. Our criterion was that if any of the emission estimates is found to be a significant contributor to the overall impacts, a sensitivity analysis was warranted. As shown in the results in Section 4, the impact contribution by the production-related emissions of pollutants that can lead to cancer (including effects related to Stockholm chemicals, vinyl chloride and 1,2-dichloroethane) and the effects of adding emissions of mercury are both small relative to total estimated health impacts. Conversely, the production-related particulate matter and global climate change effects as well as end-of-life impacts, especially of possible dioxin emissions, have the potential to be very important. Therefore, a sensitivity analysis of the latter effects was warranted.

Note that, for the PM and global climate change effects, uncertainties in effect levels per unit of emissions are larger than uncertainties in emission levels, which are usually well correlated with energy consumption. On the other hand, the dioxin emission due to burning either in landfill fires, back-yard barrels or incinerators at the end of life are an especially important source of the overall emission-related uncertainties.

In order to address the magnitude of uncertainties associated with the available scientific knowledge, we conducted a bounding analysis for the dioxin emission factors per unit of material disposed. This assessment attempts to bound the influence of uncertainties in several stages of our model, including the end-of-life fates of materials (landfill versus incineration versus uncontrolled burning versus recycling); the risk of landfill fires; the mix of materials burned in landfill fires; and emissions factors for fires and uncontrolled burning. Our modeling is based on experimental emission data related to barrel burning and incinerators, statistics of municipal waste quantities and components, statistics of landfill fires, and the estimated chlorine content of each material. Although the range of emission values should theoretically capture the true emissions per unit of material disposal, it is possible that our average estimate may be significantly different from the true dioxin emission factors due to the following four reasons: (1) the emission factors were developed based on only one experimental study (Lemieux et al., 2003) for barrel burning and two experimental studies for incinerators (Katami et al., 2002;

Yasuhara et al., 2003) (2) the fate of each building material at the disposal phase (ie, %incineration vs %backyard burning vs % landfill) is not well known, (3) even if the disposal fate assumption is accurate, landfill fires and backyard burning, which account for a large portion of the dioxin emissions per unit of material disposal, may or may not involve the building materials of our interest (ie, PVC and nonPVC), and (4) even if our assumptions about (2) and (3) are accurate, dioxin emissions per unit material burned vary depending on fire temperatures, emission controls and the availability of elements that affect the dioxin formations, such as chlorine, oxygen, carbon and metal catalysts. Therefore, the uncertainty bounds and the best estimate of dioxin emission factors for the disposal phase should be refined in the future as more information becomes available.

6.3 Data Gaps for Exposure and Population Risk Analysis

As discussed above, the uncertainty in the human health effect levels of PM and greenhouse gases is large. If we assume the low-end effect levels of per unit of PM and greenhouse gases, the average overall DALY impacts tend to change in favor of cast iron pipe and aluminum siding relative to PVC. Therefore, these risk estimates should be refined in the future as more scientific information becomes available.

There is a large uncertainty in the exposure levels per unit of mercury emissions. However, given that the overall contribution of mercury effects is relatively small, the uncertainty related to mercury effects is not a significant source of overall uncertainty in our analysis.

For the exposure estimates of air toxics and particulate matter, intake fractions (as incorporated in TRACI) were used (Nishioka et al., 2005; Bennett et al., 2002). The intake fractions for air toxics were developed based on the average U.S. terrain and climates and for an average person in terms of characteristics and food intakes. This approach is useful to estimate the total impacts in the U.S. for emissions summed over a large set of sources in many locations, which is how we have used the approach in this study.

6.4 Material-specific Concerns

The non-PVC building materials are under-represented in the database, particularly with regard to toxicological endpoints and exposure assessment. For example, of the 17 articles on fiber cement siding, roughly half focus on potential adverse health effects, while the rest relate to life cycle assessment. Several of the aluminum articles discuss adverse health effects, and virtually all of these deal with neurotoxicity. Although some epidemiological studies are presented for the non-PVC materials, in general, very few of these articles provide dose-response data. The need for additional studies on these building materials was discussed further in Section 4.

7 Summary of Findings, Conclusions and Recommendations

Summary of Findings

No single material shows up as the best across all the environmental impact categories, nor as the worst. The findings from the integration of LCA and risk assessment are summarized in Table 7-1 and Table 7-2, and explained below. The life cycle performance of PVC relative to other materials depends upon two factors:

- Whether we focus on *human health impacts* or *environmental impacts*. (The Task Group refrained from weighting and aggregating the separate health and environmental results into a final overall "indicator" score.)
- *Life cycle scope.* The performance of PVC relative to the alternative materials changes as we expand the life cycle scope from cradle-through-use, by adding end-of-life with accidental fires and backyard burning, and occupational exposures by integrating LCA and risk assessment.

The influence of these two scope dimensions on the relative performance of PVC is fairly consistent across the four product groups that were studied, and may be summarized as follows.

- Relative to ***human health impacts***, aggregated in terms of total risk of mortality and morbidity (including pathways of cancer, particulate inhalation, global climate change, and impacts of mercury exposures), the performance of PVC compared with other materials depends on the life cycle scope, for the four product groups studied.
    - For a narrow life cycle "cradle-through-use" assessment, PVC performs better than some alternatives for window frames, siding, or pipe, while it performs worst among the flooring materials.
    - When we add end-of-life with accidental landfill fires and backyard burning, the additional risk of dioxin emissions puts PVC consistently among the worst materials for human health impacts, unless the end-of-life emissions from landfill fires and backyard burning are near the lower end of the wide range of uncertainty about these emissions. When end-of-life emissions are near our best-estimate value or nearer to the upper end of this range, landfill fires account for at least 80% of the total end-of-life dioxin emissions for PVC.
    - When we also add occupational exposures that we were able to model (the literature was much less complete regarding occupational exposure data for manufacture of materials other than PVC), PVC remains among the worst materials for human health, although the lack of data affects this finding.
- Relative to the ***environmental impact categories*** (acidification, eutrophication, ecotox, smog, ozone depletion, and global climate change), PVC performs better than several material alternatives, regardless of the life cycle scope, for three of the four product groups studied; the exception is flooring, for which sheet vinyl is consistently the worst material on all environmental categories except eutrophication.
- Risk estimates for ***residential exposures*** at air monitoring stations in Kentucky and Louisiana, which could not be integrated into the above findings, exceed the state and federal cancer risk limits of 1 in one million, indicating that exposure of the general population to VCM in these two locations may result in an increased chance of

developing cancer. (Fenceline risk estimates were not generated for other building materials due to a lack of exposure data.)

Conclusions

In light of the findings summarized above, we draw the following conclusions about the expected impacts of a credit that rewards avoidance of PVC, for the four material alternatives studied and across the four product groups studied, and for life cycles that include risks from dioxin emissions from accidental landfill fires and backyard burning:

- ***Human Health Risk.*** The evidence indicates that a credit rewarding avoidance of PVC could steer decision makers toward using materials that are better for human health in the case of resilient flooring. If buyers switched from PVC to aluminum window frames, to aluminum siding, or to cast iron pipe, it could be worse than using PVC. Data on end-of-life emissions are highly uncertain and therefore there is a wide range of exposure possibilities; if end-of life emissions are close to the upper end of our range, then PVC is among the worst materials studied for health risk, but if end-of-life emissions are close to the lower end of our range of possible values, then PVC is among the mid or better materials studied for health risk in the product categories of window frames, pipe, and siding.
- ***Environmental Impact.*** The evidence indicates that a credit that rewards avoidance of PVC could steer decision makers toward using materials that are worse on most environment impacts, except for the case of resilient flooring, in which sheet vinyl and VCT are worse than the alternative materials studied for most environmental impacts.

Section 6 of this report discusses data gaps, or missing information related to PVC and competing materials, and provides a detailed assessment of the subject areas that, if information became available, could alter the results of the analysis.

Recommendations

The foregoing conclusions represent the TSAC's response to the charge assigned to it by the LEED Steering Committee. Additionally, based on its work on this issue TSAC has developed several recommendations related to how materials are assessed in LEED. These recommendations are separate from the formal conclusions and should be weighed with other factors as they are considered by the Steering Committee.

- ***Need for integrated methods for materials evaluation.*** The importance of the remaining data gaps, together with the demonstrated power of integrated analysis to find key chemicals and pathways within the life cycles of product alternatives, leads TSAC to recommend that the Steering Committee use the evidence provided in this report as a basis for working towards increased use of integrated methods for material evaluation, not only to pass judgment on a particular credit for a particular material.
- ***Need for credits based on a more complete assessment of environmental and human health concerns***. In the long-term, the Steering Committee is encouraged to consider developing credits informed by both LCA and risk assessment to address critical environmental and human health issues explicitly and more systematically. Such credits should use a comprehensive, whole-building approach to critical issues. Examples could include a comprehensive whole building approach to issues such as bioaccumulative pollutants, particulate emissions and climate change.

- ***Need to address end-of-product-life phase of the life cycle.*** Our findings about the potential importance of the health impacts from accidental landfill fires and backyard burning argue for much greater attention to the end-of-product-life phase of the life cycle of PVC and other building materials. This means better data and modeling, and if the risk of major health impacts is confirmed by further empirical work, then policies to reduce this important source of health risk are recommended.
- ***Need to reward development and use of improved materials.*** Avoid the "blunt instrument" problem of material-based credits inadvertently steering decision makers to replace one high-negative-impact material with another, and instead create an ongoing market incentive for continuous development and improvement of building materials. This initiative includes two possible approaches: 1) by seeking out means for incentivizing the improvement of all buildings materials in terms of environmental and human health impacts; 2) incentivizing the substitution of problematic materials with others that are demonstrably better with regard to environmental and human health impacts over their life cycles.
- ***Need to gather and use information on occupational and life cycle impacts of products.*** The literature on occupational and fenceline risks during manufacturing is overwhelmingly focused on PVC relative to the other supply chains studied in this report, leaving large data gaps in assessments of those other product life cycles on these important topics. These data gaps need to be filled, and significant exposures then need to be included in product life cycle evaluations
- ***Opportunity to engage Innovation and Design credits in LEED.*** Develop guidelines for approval of innovation credits that move the industry forward. Recognizing that there are many possible ways to address this challenge, the capabilities and motivation of the marketplace should be engaged as a resource. Without constraining the possibilities by our current perspective, guidance should be developed for encouraging and evaluating Innovation and Design credits that can benefit the industry with additional sources of information and new approaches to materials evaluation.

Table 7-1: Ranking of Materials by Adverse Human Health Impacts – All Applications

Product group	Cancer Only			All Human Health		
	Cradle thru Use	**+End of life**	**+Occupational**	**Cradle thru Use**	**+End of life**	**+Occupational**
Window frames	1. Aluminum high Aluminum avg Aluminum low ***PVC high*** 5. Wood high Wood avg Wood low ***PVC avg*** ***PVC low***	1. ***PVC high*** 2. ***PVC avg*** 3. Aluminum high Aluminum avg ***PVC low*** 6. Aluminum low 7. Wood high Wood avg Wood low	1. ***PVC high*** 2. ***PVC avg*** 3. Aluminum high ***PVC low*** 5. Aluminum avg 6. Aluminum low 7. Wood high Wood avg Wood low	1. Aluminum high Aluminum avg Aluminum low 4. ***PVC high*** 5. ***PVC avg*** Wood high ***PVC low*** Wood avg Wood low	1. Aluminum high Aluminum avg Aluminum low 4. ***PVC high*** ***PVC avg*** ***PVC low*** Wood high Wood avg Wood low	1. ***PVC high*** Aluminum high Aluminum avg Aluminum low 5. ***PVC avg*** 6. ***PVC low*** 7. Wood high Wood avg Wood low
Pipe	1. Cast iron high, ***PVC high*** 3. Cast iron avg 4. Cast iron low 5. ***PVC pipe avg*** 6. ***PVC pipe low,*** ABS high, ABS avg, ABS low	1. ***PVC high*** 2. ***PVC avg*** 3. ***PVC low*** 4. Cast iron high 5. Cast iron avg 6. Cast iron low 7. ABS high ABS avg, ABS low	1. ***PVC high*** 2. ***PVC avg*** 3. Cast iron high 4. ***PVC low*** 5. Cast iron avg, ABS high 7. ABS avg 8. Cast iron low 9. ABS low	1. Cast iron high 2. Cast iron avg 3. Cast iron low, ***PVC high*** 5. ***PVC avg,*** ABS high, ABS avg, ABS low 9. ***PVC low***	1. Cast iron high, ***PVC high*** 3. Cast iron avg, ***PVC avg*** 5. C.iron low, ABS high, ABS avg, ABS low, ***PVC low***	1. ***PVC high,*** Cast iron high 3. Cast iron avg, ***PVC avg*** 5. Cast iron low, ABS high, ABS avg 8. ABS low, ***PVC low***
Siding	1. Aluminum high, ***PVC high,*** Aluminum avg 2. FiberCement high, Wood high, Aluminum low 7. FiberCement avg, Wood avg 9. Wood low, FiberCement low *11.**PVC avg*** ***12. PVC low***	1. ***PVC high*** 2. ***PVC avg*** 3. ***PVC low,*** Aluminum high, Wood high, Aluminum avg 7. Wood avg 8. FiberCement high 9. Aluminum low, FiberCement 11. Wood low 12. FiberCement low	1. ***PVC high*** 2. ***PVC avg*** 3. ***PVC low*** 4. Aluminum high 5. Aluminum avg 6. Wood high 7. Wood avg 8. FiberCement high 9. Aluminum low, FiberCement avg 11. Wood low 12. FiberCement low	1. Aluminum high 2. Aluminum avg, FiberCement high, FiberCement avg, Wood high, FiberCement low, Aluminum low 8. ***PVC high,*** ***PVC avg,*** ***PVC low,*** 11. Wood avg 12. Wood low	1. Aluminum high, ***PVC high*** 3. Aluminum avg, Wood high, ***PVC avg,*** F.Cement high, F.Cement avg, F. Cement low, Aluminum low, 10. ***PVC low,*** Wood avg, 12. Wood low	1. ***PVC high,*** Aluminum high 3. Aluminum avg, ***PVC avg,*** Wood high, F.Cement high, F.Cement avg, F.Cement low, Aluminum low 10. ***PVC low*** 11. Wood avg, 12.Wood low
Flooring	1. ***Sheet Vinyl high*** 2. ***VCT high*** 3. ***Sheet Vinyl avg,*** ***VCT avg*** 5. Linoleum high, ***Sheet Vinyl low***, Cork high ***8. VCT low*** 9. Linoleum avg 10. Cork avg 11. Linoleum low 12. Cork low	1. ***Sheet Vinyl high*** 2. ***Sheet Vinyl avg,*** ***VCT high*** 4. ***VCT avg*** ***5. Sheet Vinyl low,*** ***6. VCT low*** 7. Linoleum high 8. Linoleum avg 9. Cork high 10. Cork avg 11. Linoleum low 12. Cork low	1. ***Sheet Vinyl high*** 2. ***VCT high,*** ***Sheet Vinyl avg*** 4. ***VCT avg*** 5. ***Sheet Vinyl low*** 6. ***VCT low*** 7. Linoleum high, Cork high 9. Cork avg, Linoleum avg 11. Linoleum low 12. Cork low	1. ***VCT high*** 2. ***VCT avg,*** Linoleum high, ***Sheet Vinyl high*** 5. ***Sheet Vinyl avg,*** Linoleum avg 7. ***VCT low*** 8. ***Sheet Vinyl low,*** Linoleum low 10. Cork high 11. Cork avg 12.Cork low	1. ***VCT high*** 2. ***Sheet Vinyl high*** 3. ***VCT avg,*** ***Sheet Vinyl avg*** 5. Linoleum high 6. Linoleum avg 7. ***VCT low*** 8. ***Sheet Vinyl low*** 9. Linoleum low 10. Cork high 11. Cork avg 12. Cork low	1. ***VCT high*** 2. ***Sheet Vinyl high*** 3. ***VCT avg,*** ***Sheet Vinyl avg*** 5. Linoleum high 6. Linoleum avg 7. ***VCT low,*** ***Sheet Vinyl low*** 9. Linoleum low 10. Cork high 11. Cork avg 12. Cork low

Cradle through Use: health impacts through environmental pathways from life cycle inventory
+End of Life: adds health impacts from end of life disposal including backyard burning, landfill fires, incineration
+Occupational: adds occupational impacts including installation

Table 7-2: Rankings of Materials for Environmental Impacts* – All Applications

Product group	Cradle through use	Add end-of-life with burning
Window frames	1. Aluminum high (1,1,1,1,1,1,1) Aluminum avg (2,2,2,2,2,2,2) Aluminum low (3,3,3,3,3,3,3) 4. Wood high (5,4,5,4,4,7,5) Wood avg (7,5,7,5,8,8,7) ***PVC high*** (4,7,4,7,5,4,4) ***PVC avg*** (6,8,6,8,6,5,6) Wood low (9,6,9,6,9,9,9) ***PVC low*** (8,9,8,9,7,6,8)	1. Aluminum high (1,1,1,1,1,1,1) Aluminum avg (2,2,2,2,2,2,2) Aluminum low (3.3.3.3.3.3.3) 4. Wood high (6,4,4,4,4,4,7) Wood avg (7,6,5,6,5,8,8) ***PVC high*** (4,5,7,5,7,5,5) Wood low (9,8,6,8,6,9,9) ***PVC avg*** (5,7,8,7,8,6,5) ***PVC low*** (8,9,9,9,9,7,6)
Pipe	1. Cast iron high (1,1,1,1,1,1,1) Cast iron avg, 2,2,2,2,2,2,2) Cast iron low (3,3,3,3,3,3,3) 4. ***PVC high,*** (4,7,4,7,4,7,4) ABS high, (7,4,7,4,6,4,5) ABS avg, (8,5,8,5,7,5,6) ABS low, (9,6,9,6,8,6,7) ***PVC avg,*** (5,8,5,8,5,8,8) ***PVC low*** (6,9,6,9,9,9,9)	1. Cast iron high, (1,3,1,1,1,1,1) Cast iron avg, (2,4,2,2,2,2,2) Cast iron low (3,4,3,3,5,2,3) 4. ***PVC high,***(4,6,4,4,3,6,4) ***PVC avg,*** (5,8,5,5,4,7,5) ABS high, (7,1,7,6,7,3,7) ABS avg, (8,2,8,8,7,4,8) ABS low, (9,7,9,9,8,5,9) ***PVC low*** (6,9,6,6,9,8,6)
Siding	1. Aluminum high (1,1,1,1,1,3,1) 2. Aluminum avg (2,4,10,3,2,7,2) Wood high (6,2,5,2,6,5,4) FiberCement high (3,6,2,6,3,1,7) FiberCement avg (5,7,3,7,4,2,8) FiberCement low (7,8,4,8,5,4,9) 7. Wood avg (9,3,7,4,11,6,5) Aluminum low (11,9,12,9,7,12,3) ***PVC high*** (4,10,6,10,8,8,10) ***PVC avg*** (8,11,9,11,9,10,11) Wood low (12,5,8,5,12,9,6) ***PVC low*** (10,12,11,12,10,11,12)	1. Aluminum high (1,1,2,1,1,2,1) 2. Aluminum avg (3,4,10,3,2,7,2) Wood high(7,2,1,2,6,5,4) FiberCement high(4,6,3,6,3,1,8) FiberCement avg, (5,7,4,7,4,3,9) FiberCement low(6,8,6,8,5,4,10) Wood avg(9,3,5,4,11,6,5) ***PVC high***(2,10,7,10,8,8,7) Aluminum low (11,9,12,9,7,12,3) ***PVC avg***(8,11,9,11,9,10,11) Wood low (12,5,8,5,12,9,6) ***PVC low*** (10,12,11,12,10,11,12)
Flooring	1. ***Sheet Vinyl high,*** (1,4,1,1,1,2,1) ***Sheet Vinyl avg,*** (2,5,2,2,2,3,2) ***VCT high,*** (5,8,5,3,7,6,5) Linoleum high (6,1,4,6,6,1,4) 5. ***Sheet Vinyl low,***(5,8,5,3,7,6,5) Linoleum avg, (7,2,7,8,8,4,7) ***VCT avg*** (4,7,6,5,4,12,6) 8. ***VCT low*** (8,9,9,10,5,7,9) 9. Linoleum low (9,3,8,12,9,8,8) 10. Cork high (10,10,10,7,10,9,10) 11. Cork avg (11,11,11,9,11,10,11) 12.Cork low (12,12,12,11,12,11,12)	1. ***Sheet Vinyl high,*** (1,4,1,1,1,2,2) ***Sheet Vinyl avg,*** (2,5,2,2,2,3,3) ***VCT high,*** (3,6,3,4,3,5,1) Linoleum high, (6,1,4,6,5,1,7) ***Sheet Vinyl low*** (5,9,5,3,8,6,5) 6. Linoleum avg, (7,2,7,8,7,4,8) ***VCT avg*** (2,5,2,2,2,3,3) 8. ***VCT low*** (5,9,5,3,8,6,5) 9. Linoleum low (9,3,8,12,9,11,11) 10. Cork high (10,8,10,7,10,7,9) 11. Cork avg (11,11,11,9,11,8,10) 12.Cork low (12,12,12,11,12,12,12)

* Note: rankings with respect to each separate environmental impact category are presented in parenthesis, with the following impact category order: acidification, eutrophication, smog, ozone depletion, global climate change, fossil fuel depletion, ecotoxicity). The materials appear in this table in the order of their average normalized performance across the seven impact categories. The order of appearance does *not* indicate an overall environmental score, since such an overall score would require value-based weighting across the impact categories.

Assessment of the Technical Basis for a PVC-Related Materials Credit for LEED
Appendices

February 2007

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Table of Contents

Appendix A: Acronyms

AAS	Ambient Air Standard
ABS	Acrylonitrile/Butadiene/Styrene Plastic
ACGIH	American Conference of Governmental Industrial Hygienists
ADD	Average Daily Dose
ADI	Acceptable Daily Intake
ATSDR	Agency for Toxic Substances and Disease Registry
BBP	Butyl Benzyl Phthalate
BEES	Building for Environmental & Economic Sustainability
CDC	Centers for Disease Control
CDD/CDF (PCDD/PCDF)	Polychlorinated dibenzodioxin/dibenzofuran
CERHR	Center for the Evaluation of Risks to Human Reproduction
CIR	Credit Interpretation Request
CIWMB	California Integrated Waste Management Board
CMAI	Chemical Marketing Associates, Inc.
CPF	Cancer Potency Factor
CSI	Clear Sky Initiative
CSF	Cancer Slope Factor
DALY	Disability-Adjusted Life Years
DBP	Dibutyl Phthalate
DEHP	Di(2-ethylhexyl)phthalate
DINP	Diisononyl Phthalate
DWV	Drain/Waste/Vent
ECMO	Extracorporeal Membrane Oxygenation
EDC	Ethylene Dichloride
EPA	Environmental Protection Agency
EPC	Exposure Point Concentration
EPDM	Ethylene Propylene Diene Monomer
EU	End Users
HHC	Human Health Cancer
HHO	Human Health Other
HHPM	Human Health Particulate Matter
HI/HQ	Hazard Index/Hazard Quotient
HTP	Human Toxicity Potentials
IARC	International Agency for Research on Cancer
ILCR	Integrated (Excess) Lifetime Cancer Risk
IRIS	Integrated Risk Information System
LCA	Life Cycle Assessment
LCI	Life Cycle Inventory
LCIA	Life Cycle Impact Assessment
LEED	Leadership in Energy & Environmental Design
LOAEL	Lowest Observed Adverse Effect Level
LPFTC	Louisville Police Firearms Training Center
MEHP	Mono(ethylhexyl)phthalate

MOE	Margin of Exposure
MQL	Method Quantitation Limit
MSDS	Material Safety Data Sheets
NIOSH	National Institute of Occupational Safety and Health
NLM	National Library of Medicine
NOAEL	No Observed Adverse Effect Level
NTP	National Toxicology Program
OCLC	Online Computer Library Center
OSHA	Occupational Safety and Health Administration
PAH	Polycyclic Aromatic Hydrocarbon
PBTs	Persistent Bioaccumaltive Toxins
PCCD/PCDF	Polychlorinated Dibenzodioxins/Dibenzofurans
PEL	Permissible Exposure Limit
POPs	Persistent Organic Pollutants
PVC	Polyvinyl chloride
QALY	Quality Adjusted Life Year
RAIS	Risk Assessment Information System
RfD/RfC	Reference Dose/Reference Concentration
SEER	Surveillance, Epidemiology, and End Results
TEQ	Toxic Equivalents
TG	Task Group
TLV	Threshold Limit Value
TPAH	Unspecified Polycyclic Aromatic Hydrocarbons
TRACI	Tool for the Reduction and Assessment of Chemical and Other Environmental Impacts
TRI	Toxic Release Inventory
TSAC	Technical & Scientific Advisory Committee
TWA	Time-Weighted Average
UF	Uncertainty Factors
USGBC	United States Green Building Council
VCM	Vinyl chloride monomer
VCT	Vinyl Composition Tile
VOC	Volatile Organic Compounds
WHO	World Health Organization

Appendix B: Database and Sources

B.1. Database

The Task Group created a relational database that organizes the available research literature in order to assess the quantity and quality of relevant evidence. The database is envisioned as a knowledge-mapping tool. It contains a list of documents or "Sources", each of which is linked to one or more cells in a matrix that represents the universe of information that may be needed to make a decision regarding a PVC-related credit. By observing the quantity and quality of the reports linked to each cell, one can see which parts of the universe of desired information are populated with data, and which are not.

The list of Sources includes basic bibliographic information on each item, as well as an abstract (if available). Sources are classified as to whether or not they were provided as stakeholder submissions, whether or not a full copy of the document was retrieved for review, and the nature of the document (for example: risk assessment, emissions study, position paper, etc.). The database also contains a field in which Task Group members can share notes on the quality and relevance of the document.

Finally, a source is classified as "inactive" if the Task Group found that it does not contribute materially to the effort. For example, a report on levels of exposure to phthalates from toys may contain information on phthalates, but is not relevant to building materials and is therefore placed in the inactive section. Sources classified as "inactive" are also flagged with an indication as to why they received that classification. In addition, some documents were added to the database but later deleted if it was found that they were not relevant to the work. No stakeholder submissions have been deleted.

The universe of information that the Task Group reviewed in seeking to fulfill its charge is represented in the database as a three-dimensional matrix, and displayed on the screen as a two-dimensional table with nested cells. The axes of the matrix are:

1. The 14 representative materials (in the four application groups) selected for study;

2. The 10 life cycle and human health impact categories; and

3. The four life cycle stages (cradle-to-site, construction, use, and end of life).

Thus, each cell in the matrix represents a specific type of environmental or human health impact from a particular material during one life cycle stage. For example, one cell represents climate-change impacts associated with PVC drain-waste-vent pipe during the raw materials extraction and manufacturing stage. Another cell represents human health carcinogens associated with linoleum during the end-of-life stage.

Figure B-1: Life Cycle Cells page of the PVC Study Database

Each active source in the database is linked to the cells in the matrix for which it provides relevant information. In addition, the relationship between a source and a cell may be flagged to indicate whether that source provides quantitative data relevant to the cell, qualitative data, or both. The list of reports that are linked to a particular cell is viewable from the screen for that cell. Clicking on a report name opens the screen for that report, which shows all the information on that report and the list of cells to which the report is linked.

The matrix of life cycle cells can be viewed in a color-coded display that provides visual representation of the number of sources linked to each cell.

Additional reporting functions allow the data points and narratives in the database to be exported for inclusion in other documents, such as this report.

B.2. Sources

All literature considered, including all stakeholder submissions, is listed and characterized. (Those sources that contributed most significantly in terms of quantitative and qualitative information are listed in Appendix M). The characteristics of these sources as they are listed include:

- the title of the document,
- the date it was published or submitted,
- the author(s),
- the sponsor(s) of the research,
- the origin of the document (journal, website, stakeholder submission, etc.), and
- other descriptive information.

All sources were received as either stakeholder submissions or through an exhaustive literature search.

B.2.1. Literature searches

The literature search was conducted using Medline, Pollution Abstracts, Online Computer Library Center (OCLC) and the Internet, in addition to the researchers' personal collections of reports. MEDLINE is the U.S. National Library of Medicine's (NLM) premier bibliographic index to journal articles in the life sciences, including "citations" from over 4,600 of the world's leading biomedical journals from 1966 to the present. Pollution Abstracts provides access to scientific research and government policies, from the standpoints of atmosphere, emissions, mathematical models, effects on people and animals, and environmental action in response to global pollution issues. We searched in Medline and Pollution Abstracts for the most relevant and up-to-date publications in professional journals.

The table below shows the number of articles found in Medline and Pollution Abstracts for each material as of June 25, 2004.

Table B-1: Articles by material

USGBC Material	Medline	PA
PVC	359	232
Phthalate	791	321
Ethylene dichloride	160	47
Cement	309	1272
Iron	1503	312
Aluminum	1859	2129
Wood	490	451
Paint	594	678
Acrylonitrile	281	99
Styrene	155	388
Butadiene	652	163
Phenol-formaldehyde	37	23
Cork	116	31
Linoleum (floors and floor coverings)	1	4

OCLC is a catalog of books and other materials in libraries worldwide. OCLC was searched for additional LCA-related articles for each material. However, none was found to add useful information in addition to the reports and databases that we had already obtained.

Additional searches were conducted using an Internet search engine (Google) for information such as reports on toxicity of chemicals by the U.S Environmental Protection Agency (EPA), Centers for Disease Control (CDC), World Health Organization (WHO), and other institutions.

In the literature searches keywords were used to identify the materials of interest and some of their primary precursors. Searches were limited to English language articles. For Medline the search criteria were also limited to "adverse effect", "poisoning", "metabolism" and "toxicity" except for wood, iron, aluminum and paint, which were not specified for "metabolism." For the latter materials, the metabolism option was either unavailable or leading to more than 2000 articles, which would have been difficult to manage given the time constraints. Also, those materials are in general well studied, so it was presumed that the other options (e.g., adverse effects) would lead to key articles.

B.2.2. The types of articles selected for the database

Of those articles identified during the literature search steps, only articles that were relevant to the materials, applications, and environmental and human health impacts associated with the scope of this study were retained.

The types of the selected articles fall into one the following categories:

- Risk Assessment – exposure and risk-related articles.
- Toxicological assessment – epidemiological studies, animal studies and discussion papers and comments based on toxicological studies.
- Emission study – those related to emissions from manufacturing sites as well as general air pollution study (i.e., ambient or indoor concentrations) related to the materials of our interest.
- Position paper – those that take one side of the argument.
- Communication/letter – letters and reports addressing various issues, presentation materials, newspaper and newsletter articles, as well as letters from the stakeholders.
- Comparative life cycle analysis – LCA studies of one or more materials of our interest.
- Life cycle inventory (LCI) data – Reports or databases containing life cycle inventory of products.
- Others – articles that do not fall into one of the above categories (e.g., an overview of the toxicological effects of chemicals), or those cover more than one category.

Examples of articles not kept include experimental method development, PVC use in medical applications, and leakage of organotin stabilizers from the plastic matrix of certain water distribution pipes. These examples were deemed to fall outside the scope of this report as described above.

B.2.3. Characterization of data sources

The database currently contains 2,457 separate sources of information, 2,225 of which are active. Sources which have been deemed inactive are those which are not appropriate for the following reasons: the source is outdated and there is more recent material available; the source deals with an irrelevant pathway of exposure or LCA stage; or the effect described in the source is not readily quantified or extrapolated.

The sources in the database are classified by material in Figure B-2. The breakdown of the sources reflects the fact that PVC and ABS plastics are much better studied building materials. Sources that are not specific to any one or more materials are not reflected in these numbers. Sources that relate to more than one material are counted more than once.

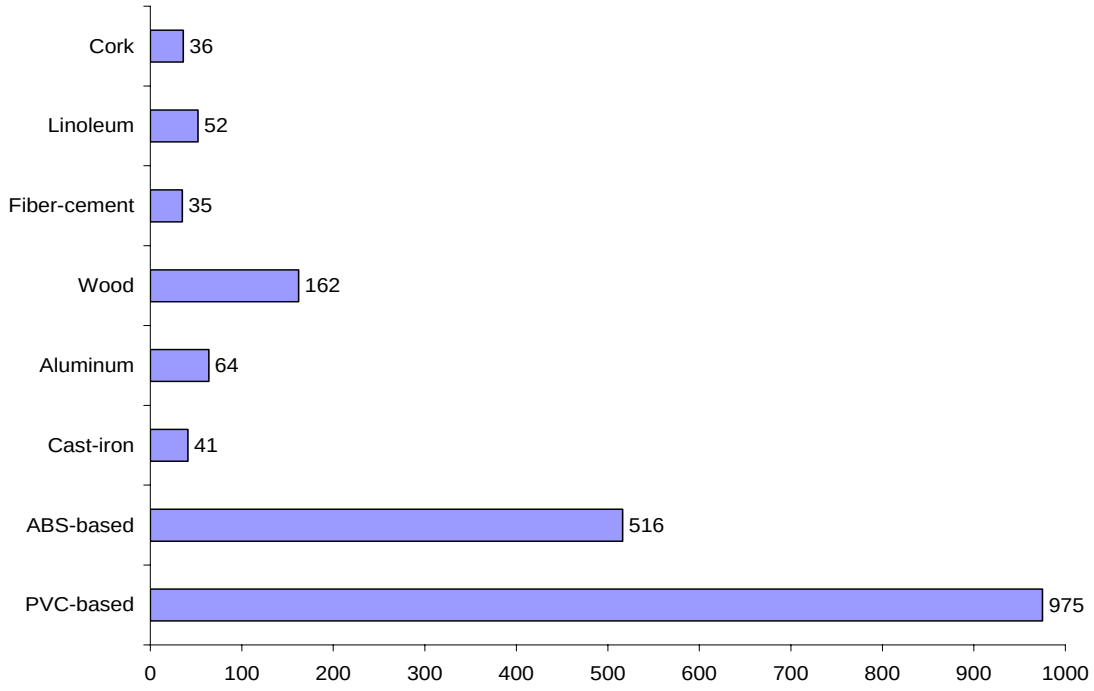

Figure B-2: Information sources by material

Sources include published scientific studies, toxicological reports on compounds used in the manufacture of PVC and plastic building materials, position papers, U.S. and foreign government and agency documents, and technical reports. The breakdown of sources by type is shown in Figure B-3.

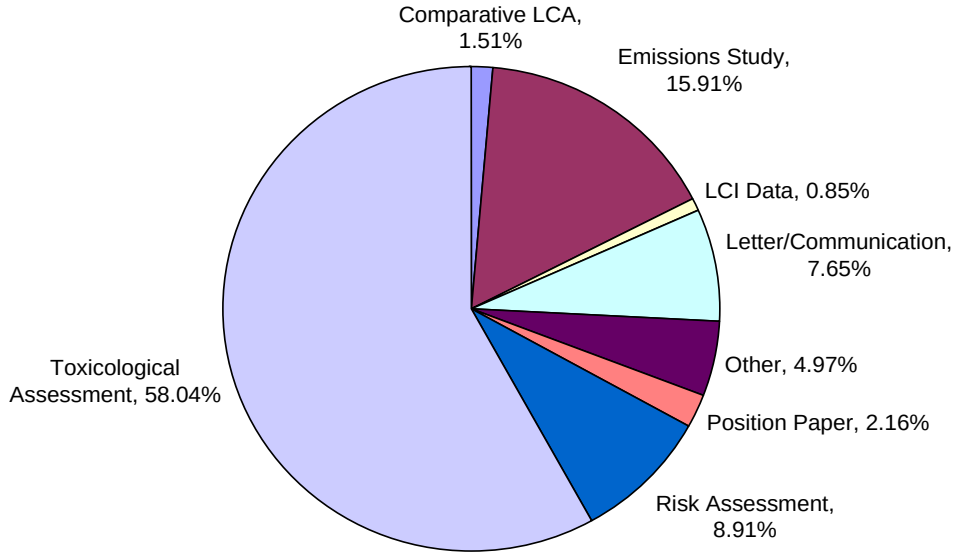

Figure B-3: Sources by type

With regard to published toxicological studies, well-controlled epidemiological studies represent the "gold standard." Those that did not contain exposure doses or approximations of doses were considered as suggestive of potential adverse effects and were considered in light of other studies in the database. Animal studies using appropriate animal models and with adequate dose-spacing (at least three non-zero doses) for the endpoints evaluated were considered over those with inappropriate models or too few doses. Case reports or anecdotal information presented in journal editorial pages were reviewed and considered for areas of further research, but these sources did not provide data that could be used in the analysis. The justification for limiting case reports, even from peer-reviewed published journals, is that these reports overwhelmingly represent exposures of individuals to extremely large concentrations of the compounds in question. For example, they often represent atypical working conditions (e.g., case report of angiosarcoma of the hand; Mohler et al., 1998) or poor industrial hygiene practices that conflict with warnings present on Manufacturing Safety Data Sheets that accompany building materials or their substrate compounds.

B.2.4. Stakeholder submissions

The database contains a rich contribution of literature generously provided by stakeholders. In total, there are 892 stakeholder submissions, encompassing the full range of source types. For those submissions addressing a building material, the sources are overwhelmingly focused on PVC (>92%; see Figure B-4). Of the PVC-related submissions, more than half (~58-59%) are related to toxicology (see Figure B-5). When stakeholder submissions are categorized by type, roughly equal numbers of toxicological studies and emissions studies were submitted (~25-38%), while comparative LCA studies and/or LCI sources constituted ≤1% of the total each. Several of the sources are scientific articles and government agency reports that were independently identified by the task group (NTP-CERHR reports on phthalates, for example). Roughly 18% were letters or communications expressing constructive criticism and suggestions for approaches to the Task Group. These submissions help to complete a picture regarding the complexity of the Task Group's charge. The sheer volume of the literature attests to the fact that there is an extensive body of data on the use of PVC.

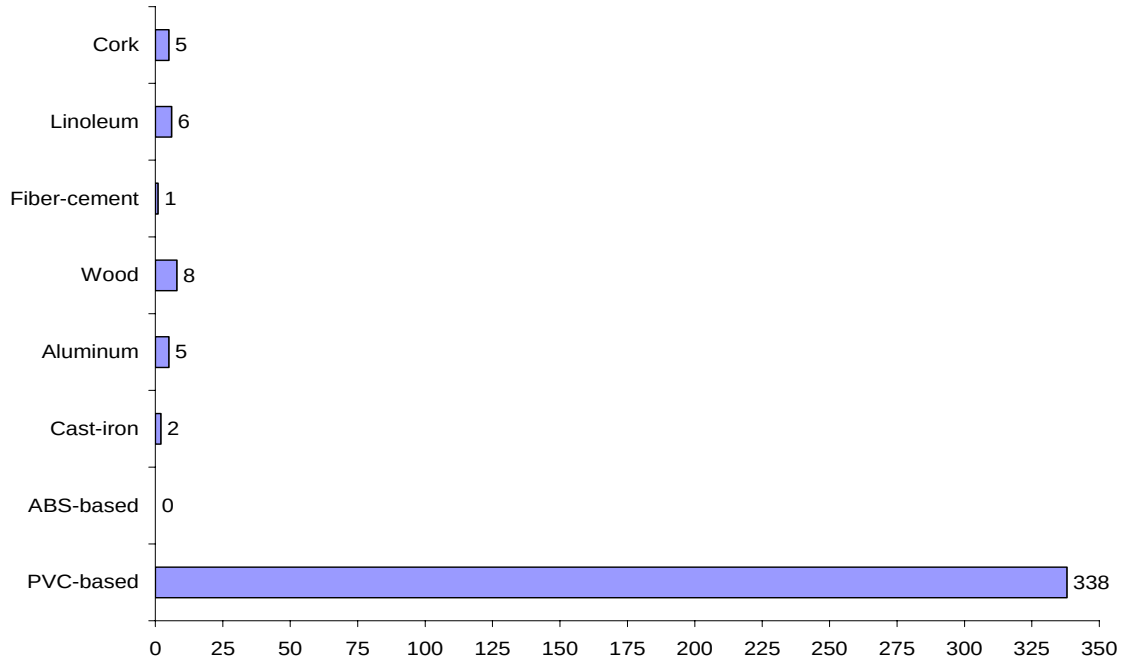

Figure B-4: Stakeholder submissions by material

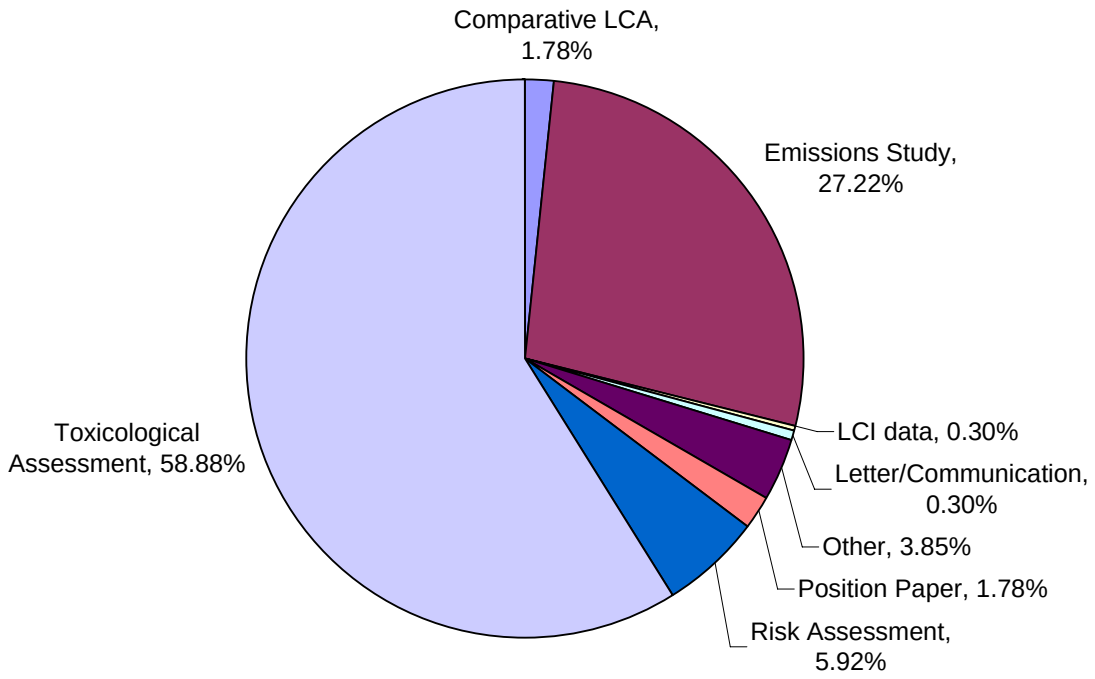

Figure B-5: PVC stakeholder submissions by type

In general, stakeholder submissions were evaluated similarly to the sources identified independently by the Task Group. Opinion pieces and letters/communications were welcomed and viewed in light of their expressed opinions. Position papers that presented new data, particularly scientific studies or independently-obtained data (such as the Vinyl Institute USGBC PVC TG Data Submission, 2004), were generally more helpful in fulfilling the charge of the task group, however. Sources that made generalized statements regarding either the "good" or "bad" qualities of a building material, without the data to justify such claims, were disregarded.

Appendix C: Life Cycle Assessment Assumptions

This section describes the four product groups selected for our study, the materials selected for each product group, and the sources of data on the energy and material inputs to final manufacturing and installation, on durability, and on emissions from final manufacturing and installation. The applications selected for study were siding, drain-waste-vent pipe, resilient flooring, and windows[1]. The specific materials studied for each of these applications are described briefly below. Note that for siding and floorings the assumptions on service life and installation waste are based on BEES (NIST, 2002) whenever possible. For piping and windows, an equal lifetime of 50 years was assumed since we found no evidence that the lifetime varies among materials. For a sensitivity analysis, an equal lifetime of 50 years for siding and 18 years for flooring was assumed. The table below shows the alternative sources of information on the lifetime. The lifetime varies among the sources, and this does not support the stakeholder comments that say PVC materials last longer than the alternative materials (as received in the online outreach forum). The delivery distance is assumed to be 500 miles for all materials. Any potentially influential assumptions are addressed in sensitivity analyses.

PRODUCT	Our assumptions (alternative assumptions)	J. Dagenais Consulting Services	The Enterprise Foundation	Accurate Inpection Service,Inc.
Vinyl Siding	40 years* (50 years)	40+ years	No data	25 - ?
Aluminum Siding	80 years* (50 years)	40+ years	40 years - life	20 – 50
Wood Siding	40 years* (50 years)	No data	life (paint regularly every 5 - 7 years)	No data
Fiber-cement Siding	45 years** (50 years)	brick veneer 50 - 150 years		35 - 50 years
ABS pipe	50 years**	drain 50 - 100, supply 50 - 75 years	drain 50 - 70, pressure 30 - 40 years	No data
PVC pipe	50 years**	drain 50 - 100, supply 20 years	30 - 40 years	drain 35 - unknown
Cast Iron	50 years**	drain 50 - 200	drain - life	drain 50 - 100 years
Sheet Vinyl Flooring	15 years*** (18 years)	No data	No data	15 - 30 years
VCT Flooring	18 years* (18 years)	15 - 20 years	10 - 15 years	No data
Generic Linoleum	18 years* (18 years)	No data	No data	No data
Cork	50 years* (18 years)	No data	No data	No data
Vinyl Windows	50 years**	35+ years	No data	15 - ?
Aluminum Window	50 years**	15 - 20 years	No data	10 - 25 years
Wood Windows	50 years**	not maint. 8 - 10 yrs, mant. 50+ years		25 - 100 (depends on maint. & exposure)

* Based on BEES data

** Based on expert judgment of the task group

[1]Initially, carpeting was also included, but it was dropped following the public stakeholder meeting in response to stakeholder comments and due to the lack of a standard, generic composition for carpet tile products.

*** Based on Potting and Blok, 1995

C.1. Siding

Four types of siding—vinyl, aluminum, wood, and fiber-cement—were analyzed for this report. They were compared on the functional unit basis of one square foot of material over a 50-year life. The delivery distance to the site (during the installation phase) is assumed to be 500 miles (805 km) for all the materials (NIST, 2002).

The useful life of products affects the number of replacements required during a 50-year period. The life span of the sidings varies by siding material -- 80 years for aluminum siding, 40 years for vinyl siding, 40 years for wood clapboards and 45 years for fiber-cement.

For aluminum, wood and fiber-cement, a painting cycle of 6 years was assumed (ATHENA, 2006). The paint was assumed to consist of water-based 71% of the time and solvent-based 29% of the time based on the market share (Paint Product Stewardship, 2001).

C.1.1. Vinyl siding

Vinyl siding was modeled with a composition of PVC resin (80%) and titanium dioxide pigment/stabilizer (20%). It is installed with galvanized steel nails, with an assumed 5% installation waste. The assumed useful life of vinyl siding is 40 years. The mass of the material is 0.333 kilogram per functional unit.

C.1.2. Aluminum siding

Aluminum siding is assumed to consist of 99% aluminum and 1% PVC coating. The recycled content of aluminum was assumed at 21.4% (U.S. EPA, 2002). Aluminum nails needed for installation were also included in the models. It is assumed to have a useful life of 80 years. The mass of material needed for the functional unit is 0.161 kg of siding and 0.0004 kg of nails. A 5% installation waste factor was assumed.

C.1.3. Wood siding

Wood siding was modeled as beveled cedar siding (clapboards)—1.183 kilogram per functional unit, with galvanized steel nails. The siding was assumed to have been installed over battens for durability, giving it a 40-year useful life. The quantity of paint used during a 50-year life was based on ATHENA data assuming the average repainting cycle of 6 years for Canada (ATHENA, 2006). Installation waste of 5% is assumed.

C.1.4. Fiber-cement siding

Certainteed and Hardie's MSDS were used to determine the components (Certainteed, 2003; Hardie Building Products, 2003). The weight fraction of Portland cement is 36.6% for Certainteed and 55% for Hardie. Galvanized steel nails were assumed for the installation, with a 5% installation waste. A 50-year useful life was assumed. The estimated mass of the material is 1.04 kilogram per functional unit.

C.2. Drain/waste/vent Pipe

The three most common materials used to make drain/waste/vent (DWV) pipe for building applications were selected for study: PVC, ABS (acrylonitrile butadiene styrene), and cast iron. These products were compared on the basis of the following functional unit: the service provided by one linear foot of schedule 40 or comparable pipe with a 3-inch inner diameter over a 50-year period.

The life span of the pipe products has been assumed to be 50 years.

Since there was no data on pipes in BEES, we use SimaPro life cycle inventory databases as the primary source of information.

C.2.1. ABS

ABS is widely used for drain/waste/vent piping in some parts of the U.S. It is assumed to contain only the ABS polymer. The estimated weight of the material is 0.95 pounds per functional unit.

C.2.2. PVC

PVC is the most common drain/waste/vent piping material used in the U.S. It is assumed to contain only PVC polymer. Its mass is estimated at 1.45 pounds per functional unit.

C.2.3. Cast iron

Prior to the widespread use of plastic piping, cast iron was the most common drain/waste/vent piping material. It is still used in many commercial applications, but rarely in residential. It is assumed to be entirely cast iron, with a mass of 5.4 pounds per functional unit. This value was calculated based on a typical outer diameter of 3.3 inches, inner diameter of 2.96 inches, and density of 0.27 pounds per cubic inch.

C.3. Resilient Flooring

Two PVC-based resilient flooring products—sheet vinyl and vinyl composition tile (VCT)—and two non-PVC products—linoleum and cork—were selected for study. These products were compared on the functional unit basis of one square foot of floor coverage over a 50-year period. The presumed life span of the flooring materials varies—18 years for linoleum and VCT, 15 years for sheet vinyl and 50 years for cork.

C.3.1. Sheet vinyl

Sheet vinyl is a widely used resilient flooring material in both commercial and residential applications. There are a range of different compositions available—several versions were selected for modeling in the life cycle analysis. Typically, these include PVC, fillers, pigments, plasticizers, and other additives. Styrene butadiene flooring adhesive was also included in the models. As noted above, the presumed life span of sheet vinyl is 15 years. Based on BEES, 5% process and installation waste was assumed.

C.3.2. VCT

Vinyl composition tile consists primarily of limestone (84%), with vinyl resins and some plasticizer. It is estimated to weigh about 1.8 kilograms per functional unit or square feet and have a useful life of 18 years. Based on BEES, styrene butadiene flooring adhesive was included in the models, as was an installation waste factor of about 1.6%.

C.3.3. Generic linoleum

Linoleum was modeled based on a composition that includes linseed oil, pine rosin, limestone, wood flour, cork flour, pigment, jute backing, and acrylic lacquer. It is assumed to have an 18-year useful life. Styrene butadiene flooring adhesive was included in the model, along with an installation waste factor of 1.1%.

C.3.4. Cork

According to BEES, the raw material transportation from Portugal is included in the life cycle inventory results. It was modeled assuming a mass of 0.51 pounds per square foot and a 50-year useful life. A water-based contact adhesive was modeled for the installation. A 5% installation waste factor was assumed. Alternatively, cork parquet in Ecoinvent data assumes 1.06kg raw cork per kg of cork slab, 0.056 kg of melamine formaldehyde resin, 0.028 kg phenolic resin, 6MJ of wood chip furnace energy, 1kWh of electivity and 0.345 tkm of transport by lorry. To reflect the transportation overseas, 1.52 tkm by ocean freight was added to the Ecoinvent data for cork slab (Ecoinvent Centre, 2005).

C.4. Windows

We compare windows that have frame materials made entirely from PVC, aluminum, or wood, based on material inputs data from ATHENA. The use phase energy consumption was modeled using DOE/LBNL's RESFEN software (LBNL, 1999). The functional unit used as a basis for comparing these windows is linear foot, over a 50-year service life. The ancillary materials considered include, ethylene-propylene-diene monomer seals, fiberglass insulation, aluminum exterior extensions / flashings for wood frame windows, vinyl extrusion extensions / flashings for vinyl and vinyl clad frames, steel fasteners, curtainwall and structural steel to connect vertical mullion in curtainwall.

All three window types were modeled using DOE/LBNL models to simulate window energy use for different window types, in different locations of the U.S. For the usage phase, an average of three cities (Sacramento, CA; Chicago, IL, and Washington, DC) which represent a range of climate was estimated based on the DOE/LBNL model. The data from the DOE/LBNL models are for annual electricity and natural gas consumptions, based on 300 square foot of window area total (2000 sq. ft. house). Then the difference from the minimum energy use was estimated across frame types in order to discount the energy loss due to the glazing. Thus, we consider only the change in energy use (as opposed to absolute energy consumptions) due to differences in frame type. The use phase energy for the 300 square foot area was interpolated to a linear-foot of farming by standardizing it to a 3'x4' window with the 14 linear feet framing.

C.4.1. Vinyl windows

The total weight per linear foot is 0.8282 kg. The framing components include PVC (85%), fiberglass (6%) galvanized steel (5%) and EPDM (5%). The use-phase energy consumptions are 500 MJ/ft and 0.4 kwh per ft.

C.4.2. Aluminum windows

The total weight per linear foot is 0.97 kg. The framing components include aluminum (86%), fiberglass (6%) galvanized steel (4%) and EPDM (4%). The use-phase energy consumptions are 1610 MJ/ft and 18 kwh per ft. A painting cycle of 7 years for the interior surface and 9 years for the exterior surface was assumed (ATHENA, 2006). We also analyzed aluminum windows with thermal breaks, using data on usage phase thermal performance from Lawrence Berkeley National Laboratory's RESFEN database, provided at http://www.efficientwindows.org. These data show that for an equivalent glazing type, thermally broken aluminum frame windows have a thermal efficiency that is intermediate between those of unbroken aluminum and vinyl frames. See Appendix K for a comparison of the results.

C.4.3. Wood windows

The total weight per linear foot is 1.9191 kg. The framing components include wood (86%), aluminum (4%), fiberglass (2%) galvanized steel (2%) and EPDM (2%). The use-phase energy consumptions are 500 MJ/ft and 0.4 kwh per ft. A painting cycle of 7 years for the interior surface and 9 years for the exterior surface was assumed.

Appendix D: Life Cycle Assessment Emission Factors

D.1. Emission Factors for Production Phases

To quantify impacts due to environmental pollution during production we develop emission factors for some key pollutants, including dioxin, mercury, VCM and EDC. Our goal is to evaluate the lower bounds and upper bounds of the emission factors based on available data. Dioxin emission factors are derived based on TRI, UNEP's Toolkit (UNEP, 2001), UNEP's POP document (UNEP, 2003), and other literature. While TRI is the best available information for the U.S., this information is usually estimated by industries and may not represent realistic values. For the upper value estimate, therefore, values in UNEP's Toolkit that represent low pollution controls are used whenever possible[2]. For mercury, TRI data as well as national annual emission and production rate are used to derive the central estimate of the emission factors. The upper values are derived based on the estimate of "missing mercury" for chlor-alkali processes.

For the other emissions such as criteria pollutants, greenhouse gases and VOCs, SimaPro provides life cycle inventories of those pollutants during manufacturing processes. The input inventories we consider include Franklin Associates, BEES, Environmental Resource Guide (American Institute of Architects, 1996) and Potting and Block (1995) for linoleum and vinyl flooring. The emissions inventory databases that we consider include Ecoinvent (Ecoinvent Centre, 2005), Franklin Associates and Sylvatica (1998), Industry Data (PRé Consultants, 2001) and BEES (NIST, 2002).

D.1.1. Dioxin emission factors during production

D.1.1.1. PVC

The lower bound of dioxin emission factors for PVC production was derived from the 2003 TRI data of 16 EDC production plants and their production rates, assuming 89% operation rate (The Innovation Group, 2005)[3]. Dioxin emission factor per ton of EDC production was converted to per ton of PVC production, assuming that 1 ton of PVC requires 2 tons of EDC. The emission factors based on the 2002 emissions are slightly higher than those based on the 2003 emissions. The mean of the 2002 and 2003 emission factors is assumed to be the 'best estimate'. On the other hand, the dioxin emission factors in Germany can be up to 33 ug I-TEQ/metric tonne VCM to air, 400 ug I-TEQ/tonne VCM to water and 400 ug I-TEQ/tonne VCM in residue (Quass et al., 2001). To test the sensitivity, we take these values as the upper bound of the emission factor estimates. The resultant upper and lower bounds of the emission factors are: 8.2 E-5 and 3.8E-2 ug TEQ/kg PVC to air, 2.2 E-4 and 4.6E-1 ug TEQ/kg PVC to water, 3.3 E-2 and 4.6E-1 ug TEQ/kg PVC in residues. Our criterion is that only if the production-related dioxin is found to be a significant contributor to the overall impacts, would a sensitivity analysis be warranted.

D.1.1.2. Aluminum

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[2] When there are control alternatives, we select emission factors for the second least control strategy in order to reflect the fact that facilities are usually regulated and that it is likely that at least a minimum control devices are installed.

[3] According to the industry information obtained from the-innovation-group.com, EDC facilities have 35.08 billion pounds capacity of production, while the domestic production was 31.208 billion pounds (domestic demand + export – import) in 2002.

For the dioxin emission factors during the production of aluminum, it was assumed that all aluminum products contain the average fraction of recycled aluminum at 21.4% (U.S. EPA, 2002). While aluminum recycling is known to produce dioxin emission, some of the dioxin emissions are due to PVC contained in aluminum products (i.e., electric conductor). Therefore, we take into account this fact by subtracting the dioxin emission attributable to PVC-coated products from the total dioxin emission factor.

In the U.S., 2132.9 million tons of scrap aluminum was produced, of which 68.7 million tons (3.3%) were electric conductor (The Aluminum Association, Inc., 2005).

Assuming 40 TEQng/kg Al (UNEP Toolkit with basic furnace with afterburner and wet scrubber, p72) for thermal wire reclamation, we estimate that 1.32 TEQ ng/kg (40*0.033) is due to PVC. We assume open burning is not the option for the thermal wire reclamation processes.

The estimated emissions rate is 5.4 ngTEQ/ kg of aluminum production, based on the assumptions that (1) 1.2 kg of aluminum scrap is required to produce 1 kg of recycled aluminum (ACR, 2005), (2) that only recycled aluminum is associated with dioxin emission, and (3) that 22.4 ngTEQ (U.S. EPA, 2005) less contribution from thermal wire reclamation is due to aluminum-only recycling.

For the lower bound, emission factor for the optimized air pollution control systems (APCS) in the UNEP Toolkit was used. The resultant emission factor is 0.256 ngTEQ/ kg of aluminum production. For the upper limit, the Toolkit value of 35 ngTEQ/kg of secondary aluminum was used with an assumption of "scrap treatment, well controlled, good APCS". The resultant emission factor is 8.97 ngTEQ/kg of aluminum production.

D.1.1.3. Cast iron

For the dioxin emissions during cast iron production, data on crude iron processing was used as a place-holder since currently data on cast iron with 100% recycled iron are not available. The dioxin emissions based on crude iron processing is likely to differ from the emissions involving scrap iron. However, because the dioxin-related cancer impacts for cast iron are a minor contributor to the total human health impacts, the effect of over- or under-estimation of dioxin emissions during cast iron production should be minimal.

Dioxin emissions during cast iron production are related to coke energy, iron pellets production and sinter production. An inventory in SimaPro indicates that 1 kg of crude iron requires 1.13 E-5 TJ of coke, 0.51 kg iron pellets and 0.99 kg sinter. For the upper and lower bounds of dioxin emission during upstream processes we use the following values available from literature: EPA's values for 100% and 0% scrap for sinter, UNEP Toolkit's values for "afterburner/dust removal" for coke, UNEP Toolkit's values for "clean scrap/virgin iron, afterburner fabric filter" and "0% scrap blast furnace" for iron pellet, and "rotary drum fabric filter" for foundries. For the lower bound values we used EPA's values for 0% scrap for sinter, UNEP Toolkit's values for "afterburner/dust removal" for coke, UNEP Toolkit's values for "0% scrap blast furnace" for iron pellet, and "hot air cupola or induction furnace, fabric filter " for foundries. The table below shows the lower and upper values of dioxin emission factors in µg per tonne of crude iron as well as the individual inputs.

	Air	Water	land
	Sinter		

Low	0.56	-	0.0030
High	4.18	-	0.0030
	Coke		
Low	0.56	1.12E-01	-
High	0.56	1.12E-01	-
	Iron pellet		
Low	0.02	0.02	2.94
High	5.88	0.02	29.41
	Foundries		
Low	0.03	-	0.2
High	4.3	-	8
	Total		
Low	1.17	0.11	3.14
High	14.93	0.11	37.41
Mid	8.05	0.11	20.28

D.1.1.4. Fiber cement

Fiber cement is a composite material formed of wood fibers bound together with Portland Cement. Cement is produced in kilns that often burn hazardous waste, in which case dioxins may be emitted. We assume that one third of cement is produced from kilns that burn hazardous waste (Beyond Pesticides, 2005).

By 1994, 37 facilities out of 111 plants in the U.S. were permitted to use hazardous waste as a fuel to replace some or all of the large amounts of fuel required. EPA reports 0.26 µg dioxin/ tonne of cement for process with non-hazardous burning and 1.11 to 30.7 µg dioxin/ tonne of cement for process with hazardous waste burning. The weighted average dioxin emission factor is 0.54 µg dioxin/ tonne of cement for the lower value and 10.4 µg dioxin/ tonne of cement for the upper value. The assumed mid value is 5.5 µg dioxin/ tonne of cement. For the emissions estimates per unit of fiber-cement siding, we consider both Certainteed and Hardie's specifications: 0.36585 kg Portland cement / kg fiber cement for Certaineed and 0.55 Portland cement / kg fiber cement for Hardie.

D.1.2. Mercury emission factors during production

D.1.2.1. PVC

There are currently nine chloralkali plants, which use outdated mercury cell technology, producing chlorine for the PVC industry (U.S. Senate, 2004). The market share of the plants using this technology in 1997 was 12% (UNEP, 2003). According to the U.S. EPA, an average chloralkali facility has 56 mercury cells, with each cell containing approximately 8,000 pounds of mercury on a given day (total, 224 tons of mercury). A report from CMAI (Chemical Market Associates, Inc., 2005), submitted by Keith Christman (Christman, 2005), states the following "Of the roughly 816,000 metric tons per year of chlorine, which could be produced via the mercury cell process, only chlorine produced via the mercury cell process at Lake Charles goes into making Vinyl products and it is very probable that not all of this chlorine goes into making Vinyl.….If all the chlorine produced at Lake Charles by the Mercury cell process goes into producing a Vinyl product, then we have the approximate maximum amount of PVC produced by mercury cell chlorine being around 4.7 percent. Should the Lake Charles plant mix their

mercury cell produced chlorine with their other produced chlorine as we suspect, then the minimum mercury cell chlorine that goes into PVC would be around 1.4 percent per year and is probably the closer number to reality."

In our analysis, we derive the lower bound of mercury emission factors based on the PPG emissions reported in TRI 2003, assuming that 1.4% of mercury cell chlorine is used for PVC manufacturing. The allocation fraction of 50% each for chlorine and sodium hydroxide is used. At PPG Industries in Westlake, LA, 30% of chlorine is used for EDC (CMAI). The production mass ratios of Cl, EDC and PVC are assumed to be 1: 3: 1.5. Considering that annual PVC production of the plant is 14.702 billion pounds, and that the annual dioxins releases are 1220 pounds per year to air, 8 pounds per year to water, and 374 pounds per year to landfills in 2003, the emission factors for air, water and landfills are 3.2E-06, 2.1E-08 and 9.8 E-07 g TEQ/kg PVC, respectively.

The plants are required to report mercury emissions; it has been reported that there are often large discrepancies between the amount plants purchase or add to the cells and the amount recorded as discharged to the Toxic Release Inventory (National Resources Defense Council, 2004). According to the U.S. EPA's Final Rule for NESHAPS on Mercury (U.S. EPA, 2003a), approximately 65 tons of mercury was unaccounted for in 2000. This amount of mercury is greater than that emitted by power plants on an annual basis in the U.S. (U.S. Senate, 2004). This admission has prompted 18 U.S. Senators to request that an inquiry be undertaken by the U.S. EPA to determine how much of the 65 tons was released by each functioning chloralkali plant and to present estimated risks to public health and the environment from mercury emissions originating from these plants (U.S. Senate, 2004). According to comments from industry personnel in response to the Final Rule (U.S. EPA, 2003a), the mercury accumulates in pipes, tanks, and other plant equipment, but no data were provided to support this claim. Other commenters to the Final Rule indicate that all mercury lost from these plants is emitted to the environment, but support for this claim was also lacking (U.S. EPA, 2003a). Based on the above EPA's assessment, the upper bound of mercury emission factor was derived by scaling up the lower bound estimate to include the potentially missing portion (i.e., the lower bound estimate x 5.64). The mid value was estimated as the average of those two extreme values.

The production of chlorine using mercury cell technology presents an environmental problem that is unique to the manufacture of PVC in that it is the only building material assessed that uses a starting material (chloralkali) that was produced using a technology that requires mercury (as opposed to the alternatives that have mercury as an unavoidable contaminant in their lifecycle). We have established our emission factors based on the best available information regarding current practices. Fortunately, the use of chlorine from chloralkali plants in PVC production is reportedly being phased out, and with it the associated mercury-related hazards.

D.1.2.2. Cast iron

We assume cast iron is made by remelting 100% scrap iron on average. We test the sensitivity of the impacts by assuming a slightly lower recycled content of 90%. We consider two sources for the mercury emission factor for cast iron: ETH and Franklin Associate's inventory data (ESU-group ETH Zurich, 1996; Franklin Associates and Sylvatica, 1998).

D.1.2.3. Fiber cement

For mercury emission factors, we use published estimates of annual mercury emissions from the American cement industry together with data on annual production to derive high, mid, and low

estimates. UNEP reports 12.9 metric tons of atmospheric emissions from cement production in North America for 1995 (UNEP, 2002). EPA's 1997 Mercury Report to Congress (1997a) reports 4.4 metric tons, while EPA's National Emission Inventory for the year 2001 is estimated at 2.2 metric tons. The production of cement in North America in 1995 and 2001 was 111,477,000 and 131,659,000 metric tones, respectively. Therefore, the high, low and mid mercury emission factor is estimated to be 116, 16.7 and 66.6 µg Hg/tonne of cement production, respectively.

D.1.3. VCM and EDC emission factors during production of PVC

To derive the emission factors of VCM and EDC related to the production of PVC, the TRI data from 2002 and 2003 were used along with the PVC production rates. For VCM, emission data from 34 VCM/EDC/PVC plants for 2002 and 30 VCM/EDC/PVC plants for 2003 are used. The production rate is 1.49 E+10 pounds in 2002 and 1.47 E+10 pounds in 2003, respectively. For EDC, emission data from 13 EDC plants for 2003 and 14 EDC plants for 2002 are used. The lower bound of the emission factor was derived from data from 2003, since they were lower than that based on 2002. The mid value of the emission factor was calculated as the average of the emission factors based on 2002 and 2003. For a sensitivity analysis, a factor of 20 relative to the average was used to test the change in results for both VCM and EDC[4].

D.2. Emission Factors for Disposal Phase

D.2.1. Overview

The Task Group obtained information indicating that accidental landfill fires and backyard burning during the end-of-life phase may be a significant source of localized dioxin concentrations. However, information on emissions for accidental landfill fires and backyard burning is very limited, and such accidental sources are usually omitted in conventional life cycle analyses. Given the lack of information in literature, but the potential importance of this life cycle phase in the total results, we developed methods to estimate these end-of-life impacts using available scientific knowledge in literature. In order to address the magnitude of uncertainties associated with the available scientific knowledge, we conducted an analysis to estimate the upper and lower boundaries for dioxin emission factors per unit of material disposal. The results are presented for a high model (upper boundary), low model (lower boundary), and average model.

In addition to dioxin emissions, we estimated emission factors for a number of pollutants such as PM, NOx, VOCs and PAHs that are emitted during combustion. We approximate emission factors for those combustion-related pollutants using information for emission from domestic combustion of wood, open burning of plastic film and uncontrolled combustion of refuse, which are available in EPA's AP-42. For PVC and ABS, we take into account additional pollutants that are not available in AP-42, such as benzene, toluene and HCl.

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[4] A factor of 20 was chosen as an arbitrary value but based on the magnitude of possible underestimation for mercury. Our criterion is, again, that if the impact of this assumption plays a significant role in the overall results, sensitivity analyses with alternative assumptions will be warranted. Note that this upper bound analysis is insignificant given that the effect contribution by VCM and EDC to the overall life cycle effects is small.

D.2.2. Dioxin emissions

D.2.2.1. Dioxin emissions related to landfill fires and backyard burning

We assume landfill fires and backyard burning have similar emission factors per kg of each material incinerated since they are both uncontrolled and combusted at low temperatures. EPA in its draft report of 2000 Dioxin Reassessment shows 1,126 gTEQ of dioxin is released per year from landfill fires and 498 gTEQ per year from backyard burning, both being potentially major sources of dioxin emissions in the U.S. According to EPA, the total dioxin releases to all media in 2000 was approximately1500 grams, or 8300 grams if other source estimates categorized as "preliminary" are also included (e.g., landfill fires). Within the quantitative inventory presented in EPA's Dioxin Reassessment, backyard burning of refuse is the top contributor (33%). However, landfill fires potentially release even more dioxins per year, more than twice as much as backyard burning. Note that the EPA's quantitative inventory does not include landfill fires because the number is still preliminary.

U.S. EPA's laboratory burn estimates for emissions of dioxins and furans using representative household waste[5] provided "baseline" emissions values which indicated that the amount of dioxins produced from recyclers' wastes were higher than non-recyclers, due presumably to the increased relative fraction of plastics in the waste left over after recycling (U.S. EPA, 1997b). In fact, according to the EPA report, one household that recycles and burns the remaining waste in a burn barrel could produce the same amount of furans as one 200-ton per day municipal waste combustion facility with appropriate pollution controls installed; four households could produce the same amount of dioxins as the combustion facility.

Additional studies performed by the U.S. EPA indicated that emissions of polychlorinated dibenzodioxins or dibenzofurans (PCDDs/PCDFs) exhibited a 1-2 order of magnitude range for individual congeners, congener groups, total values and toxic equivalents (TEQs). Irrespective of the source (e.g., inorganic or organic), increasing chlorine content of the waste increased TEQ emissions.

Additional research on the effect of changing waste composition on the emission of PCDDs/PCDFs (Gullett et al., 2000) indicates that in addition to increasing organic or inorganic chlorine content, increased copper in the waste also increases dioxin and furan emissions. Further, modifying other characteristics of the trash, such as the amount of moisture present or compressing the waste also increased dioxin and furan emissions.

Waste quantities

Approximately 230 million tons of MSW is generated per year. Of these, 131 million tons go to landfills. The table below shows the estimated amount of waste generated, recovered, burned in backyard, incinerated and landfilled. The amount burned in backyards is estimated to be 6.49 million tons in total, based on EPA's estimates on annual emission (498.53 g TEQ/yr) and the emission factor (76.8 ng TEQ/kg waste combusted). The component of waste burnt in backyards is estimated according to a representative baseline waste composition of domestic waste reported from a New York State Department of Environmental Conservation survey. We use the baseline weight fractions as reported in Lemieux et al. (2003) (see Table D-2). The quantity of plastic

[5] Based on New York State residents who do not typically recycle

waste was allocated to PVC and non-PVC waste based on EPA's characterization of municipal solid waste by weight (U.S. EPA, 2002). The EPA reports that in 2000, the amount of plastics in MSW was 23.37 million tons, of which 1.39 million tons (or ~6%) was PVC. As a result, we estimate 1.59 million tons of PVC waste is generated each year, of which 0% is recovered, 79.2% is landfilled, 20.6% is incinerated and 0.2% burned in backyards. We assume that the waste that is not recovered, landfilled or burnt in the backyard is incinerated in municipal waste incinerators.

Table D-1: Estimated waste quantities by component

Waste components	Generated (million tons)	Recovered (million tons)	Backyard (million tons)	Landfill (million tons)	Incineration (million tons)
Paper	83.10	40.00	4.01	34.14	4.95
Yard Waste	28.60	16.10	0.05	9.90	2.55
Food Waste	27.60	0.75	0.29	21.27	5.29
nonPVC plastic	**25.11**	1.39	0.49	18.79	4.44
PVC	**1.59**	-	0.0033	1.26	0.33
Steel	14.00	5.09	0.49	7.06	1.36
Alum	3.23	0.69	0.11	2.01	0.42
Other nonferrous metals	1.59	1.06	0.06	0.42	0.05
Rubber, Leather and Textiles	17.42	1.10	0.24	12.93	3.15
Glass	12.50	2.35	0.66	8.04	1.45
Wood	13.60	1.28	0.02	9.76	2.54
Other	7.94	0.98	0.08	5.51	1.36
Total	236.28	70.79	6.49	131.10	27.90

To heterogeneously form dioxins (i.e., PCDDs/PCDFs), a chlorine source, a carbon source, and a solid-phase metal catalyst must be present (Lemieux et al., 2003). Lemieux et al. (2003) performed a detailed, systematic study to investigate the variables affecting emissions of PCDD/Fs from burn barrels. The table below shows the content of a typical household waste that they simulated.

Table D-2: Waste composition

Table 1. Waste composition.

Waste Category	Waste Description	Target wt %
Paper	Newspaper, books, office paper	32.8
	Magazines and junk mail	11
	Corrugated cardboard, Kraft paper	7.6
	Paperboard, milk cartons, drink boxes	10.3
Plastic resin	Polyethylene terephthalate (PET) #1, soda bottles	0.6
	High-density polyethylene (HDPE) #2, detergent bottles, pieces	6.6
	Polyvinyl chloride (PVC) #3, schedule 40 pipe	0.2
	Polystyrene (PS) #6, food trays	0.1
	Mixed #7, Poly-Fil polyester	0.1
Food	Frozen processed potatoes	5.7
Textile/leather	Rubber and leather sneakers	3.7
Wood	Chipboard, plywood	1.1
Glass/ceramics	Bottles, jars	9.7
	Broken ceramics, flower pots	0.4
Metals (ferrous)	Iron (cans), dog food cans	7.3
Metals (nonferrous)	Aluminum cans, foil, soda cans	1.7
	Wire, Cu pipe, batteries	1.1
Total		100

(Source: Lemieux et al., 2003)

They performed 25 combustion studies of the simulated waste and measured PCDD/F by slightly varying the conditions. The controlled independent variables in the 25 runs are:

1) polyvinyl chloride (PVC) level in waste (0, 0.2, 1.0, 7.5%)
2) calcium chloride (CaCl2 an inorganic chlorine) level (0, 7.0%)
3) Cl level from added PVC (60% Cl) or CaCl2 (64% Cl) (0. 0.12, 0.60, 4.50%)
4) copper (Cu) level (Cu is thought to be a catalyst for dioxin formation) (0.07, 2%)
5) moisture content (no added water, added water)
6) density (not compressed, compressed)
7) total mass placed in barrel or in pile (single load, double load)
8) burning inside a barrel or in an open pile (steel barrel, open pile)

The uncontrolled dependent variables measured during burning are:

- dioxin/furan/PCB air emissions (by congener, totaled, and calculated TEQ)
- HCl emissions (hydrogen chloride)
- CO emissions (carbon monoxide)
- O2 emissions (oxygen)
- CO2 emissions (carbon dioxide)
- Cu emissions (copper)

- temperatures at various locations within and above the burning waste (e.g. TC6max)

The following table shows the results.

Table D-3: PCDD/F and PCB estimated emissions

Table 2. PCDD/F and PCB estimated emissions (ng/kg waste burned and ng TEQ_{WHO98}/kg waste burned).

Run	Description	TEQ_{DF}	Total PCDD + PCDF	TEQ_{PCB}	PCB Total
A	Baseline	139	11,887	0.03	136,663
B	Baseline	84	4,601	6.61	181,712
C	Baseline	25	1,756	0.01	123,877
K	0% PVC	2	306	0.01	75,411
D	Baseline	9	599	0.02	66,869
L	1% PVC	242	12,095	13.8	148,354
M	1% PVC	179	10,940	0.06	88,452
O	7.5% PVC	3,543	248,037	137.5	493,899
P	7.5% PVC	6,655	425,247	282.6	817,758
E	Baseline	148	14,418	0.04	120,698
S	0% PVC	28	2,792	NM	NM
T	$CaCl_2$	610	55,392	NM	NM
U	$CaCl_2$	934	79,549	NM	NM
Q	High Cu	2,725	252,536	NM	NM
W	Wetted	253	18,679	NM	NM
G	Compressed	358	28,213	NM	NM
X	Baseline	61	4,521	NM	NM
Z	Double	40	1,744	NM	NM
AA	Compressed	9	562	NM	NM
Y	High Cu	19	1,428	NM	NM
AC	Baseline	50	2,823	NM	NM
AD	Wetted	992	51,714	NM	NM
AH	Open	61	4,760	NM	NM
AF	Double	251	10,217	NM	NM
AE	Double	231	17,504	NM	NM

Note: NM = not measured.

(Source: Lemieux et al., 2003)

The seven baseline tests had emissions ranging from 9 to 148 ngTEQ/kg. The mean and median emissions were 73 and 61 ng TEQ/kg, respectively.

Based on the above experiments, three papers provide explanations regarding the contribution of each variable. Gullet (2001) developed one to three-predictor models of log (TEQ) based on the first 16 runs and report the following:

- Combustion measurements such as emissions of CO, Cu (as a catalyst for organohalogen synthesis) and HCl in conjunction with temperature and waste Cl content are among the most significant predictors of TEQ emissions. A product of CO and temperature at the upper most portion of the barrel is the most significant predictor in the on-predictor model, while CO alone is the best one-predictor, one-term model, explaining almost 60% of the log (TEQ) variation. A select, two-predictor model [log (Cl), log (Cu)] indicates that waste Cl content and emissions for Cu result in a slightly better model than CO alone.
- The fuel chlorine content rather than the form (PVC or $CaCl_2$) is more significant. (Note that PVC has chlorine content of about 60% and $CaCl_2$ has about 64%.)

- Variation in combustion conditions appear to be only weakly related to composition changes and is more likely related to the random orientation of waste and waste proximity factors in the barrel.

As part of various analyses, Lemieux et al. (2003) analyzed 15 runs in which only Cl levels were changed and reported the following:

- There is a statistically significant (alpha = 0.05) difference in log (TEQ) values between the 7.5% PVC runs and all other runs (1% PVC, baseline, 0% PVC) except for CaCl2. Distinctions in these runs are clearly related to the Cl content of the waste: log (TEQ) can be modeled with log (Cl) alone (R^2= 0.74). The 15 runs are well modeled for log (TEQ) (R^2= 0.9) by log (Cl), temperature at the upper most portion of the barrel, and CO.
- No distinction is observed in log (TEQ) for inorganic (7% CL in CaCl2) versus organic (7% Cl in PVC) Cl sources.

They also note the following: "In summary, although Cl in the waste does appear to influence emissions of PCDDs/Fs from burn barrels, this effect can be observed only at high levels of Cl, atypical of household trash (i.e., above 1%), and is independent of the source of the Cl (organic or inorganic). At moderate levels of Cl, a statistically significant effect of waste Cl concentration is not observed, because other more important variables have a much greater influence on the emissions of PCDDs/Fs."

Another interesting observation in Lemieux et al. (2003) is that the majority of the emissions occurred during the later stages of the burn, which represented the smoldering phase. They note that that observation may have implications for landfill fires where the smoldering stage could constitute the majority of the fire.

Neurath (2003) conducted a reanalysis of the EPA's data by Lemieux. In this study, least square regression models were used to determine the strength of relationships between variables, while ANOVA was used to compare PVC with CaCl2 as chlorine sources. To test EPA's conclusion that the effect of Cl can be observed only at high levels of Cl, Neurath (2003) ran regression analyses on the following data subsets: all data (N=25), all with "moderate Cl" (N=21, where PVC was 1% or less and without the samples with high CaCl2), all with only PVC varied (N=13) and all with "moderate" Cl and only PVC varied (N=11). Table D-4 below shows the results.

One criticism of the EPA's data was they may not have accounted for all potential PVC and chlorine sources in the waste. Although this may be true, this would overestimate/underestimate the emission factors only if they had high unaccounted sources of PVC/chlorine when their known PVC content was high/low, and they had low unaccounted sources of PVC/chlorine when their known PVC content was low/high. This could have been the case by a random chance, but based on their description, there is no reason to believe that this had happened in a systematic way. As long as the unknown sources of PVC/chlorine was present in the waste "randomly", the slope of the regression based on their known PVC content is still a valid estimate of emission factor.

Table D-4: Regression of dioxin emissions as a function of chlorine content (from Neurath 2003)

Regression of log(TEQ) with log(%Cl) for various data set choices.

runs included	N	R^2	p-value
all	25	0.50	>0.0001
all with "moderate" Cl	21	0.23	0.028
all with only PVC varied	13	0.80	>0.0001
all with "moderate" Cl and only PVC varied	11	0.54	0.01

Neurath concludes that the effect of PVC on dioxin emissions is observable at the lower PVC levels, contrary to the EPA conclusion.

In addition, Neurath makes the following observations:

- Using ANOVA, the data show that emissions as log (TEQ) are significantly different with p=0.04 between organic Cl from PVC compared to inorganic Cl from CaCl2, contradicting the EPA conclusion that there is no difference between the forms of Cl.
- Effects of waste Cl concentration at moderate levels of Cl depend on what runs are excluded from the regression. When 20 runs with moderate %Cl (i.e., 1% or less) are included, %Cl has a very strong positive correlation with dioxin wheras CO and temperature have virtually no correlation and no effect. When all 25 runs are included, the dominance of %Cl is even stronger. A smaller subset of runs where CO and temperature dominate is those 15 runs where only %Cl is varied. "The EPA seems to have selectively chosen this subset of their data to bolster their conclusion".

Based on the EPA data, Neurath developed the following regression model: log10(TEQng/kg of waste) = 2.71 + 0.85 log10(%Cl)

Dioxin generation at high temperatures

Katami et al. (2002) and Yasuhara et al. (2003) measured PCDD/Fs during combustion of PVC (some with PE, HDPE or PET) and various woods (some with elevated chlorine contents) using the same apparatus. Katami (2002) performed eight experiments with two of them using just PVC samples with varying temperatures. Yasuhara et al. (2003) performed nine experiments with four of them using plain wood (i.e., not impregnated with seawater or chlorine-containing preservatives). They record temperature, CO, HCl, CO2, O2 levels in addition to PCDD/F concentrations. Samples of exhaust gases were measured at the point before exhaust gases reach cyclone and chimney (see the figure below).

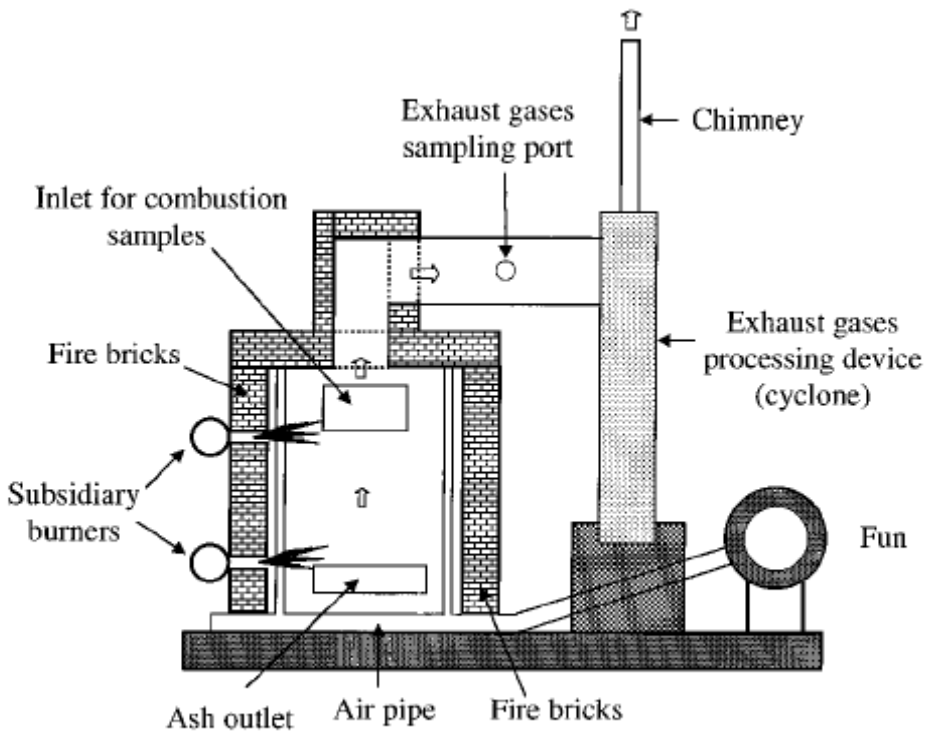

FIGURE 1. Apparatus used for combustion of PVC and related materials.

Source: Katami et al. (2002); Yasuhara et al. (2003)

Figure D-1: Apparatus used for combustion of PVC and related materials

The table below shows the estimated TEQ (g/kg) for each sample and combustion conditions.

Table D-5: Estimated dioxin emissions (g/Kg) of PVC and wood with various combustion conditions

Source	Katami (2002)		Yasuhara (2003)			
Samples	PVC high temp (pvc1)	PVC Low temp (pvc2)	I: pine (wood1)	II: cedar (wood2)	VI: pine (wood3)	VII: beach (wood4)
Avg grate temp (Celcius)	900	742	597	544	1006	969
Avg chamber temp (Celcius)	542	448	563	537	958	917
Avg exhaust gas temp (Celcius)	298	267	472	444	712	667
Avg O2 concn (%)	13.6	16.5	15.7	15.9	10.4	10.9
Avg CO2 concn (%)	5.2	3.2	4	3.9	7.4	7.1
Avg CO PPM	42	880	310	730	<5	<5
Cl ion concn (mg/N M3)	1600	1800	3.2	6.5	2.3	2.9
Cl content (%)	51.3	51.3	<0.01	0.02	<0.01	<0.01
TEQ (g/kg)	4.19E-05	2.09E-04	1.18E-06	1.00E-06	9.46E-09	2.85E-08

Katami (2002) found there is a strong correlation between dioxin formation and chlorine content. The authors indicate that PVC contributes significantly to the formation of PCDDs and PCDFs from mixtures of plastics upon combustion (Katami, 2002). The results of their study indicate that upon combustion under low temperatures and high CO concentration conditions, PVC produces high levels of both PCDDs and PCDFs

Yasuhara et al. (2003) note that their results indicate that there is an obvious relationship between chlorine content and the total of PCDDs, PCDFs, and coplanar PCBs as long as the samples are combusted below 800 °C.

Derivation of dioxin emission factors for landfill fires and backyard burning

For our disposal analysis, we need emission factors of grams of dioxin released per kilogram of waste disposed of; thus, we need factors per kg of material burned in backyard barrels, and per kg sent to landfill (noting that only a small fraction, to be estimated below, of the mass sent to landfill actually ends up burning in an accidental landfill fire). To arrive at dioxin emissions per unit of produced amount, we first use the EPA's data (Lemieux et al., 2003) and derive emission factors per unit of waste incinerated either in landfill fires or backyard barrels.

We develop regression models on dioxin (TEQ) against chlorine content (kg) using both 11 data points where PVC levels are 1% or less and 13 data points where PVC levels are 7.5% or less. The mass of chlorine was derived based on %PVC and the total barrel waste of 6.8 kg (Lemieux et al., 2003), assuming PVC contains 57% of chlorine. The tables below show the regression models.

Regression model for Log (TEQ) using **13** data points where % PVC is varied at **7.5%** or less

	Coefficients	t Stat	P-value	Lower 95%	Upper 95%	*Model F Stat*	*R Square*
Intercept	3.44	11.61	1.6E-07	2.79	4.09	0.01	0.43
Log (Cl)	0.22	2.91	0.01	0.05	0.38		

Note: for 0%PVC, chlorine content of 1E-9kg was assumed

Regression model for Log (TEQ) using **11** data points where % PVC is varied at **1%** or less

	Coefficients	t Stat	P-value	Lower 95%	Upper 95%	*Model F Stat*	*R Square*
Intercept	2.94	13.52	2.8E-07	2.45	3.43	0.02	0.45
Log (Cl)	0.14	2.69	0.02	0.02	0.26		

Note: for 0%PVC, chlorine content of 1E-9kg was assumed

The plot below shows the fitted lines based on the two models.

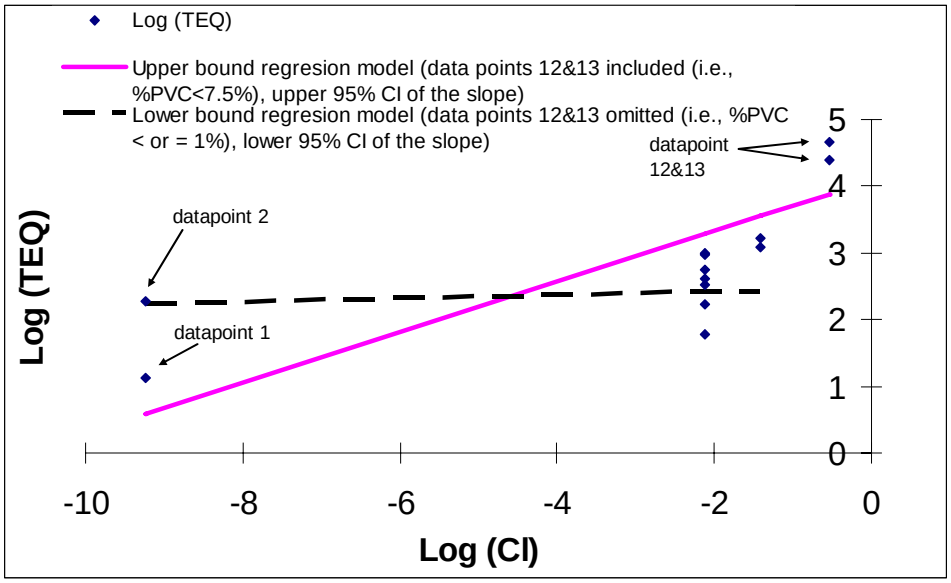

Figure D-2: Regression models for dioxin emission factors based on Lemieux et al. 2003

Based on the above regression models, dioxin emission factors were derived for non-PVC material as well. To use the regression model, we adopt the chlorine content of each material based on a European report and other sources, as shown below (Jacquinot et al., 2000; BuildingGreen, 1994; Yasuhara et al., 2003).

Household waste category	chlorine %wt
Paper	0.37
nonPVC plastics	0.01
PVC	57
Food waste	0.76
textile/leather	1.00
Wood (inc chlorine treated wood)	0.76
Wood (non-chlorine treated)	0.01
Glass/ceramics	0.01
Metals	0.03

Note: 57%wt was assumed for PVC. For non-treated wood, 0.01% of chlorine content was assumed. No information was found about chlorine content of non-PVC plastics, but 0.01% was assumed as an ad hoc value.

The table below shows the resultant dioxin emission factors (gTEQ/kg) upon combustion for landfill fires and backyard burning, based on the two regression models.

Table D-6: Dioxin emission factors for landfill fires and backyard burning

		%PVC<= 7.5 (N=13, gTEQ/kg upon combustion)			%PVC<=1 (N=11, gTEQ/kg upon combustion)		
category	kgCl per kg of component	High	low	mid	high	Low	mid
paper	0.00369	9.2E-07	2.2E-07	4.6E-07	4.1E-07	1.2E-07	2.3E-07
nonPVC plastic	0.0001	2.5E-08	6.1E-09	1.2E-08	1.1E-08	3.3E-09	6.1E-09
PVC	0.57	1.4E-04	3.5E-05	7.0E-05	6.4E-05	1.9E-05	3.5E-05
food waste + other	0.00756	1.9E-06	4.6E-07	9.3E-07	8.5E-07	2.5E-07	4.6E-07
Textile/lether	0.00999	2.5E-06	6.1E-07	1.2E-06	1.1E-06	3.3E-07	6.1E-07
wood (inc treated)	0.00756	1.9E-06	4.6E-07	9.3E-07	8.5E-07	2.5E-07	4.6E-07
wood (untreated)	0.0001	2.5E-08	6.1E-09	1.2E-08	1.1E-08	3.3E-09	6.1E-09
glass/ceramics	0.0001	2.5E-08	6.1E-09	1.2E-08	1.1E-08	3.3E-09	6.1E-09
metals	0.00032	8.0E-08	2.0E-08	4.0E-08	3.6E-08	1.1E-08	2.0E-08
fiber cement	0.0001	2.5E-08	6.1E-09	1.2E-08	1.1E-08	3.3E-09	6.1E-09

Note: 0.01%wt of chlorine content assumed whenever there is no known content of chlorine

We assume 0.01% of chlorine content whenever there is no known content of chlorine. For our lower-bound analysis, linoleum and cork were assumed to have the same emission factor as wood (i.e., same rate of recovery, incineration, landfill and backyard burning). For ABS we use combustion emission factors for non-plastics with an assumption that the rates for recycling, landfill, backyard burning and incineration are equivalent to non-plastics. Fiber cement is assumed to be 97.3% landfilled, 2.7% barrel burned: An MSDS (Certainteed, 2003) show landfills as the only option as the disposal method and 2.7% is the average rate of backyard burning (6.49 million ton backyard burning / 236.28 million tons of waste generated per year). For paints and phthalates, no recycling was assumed but the rate of landfill as opposed to incineration was assumed the same as those materials with which they are disposed (i.e., PVC for phthalates; and wood, aluminum or fiber cement for paints). For aluminum the rate of recycling was determined based on the overall recycling rate of 21.4% (U.S. EPA, 2002).

Quantity of burned waste in landfill

Landfill fires fall into one of two categories, surface and underground fires (Federal Emergency Management Agency, 2000). For our analysis, we vary the fraction of the fire occurring on the surface (the top layer that accumulates within a year for a newly landfilled waste) as opposed to underground (layers below the top layer or after a year and beyond of landfilled waste) as well as the duration of waste that stays in landfills.

Proportion of annual landfill input burned in fire over n years is estimated using the following formula:

$$\underline{I-(I-L*P)[1-L*(1-P)/(A*I)]\ n-1}$$

where,

I= annual landfill input (131.1 million tons/year) (U.S. EPA, 2002)
L= average amount of waste in landfills that catches fire per year (1.126 million tons burned each year). This amount is estimated based on EPA's estimate of dioxin emissions from landfill

fires (1126 g per year) divided by UNEP's emission factor 1000ng/kg for landfill fires (U.S. EPA, 2005)
P= proportion of L in first year's landfill input (25 –95%)
I-L*P= size of initial landfill input after 1 year in landfill (i.e., initial input less amount burned)
A = average age of existing active landfill in years (13 years) (U.S. EPA, 2004c)
A*I = total amount of existing landfill waste from previous years
1-L*(1-P)/A*I = annual rate of underground fire
$I-(I-L*P)[1-L*(1-P)/(A*I)]^{n-1}$= remaining size of new landfill waste after n years
n = number of years the initial input stays in landfill (30-100 years)

As the table below shows, the proportion catching fire over years of an initial input varies between 0.01 to 0.05, depending on the assumed number of years in landfills as well as the proportion of landfill fires happening on the surface as opposed to underground.

p = proportion of landfill fires happening on the surface(new landfill waste)	years in landfill	Proportion of initial landfill input burned over years in landfill
0.95	1	0.00817
0.95	30	0.00912
0.75	30	0.0112
0.5	30	0.0138

EPA reports that 5% of landfill fires occur underground. For our bounding analysis, we assume 0.8% as the lower bound of the proportion of landfill inputs burned over years and 2% as the upper bound.

The table below shows the resultant range of dioxin emission factors per kg of waste generation. The upper bound is based on the upper 95% confidence limit of the regression model with %PVC<=7.5% with an assumption that all waste is either recovered, landfilled, incinerated or burned in backyards according to the waste statistics, and the lower bound is based on the lower 95% confidence limit of the model with %PVC <=1.5% with an assumption that all waste is landfilled after recovery. The mid value is the mean of the two bounding values.

Table D-7: Dioxin emission factors for landfill fires and backyard burning (g TEQ/kg generated as waste)

	landfill (g TEQ/kg generated as waste)		
	low	**high**	**mid**
Paper	4.05E-10	7.59E-09	4.00E-09
nonPVC plastic	2.00E-11	3.74E-10	1.97E-10
PVC	1.21E-07	2.26E-06	1.19E-06
food waste + other	1.52E-09	2.85E-08	1.50E-08
textile/leather	1.98E-09	3.71E-08	1.95E-08
Wood (untreated)	7.45E-12	2.33E-10	1.20E-10
Glass/ceramics	1.72E-11	3.22E-10	1.70E-10
Metals	4.31E-11	8.07E-10	4.25E-10
Fiber cement	2.60E-11	5.00E-10	2.63E-10
	Backyard burning(g TEQ/kg generated as waste)		
	low	**high**	**mid**
Paper	5.93E-09	4.45E-08	2.52E-08
nonPVC plastic	6.51E-11	4.88E-10	2.77E-10
PVC	3.97E-08	2.98E-07	1.69E-07
food waste + other	2.63E-09	1.97E-08	1.12E-08
textile/leather	4.59E-09	3.45E-08	1.95E-08
Wood (untreated)	4.27E-10	3.20E-09	1.81E-09
Glass/ceramics	5.64E-12	4.23E-11	2.40E-11
Metals	1.75E-10	1.31E-09	7.44E-10
Fiber cement	3.72E-10	2.79E-09	1.58E-09

Note: The lower bound of wood is based on 2ngTEQ /kg for residential wood combustion instead of based on regression models, since many types of wood have less than 0.01% of chlorine content.

D.2.2.2. Dioxin emission factors for controlled combustion/incinerators

For the controlled incineration of the other materials, such as ABS, paint, as well as components of products that may be incinerated (e.g., phthalates) we use the range of emission factors for non-plastics assuming that the chlorine content of 0.01% would be appropriate. Based on the estimated emission factors per kg of waste combustion, emission factors per functional units were estimated.

Wood combustion

For incineration in municipal waste incinerator (MWIs) (i.e, a high temperature combustion) we use EPA's emission factors: 0.6-132 ng TEQ/kg during industrial wood combustion for power/energy generation (U.S. EPA, 2005). The emission factor of 0.6 ng I-TEQ/kg is for non-salt-laden wood, while that of 13.2 ng I-TEQ/kg is for salt-laden wood. Because the wood used for sidings and window frames are not usually salt-laden, we assume that the range of 0.6 – 6.9 ng TEQ/kg would represent non-salt-laden wood for our purposes. Therefore, the central value of wood-related emission factors during incineration in MWI is 3.75 ng TEQ/kg. The resultant mid, lower and upper TEQ values per kg of wood waste generated are 7.06E-10, 1.13E-10 and 1.3E-9, respectively.

PVC combustion

We derive dioxin emission factors during the incineration of PVC in MWI based on the above wood dioxin emission factors (0.6 to 6.9E-9 gTEQ/kg) and scaling factors that we develop based on the grate temperatures, COs and TEQs for PVC and wood combustions as reported by Katami (2002) and Yasuhara et al. (2003). To determine the scaling factor, we use the following objective function and find the weighting factors for pvc1 (b) and the weighing factors for wood (a1-a4) so that we minimize Q

where:

$$Q= [(a1*Twood1+a2*Twood2+a3*Twood3+a4*Twood4)-(b*Tpvc1+(1-b)*Tpvc2)]^2-[(a1*COwood1+a2*COwood2+a3*COwood3+a4*COwood4)-(b*COpvc1+(1-b)*COpvc2)]^2,$$

and T=temperature (Celsius) and CO = carbon monoxide concentration (PPM). Our important constraint is that the ratio of Total COwood / Total COpvc should be greater than 0.9. This constrain reflects the observation that CO may be a better predictor than temperature (Lemieux et al., 2003). For a high temperature process, we use b=0.75. The corresponding best solutions for a1-a4 are: 0, 0.34, 0.66, 0, respectively. The scaling factor, calculated as a ratio of weighted TEQpvc and TEQwood is 241.3. Using this scaling factor, the lower bound for PVC incineration in MWI is estimated to be 2.85e-8 g/kg (6E-10*241*1.55/7.88) of waste. The estimated upper and mid values are 2.47E-7 g/kg and 1.46E-7 g/kg, respectively.

Combustion of other materials

For the controlled incineration of the other materials, such as ABS, paint, as well as components of products that may be incinerated (e.g., phthalates, limestone) we use the range of emission factors for wood (0.6E-9 to 6.9E-9 gTEQ /kg) assuming that the chlorine content is negligible. Based on the estimated emission factors per kg of waste, emission factors per functional units were estimated.

D.2.2.3. Summary of dioxin emission factors

The table below shows the summary of resultant dioxin emission factors via landfill, incineration and open barrel burning (gTEQ/kg waste generation).

Table D-8: Dioxin emission factors for landfill fires and backyard burning.

	PVC			
	low	high	mid	Assumptions for emission factors and waste quantities
Landfill	1.21E-07 (100%)	2.26E-06 (80%)	1.19E-06 (79%)	High: Upper 95% confidence limit of the regression-based dioxin emission factor assuming % chlorine in PVC is 57% while 79.2% of all PVC waste goes to landfill (MSW statistics) , Low: Lower 95% confidence limit of the regression-based dioxin emission factor assuming % chlorine in PVC is 57% while 100% of all waste after recovery goes to landfill
Incineration	0.00E+00 (0%)	2.58E-07 (9%)	1.53E-07 (10%)	High: EPA's emission factors 6.9 ng TEQ/kg during industrial wood combustion, adjusted by a scaling factor for PVC considering the grate temperatures, COs and TEQs for PVC and wood combustions and 20.6% of all PVC waste goes to

				incinerators (MSW statistics). Low: same as "high" except that industrial wood combustion is assumed to emit 0.6 ng TEQ/kg and 0% of all PVC waste goes to incinerators
Open burning	0.00E+00 (0%)	1.97E-08 (11%)	1.12E-08 (11%)	High: Upper 95% confidence limit of the regression-based dioxin emission factor assuming % chlorine in PVC is 57% and 0.2% of all PVC waste is burned in backyard (MSW statistics). Low: Lower 95% confidence limit of the regression-based dioxin emission factor assuming % chlorine in PVC is 57% and 0% of all PVC waste after recovery is burned in backyards
Total	1.21E-07	2.54E-06	1.35E-06	
		Wood/linoleum/cork		
Landfill	7.45E-12 (100%)	2.33E-10 (15%)	1.20E-10 (14%)	High: Upper 95% confidence limit of the regression-based dioxin emission factor assuming % chlorine in wood is 0.01% and 72% of all wood waste goes to landfill (MSW statistics), Low: Lower 95% confidence limit of the regression-based dioxin emission factor assuming % chlorine in wood is 0.01% and 100% of all waste after recovery goes to landfill
Incineration	0.00E+00 (0%)	1.29E-09 (82%)	7.00E-10 (83%)	High: EPA's emission factors 6.9 ng TEQ/kg during industrial wood combustion and 18.7% of all wood waste goes to incinerators (MSW statistics). Low: same as "high" except that industrial wood combustion is assumed to emit 0.6 ng TEQ/kg and 0% of all wood waste goes to incinerators
Open burning	0.00E+00 (0%)	4.23E-11 (3%)	2.40E-11 (3%)	High: Upper 95% confidence limit of the regression-based dioxin emission factor assuming % chlorine in wood is 0.01% and 0.17% of all wood waste is burned in backyard (MSW statistics). Low: Lower 95% confidence limit of the regression-based dioxin emission factor assuming % chlorine in wood is 0.01% and 0% of all wood waste after recovery is burned in backyards
Total	7.45E-12	1.56E-09	8.44E-10	
		Non-PVC plastics (e.g., ABS)		
Landfill	2.00E-11 (100%)	3.74E-10 (34%)	1.97E-10 (33%)	High: Upper 95% confidence limit of the regression-based dioxin emission factor assuming % chlorine in NON-PVC PLASTICS is 0.01% while 75% of all NON-PVC PLASTICS waste goes to landfill (MSW statistics), Low: Lower 95% confidence limit of the regression-based dioxin emission factor assuming % chlorine in NON-PVC PLASTICS is 0.01% and 100% of all waste after recovery goes to landfill.
Incineration	0.00E+00 (0%)	2.28E-10 (21%)	1.24E-10 (21%)	High: EPA's emission factors 6.9 ng TEQ/kg during industrial NON-PVC PLASTICS combustion and 17.7% of all NON-PVC PLASTICS waste goes to incinerators (MSW

				statistics). Low: same as “high” except that industrial NON-PVC PLASTICS combustion is assumed to emit 0.6 ng TEQ/kg and 0% of all NON-PVC PLASTICS waste goes to incinerators High: Upper 95% confidence limit of the regression-based dioxin emission factor assuming the % chlorine in NON-PVC PLASTICS is 0.01% and 1.95% of all NON-PVC PLASTICS waste is burned in backyard (MSW statistics). Low: Lower 95% confidence limit of the regression-based dioxin emission factor assuming the % chlorine in NON-PVC PLASTICS is 0.01% and 0% of all NON-PVC PLASTICS waste after recovery is burned in backyards
Open burning	0.00E+00 (0%)	4.88E-10 (45%)	2.77E-10 (46%)	
Total	2.00E-11	1.09E-09	5.98E-10	
			VCT (BEES spec)	
PVC 12%	1.45E-08	3.04E-07	1.62E-07	Same as PVC
Limestone 84%	0.00E+00	1.57E-09	8.58E-10	for open burning emission factor for 0.01% chlorine content is assumed; for incineration, emission factor of wood is assumed
Phthalates 4%	0.00E+00	7.47E-11	4.09E-11	for open burning emission factor for 0.01% chlorine content is assumed; for incineration, emission factor of wood is assumed
	1.45E-08	3.06E-07	1.63E-07	
			VCT AIA per kg VCT	
PVC 13%	1.57E-08	3.30E-07	1.76E-07	Same as PVC
Limestone 76%	3.90E-11	1.42E-09	7.76E-10	for open burning emission factor for 0.01% chlorine content is assumed; for incineration, emission factor of wood is assumed
Phthalates 5%	2.57E-12	9.34E-11	5.11E-11	for open burning emission factor for 0.01% chlorine content is assumed; for incineration, emission factor of wood is assumed
Total	1.57E-08	3.31E-07	1.77E-07	
			Sheet vinyl (AIA spec)	
PVC 47%	5.66E-08	1.19E-06	6.36E-07	Same as PVC
Limestone 29%	1.49E-11	5.42E-10	2.96E-10	for open burning emission factor for 0.01% chlorine content is assumed; for incineration, emission factor of wood is assumed
Phthalates 16%	3.20E-12	1.74E-10	9.56E-11	for open burning emission factor for 0.01% chlorine content is assumed; for incineration, emission factor of wood is assumed
Total	5.67E-08	1.19E-06	6.37E-07	
			Sheet vinyl (Potting spec)	
PVC 50%	6.03E-08	1.27E-06	6.77E-07	Same as PVC
Limestone 15%	7.70E-12	2.80E-10	1.53E-10	for open burning emission factor for 0.01% chlorine content is assumed; for incineration, emission factor of wood is assumed
Phthalates 30%	1.54E-11	5.60E-10	3.06E-10	for open burning emission factor for 0.01 % chlorine content is assumed; for incineration, emission factor of wood is assumed

Total	6.03E-08	1.27E-06	6.77E-07	
			Paint	
on wood landfill/open burning	5.00E-11	4.43E-10	2.47E-10	Emission factor for open burning with chlorine content of 0.01% is assumed
on wood incineration	0.00E+00	1.42E-09	7.72E-10	Emission factor for incineration of wood is assumed
Total on wood	5.00E-11	1.86E-09	1.02E-09	
on aluminum landfill/open burning	3.09E-10	1.50E-09	9.07E-10	Emission factor for open burning with chlorine content of 0.01% is assumed
on aluminum incineration	0.00E+00	1.13E-09	6.13E-10	Emission factor for incineration of wood is assumed
Total on aluminum	3.09E-10	2.63E-09	1.52E-09	
on fiber cement landfill/open burning	4.88E-11	5.00E-10	2.75E-10	Emission factor for open burning with chlorine content of 0.01% is assumed
on fiber cement incineration	-	-	-	Emission factor for incineration of wood is assumed
Total	4.88E-11	5.00E-10	2.75E-10	
			Phthalates in PVC	
landfill/open burning	5.14E-11	4.49E-10	2.50E-10	emission factor for open burning with chlorine content of 0.01% is assumed
incineration	0.00E+00	1.42E-09	7.71E-10	emission factor for incineration of wood is assumed
Total	5.14E-11	1.87E-09	1.02E-09	

D.2.3. Emissions related to landfill fires and backyard burning: other than dioxin

When materials are combusted, a number of pollutants such as PM, NOx, VOCs and PAHs are emitted. We approximate emission factors for those combustion-related pollutants using information for emission from domestic combustion of wood, open burning of plastic film and uncontrolled combustion of refuse, which are available in EPA's AP-42. For PVC and ABS, we take into account additional pollutants that are not available in AP-42, such as benzene, toluene and HCl based on Takasuga et al. (2003).

Appendix E: Life Cycle Impact Assessment Characterization of Human Health Impacts

Characterization factors are used to estimate the population impacts associated with life cycle emissions. The method to arrive at a range of characterization factors is described below.

E.1. Particulate Matter

E.1.1. Exposure

As in a health risk analysis, the PM-related human health impact in LCIA is estimated as a function of exposure and potency. To estimate the population exposure per unit of emissions of PM and the precursors we rely on the concept of intake fractions (iF)(a dimensionless ratio between the amount of pollutant intake and the amount of a pollutant emitted) as a summary measure related to the emission-concentration relationship. The intake fraction concept is most useful in the context of life cycle assessment where the number of sources to model is usually large and the input information needed to construct detailed dispersion models is often unavailable or prohibitively expensive. With the intake fraction approach, population exposure (i.e., midpoint) is estimated simply as a product of emission and the intake fraction for the relevant region.

Population exposure is estimated as a function of air concentrations in various locations as a result of atmospheric dispersion and the breathing rate of the population residing in the affected regions (Harrison et al.,1986; Smith,1988; Phonboon,1996).

The intake fraction is mathematically expressed as:

$$\mathrm{iF} = \frac{I}{Q} = \frac{BR \times \sum_i \Delta Ci \times Ni}{Q} \qquad (1)$$

where I = intake rate [μg/day]; Q= emission rate of pollutant or pollutant precursor from selected source [μg/day]; iF = intake fraction; BR = population-average breathing rate (assumed to be 20 m^3 /day for adults); $\in Ci$ = change in concentration of pollutant at receptor i given emissions from selected source (μg/m3); and Ni = number of people at receptor i.

In this study, we apply the U.S. intake fractions developed by Andy Wilson (Wilson, 2003) and Sue Greco (Greco et al., 2005) at the Harvard School of Public Health, who have developed power plant and mobile source intake fractions based on a source-receptor (S-R) matrix[6] (Wilson, 2003; Greco et al., 2005). The average intake fractions of primary PM2.5 vary from 1.2E-6 (power plant) to 1.7E-6 (mobile) for the U.S. For secondary PM2.5, intake fractions for

[6] The S-R matrix, which was originally developed by Abt Associates for regulatory impact analyses, contains county-to-county transfer factors across the United States for primary particles and secondary particles including sulfates and nitrates (Abt Associates et al., 2000). In the S-R matrix, regional variability in climate (e.g., temperature, relative humidity, precipitation) as well as the background concentrations of ammonia, sulfate, and nitrate for secondary particle formation (i.e., rate-limiting conditions of nitrate formation) have been incorporated (Levy et al. 2003).

SO2/sulfate-nitrate do not vary much between regions, roughly 3E-7 for power plants and 4E-7for mobile sources. The average intake fractions of NOx/nitrate vary from 5.1E-8 (power plant) to 8.5E-8 (mobile) for the U.S. Assuming the power plant and mobile source intake fractions represent upper and lower bounds, respectively, the average intake fraction for the U.S. are 1.5E-06 for PM2.5, 3.6E-07 for sulfates, 6.9E-08 for nitrates.

E.1.2. Potency

For the health endpoints, we principally focus on premature mortality, since it contributes a large portion of total costs based on past studies (Abt Associates et al., 2000; U.S. EPA, 1999). Various morbidity categories have also been associated with particulate matter in a number of epidemiologic studies. For this analysis, we include only those for which reasonable scientific and epidemiological evidence are available, and which make a non-negligible contribution to the total impacts; namely, chronic bronchitis (CB), cardiovascular hospital admissions (CVHA) and restricted activity days (RAD). We also aggregate the mortality and morbidity impacts in Disability Adjusted Life Years (DALYs).

For the potency of PM in premature mortality, we follow the approach taken in Nishioka et al. (2002) and draw the central estimate from the updated Pope study (Pope et al., 2002), in which a 10 μg/m3 increase in annual mean PM2.5 concentrations was associated with a 1.06 relative risk for premature mortality (95% confidence interval: 1.02, 1.11). For small changes in concentrations, these estimates translate into approximately 0.5% increase in premature deaths for each μg/m3 increase in annual mean PM2.5 concentrations, which can be applied to a baseline mortality rate of 0.014 deaths/person/year for individuals 30 years of age or older (Murphy, 2000). To test the sensitivity of the PM potency for the total human health DALY effects, we apply the lower confidence limit of relative risk (1.04).

For the potency of chronic bronchitis, we apply the findings by Abbey (1995), who found based on a sample of California residents, in a cohort of non-smoking adults, a 45 μg /m3 increase in PM2.5 was associated with a relative risk of 1.81 for chronic bronchitis (95%CI: 0.98, 3.25).

Samet et al. (2000) pooled cross-city results for 14 U.S. Cities and Schwartz (1999) results for 8 U.S. cities and provide the most precise estimates for the relationships between U.S. ambient PM10 exposures to increased risk for cardiovascular disease hospitalization. For cardiovascular disease hospitalization, Studies have confirmed likely excess risk of CV-related hospital admissions for U.S. cities in the range of 3-10% per 50 μg /m3 PM10, especially among the elderly (65 yr). Overall, we determine the central estimate of the CVHA C-R slope to be equivalent to 0.12 % increase in CVHA per μg/m3 of daily average PM2.5 (95% CI: 0.05%, 0.2%).

For restricted activity days, the relationship between RAD and PM2.5 exposure was derived from a study of adults age 18 to 65 included in the Health Interview Survey between 1976 and 1981 (Ostro, 1987). The EPA determined a pooled estimate from the six year-specific regressions using a weighted average with weights that reflect the inverse of the variance in the reported coefficients (Abt Associates, 1999). This pooling methodology yielded an estimated 0.47% increase in RAD per μg/m3 of daily average PM2.5 (95% CI: 0.42%, 0.53%), applied to a background daily incidence rate of 0.0177 for all adults age 18 to 65.

E.1.3. Derivation of disability-adjusted life years per case

The “burden of disease” measure developed by Murray and Lopez (1996) can be used to estimate the health loss associated with air pollution. Severity weights are reported in Murray and Lopez (1996) for several hundred different health outcomes. Also, for 56 diagnostic groups separating more than 100 different disease stages disability weights for The Netherlands have been derived (Stouthard et al., 1997; Stouthard et al., 2000). Environmental disease related disability weights have been provided by De Hollander et al. (1999) based on Stouthard et al. (1997). De Hollander et al. (1999) used both the Global Burden of Disease project and the Dutch Burden of Disease project to attribute weight to environmental health impacts.

Disability-adjusted life years (DALYs) measure combines years of life lost (YLL) and years lived with disability (YLD) that are standardized by means of severity weights. Thus, the annual number of DALYs lost can be calculated as:

DALY = N x D x S. N is the number of cases, D is the average duration of the response, including loss of life expectancy as a consequence of premature mortality and S is the discount weights to the unfavorable health conditions.

For each human health endpoints related to PM, we apply values available from literature (De Hollander et al., 1999; Hofstetter, 1998). The resulting DALYs per case for premature mortality, chronic bronchitis, cardiovascular hospital admissions, and restricted activity days are 10.9, 0.31, 0.027 and 0.00027, respectively.

E.2. Cancer[7]

E.2.1. Exposure

Cancer risk is evaluated as a product of exposure and potency (or cancer potency factors). For the exposure calculations, intake fractions developed by Debbie Bennett were used (Bennett et al., 2002). The intake fraction of a toxic release is defined as the fraction of the pollutant mass emitted in the environment that is eventually inhaled, ingested or dermally absorbed by the affected population. In other words, it corresponds to the exposure level per unit of a pollutant emitted to the air and the water.

The major reason that the intake fraction methodology was used to address the human health cancer and non-cancer impact categories, in place of the standard TRACI characterization factors for cancer and non-cancer impacts, was to achieve consistency with the occupational risk assessment work (and thus provide the ability to sum and compare results). The TRACI factors provide an estimate of the *relative* cancer risk (among emissions, and among product alternatives) posed by the total exposures via environmental pathways for pollutants released across the life cycle. This relative risk evaluation is fine for comparing products or emissions. But in the present report, estimation of *absolute risk* per functional unit of product alternative is much more useful, to enable result integration and aggregation across impact categories.

In the population life cycle framework population cancer risk has been estimated for exposures during manufacturing and disposal. Occupational health risks, risks during use phase as well as risks to the fenceline population are treated in separate sections later. The use of intake fraction estimates enables the estimation of cancer risk from life cycle releases to the environment. These estimates of lost life years can then be compared and aggregated with the mortality and morbidity impacts from other impact categories (i.e., PM, mercury, global warming, occupational/use phase and fenceline population), since the latter have been converted to disability-adjusted life years (DALYs). This comparability and sum-ability enables decision makers to deal in an integrated way with impact categories that address different pathways to the same endpoint of concern (human mortality), and to assess the relative importance of these different pathways. It also enables decision-makers to compare and assess the *absolute* importance of the product life cycles on human health.

The intake fractions developed by Bennett et al. (2002) assume that each environmental medium is homogeneous with respect to the chemical concentration within the region under consideration. Population density and intensity of food production are likewise assumed to be uniform within the region. Macleod (2004) points out that the characteristics of the source locations are important predictors of intake fractions. For example, they found that the important parameter to consider for volatile chemicals such as benzene is local population densities, while

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[7] For cancer and non-cancer characterizations, TRACI uses human toxicity potentials (HTPs) developed by Edgar Hertwich (2001a). In his method, an HTP of a carcinogenic pollutant *i* is the dose and potency-dependent benzene-equivalent risk associated with a unit emission of the pollutant into the environment. Since the modeled domains for HTPs depend on the travel distance of each chemical, the information on the distance and population living in the affected area are needed to estimate risks. On the other hand, an advantage of using intake fractions is that they are already averaged for the United States and therefore there is no need to know the travel distances of each chemical. However, the existing intake fractions do not consider metals, and therefore we deal with mercury and other metal-related risks as a separate category.

that for persistent pollutants are local agricultural productions (e.g., leafy vegetables and grains for B[a]P and meat and milk for TCDD). While regional impact assessment is a valid delineation, our goal is to evaluate the total population health risks. In other words, our LCIA methods treat each individual within society equally.

We note that intake fractions are a screening tool for a comparative risk assessment or LCA (Bennett et al., 2002). While further refinement would be necessary for the evaluation of absolute risks associated with any particular product, we stop at the screening level evaluations of exposure. This approach is sufficient for our purposes for two reasons: (1) we compare different products and do not evaluate absolute risks (i.e., actual social impacts related to particular products), (2) the major contributors to the total human health risks such as dioxins from the disposal phase would be generated at the same locations across different products (e.g., landfills) and (3) we have found no evidence that, on average, important source locations of important contributors for one product are significantly different than the others in terms of proximity to agricultural lands and population density.

Using CalTOX ™, a multimedia total exposure model, Bennett has modeled intake fractions of 308 chemicals for an average person in the U.S.[8] In her model, population intake fraction is calculated as follows:

$$\textit{Population Intake Fraction} = \textit{Source to Dose } \{(mg/kg/day)x(mg/day)^{-1}\} x \textit{ Body weight } \{kg\} \textit{ x population}$$

E.2.2. Potency

For the cancer potency factors, values published by the U.S. EPA and California EPA (Cal/EPA) were primarily used (Hertwich, 2001b).

Using the population intake fractions, a cancer risk associated with X mg of pollutant *i* emitted

$$\textit{Population Cancer Risk(i)} = \textit{Dose(i) x Potency(i)}$$
$$= \left[\textit{Total emissions (i) } \{mg\} / 50\{yrs\}/365\{dys\} /BW \{kg\} \ x \ iF\_tot(i)\_pop\right] x \ CPF(i) \ \{1/mg/kg/day\}$$

over a 50-year period is calculated as follows:

where,

iF_tot(*i*)*_pop* is the total intake fractions of the U.S. population from all exposure pathways (inhalation, ingestion and dermal) for chemical *i* and CPF(*i*) is the iF-weighted cancer potency factors of chemical *i*.

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[8] Bennett et al. (2002) provides intake fractions for most of the semi-volatile organic pollutants for which toxicity information is available, and we use Bennett's intake fractions whenever possible. Dr. Dinah Koehler estimated intake fractions of some dioxins and polycyclic aromatic hydrocarbons in her dissertation. We use Koehler's intake fractions for the air emission of 1,2,3,4,7,8-HEXACHLORINATED DIBENZOFURAN, which is not available in Bennett et al. (2002). For the water emission of this pollutant we derived the characterization factors using the iF-based characterization factor for 2,3,7,8-TCDD and the ratio of HTPs for 1,2,3,4,7,8-HxDF and 2,3,7,8-TCDD. In addition, iFs for 2-methylphthalate have been modeled using CalTOX ™ and used in the derivation of characterization factors for non-cancer health effects.

To compare the magnitude of cancer risks with the other human health risks, cancer risk was converted into disability-adjusted life years (DALYs), which is the sum of the life years lost per cancer incidence and disability-weighted years lived with the disease (i.e., duration of disease). Patrick Hofstetter, in his dissertation, shows that based on the statistical data for Canada, USA, Hamburg, Germany and Switzerland, cancer incidence would lead to 13.1 years of DALYs as an average for various cancer sites including lungs, skin, nasal passages, nasopharynx, liver and leukemia, with no age adjustment (Hofstetter, 1998). Using his figure, the total DALYs associated with the life cycle emissions of air toxics are estimated as the product of the total cancer risk and 13.1 (DALYs per cancer incidence).

E.3. Non-Cancer

E.3.1. Exposure

For the estimation of population exposure per unit of emission, we use the same intake fraction approach as cancer risks.

E.3.2. Potency

For the non-carcinogenic effects, the information on the disease categories associated with a pollutant is not available. Instead, the threshold levels of an average daily dose (or reference dose, RfD) have been established by regulatory agencies. The hazard quotient of pollutant *i* is calculated as the ratio of dose to RfD. The hazard index (HI) is the sum of the hazard quotients and is used to compare the magnitude of non-carcinogenic risks associated with the products of our interest, assuming dose additivity.

Hazard quotient associated with Y mg of pollutant *i* emitted over a 50-year period is calculated as follows:

$$\begin{aligned} &Population\,Hazard\,Index\,(i) = Dose(i) \div ADI\,(i) \\ &\quad = \left[Total\ emissions\,(i)\ \{mg\} / 50\{yrs\} / 365\{dys\} / BW\{kg\}\ x\ iF\_tot(i)\_pop\right] \div ADI\,(i)\,\{mg/kg/dy\} \end{aligned}$$

where,

ADI (*i*) is the acceptable daily intake established as RfD (reference dose through) or RfC (reference concentration through inhalation).

E.4. Global Warming

For the characterization factors for greenhouse gases, we relied on the EcoIndicator approach, a damage-oriented LCA impact assessment method developed by Goedkoop and Spriensma (2001). We apply their marginal DALY values that are estimated based on the FUND model, a benchmarking model that calculates climate change-related damages including vector-borne diseases, infectious disease and psychological disorder related to sea-level rise, population displacement and damage to infrastructure.

Based on Goedkoop and Spriensma (2001), the DALY related to global warming is 0.00021 DALY per tonne of CO2 equivalents. However, this estimate involves a lot of uncertainties and has a squared geometric standard deviation of 2.4. Where human health related to global warming is a large driver of the total health impacts, we test the sensitivity of the characterization factor by using the lower 95% confidence limit 0.0000875 DALY per tonne of CO2 and CO2 equivalents.

E.5. Metals

The types of metals we considered for carcinogenic effects include: lead, cadmium, zinc, nickel and arsenic. Intake fractions for metals are not yet available, while the human toxicity potentials based on the same CalTOX model are available in TRACI. In order to test the magnitude of impacts from metals, we have developed tentative characterization factors based on human toxicity potentials, a toxicity- and exposure-weighted aggregated measure of pollutant releases. For cancer characterization of each metal, human toxicity potentials of TCDD and intake-fraction-based characterization factors (DALYs/g) for TCDD was used as a scaler to derive iF-based DALYs for each metal based on the metal HTP.

For cancer:

$$DALYs/g\ metal(i) = DALYs/g\ TCDD \times \frac{HTP/g\ metal(i)}{HTP/g\ TCDD}$$

The types of metals we considered for non-cancer effects include: arsenic, cadmium, mercury, lead, nickel, tin and zinc. For the non-cancer characterization of each metal, HI of TCDD was used instead of DALYs of TCDD to derive HIs related to each metal.

For noncancer:

$$HIs/g\ metal(i) = HIs/g\ TCDD \times \frac{HTP/g\ metal(i)}{HTP/g\ TCDD}$$

E.6. Mercury

Since mercury is one of the major pollutants of concern, we derive alternative health characterization factors based on Rice and Hammitt (2005) to estimate the population risks. This study characterizes the impact that reductions in mercury emissions from power plants would have on methylmercury exposures in the U.S. population and quantitatively estimates the economic benefit of the plausible improvements in the health of the U.S. population. In their analysis, five alternative emissions scenarios were considered for the patterns of mercury depositions: current emissions, mercury emissions in 2010 including projected changes in U.S. mercury emissions (Baseline 1), mercury emissions in 2010 with a 47% reduction in U.S. power plant emissions associated with implementing the Clear Sky Initiative (CSI)[9] (Scenario 1), mercury emissions in 2020 (Baseline 2), and mercury emissions in 2020 with a 69% reduction in U.S. power plant emissions associated with implementing the CSI (Scenario 2) (U.S. EPA, 2003b,c).

Based on their analysis, we use their freshwater fish (pike) impacts to male consumers as a lower bound and the freshwater and marine fish impacts as an upper bound estimate. The assumptions for the lower bound estimate are based on the Salonen male cohort data (Salonen et al., 1995). The rationale behind is that only pike consumers are affected by mercury emissions since pike has relatively high methylmercury and low n-3 fatty acids to antagonize the effects, while the non-pike freshwater fish consumers likely consume relatively higher levels of n-3 fatty acids from other fish that can antagonize the adverse effects of mercury. The assumptions for the upper bound estimate is that there are proportional relationships between emissions and fish methylmercury concentrations as well as health effects in both fresh and marine water.

For the lower estimate, the study estimates annual benefits of 7.4 QALY gained per ton of emission removed under Scenarios 1 for freshwater fish consumers, most of which are the result of decreases in premature mortality. For the upper estimate, the study estimates annual benefits of 807 QALYs per ton of emissions removed under Scenario 2 for freshwater and marine fish consumers, based on reduced neurotoxicity, non-fatal acute myocardial infarctions, and premature mortality. Assuming the equivalency of QALY and DALY, we assume 7.4 DALYs and 807 DALYs per ton of mercury emission as the lower and upper bounds of the mercury characterization factor.

E.7. Treatment of chemical groups

The TRACI characterization factors as well as the modified characterization factors that we developed in order to estimate mortality risk based on pollutant exposures (intake fractions) are chemical specific. However, a life cycle inventory, as generated in SimaPro and BEES, sometimes includes chemical groups for which we do not know the types and quantities of the constituents. In order to account for the impacts associated with those chemical groups new characterizations were developed. Such chemical groups include unspecified dioxins, polycyclic aromatic hydrocarbons (PAHs), aldehydes, phthalates and PM.

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[9] EPA's proposal to reduce power plant emissions of mercury through the application of a market-based "cap and trade" approach over the next two decades

For phthalates and aldehydes, the derived characterization factors for all those groups of chemicals are the simple averages of the TRACI factors for the chemicals that are potentially included in the group (see below). By taking the average of those characterization factors, we assume that the constituents in each chemical group are in the same proportions and that there are no other species than the ones listed below. Without any additional information regarding the constituents we are forced to proceed with this assumption. The groups of chemicals for which this simplifying assumption was used are:

- **Phthalates:** bis(2-ethylhexyl)phthalate, di-n-octyl phthalate, diethyl phthalate, dimethyl phthalate, butyl benzyl phthalate, di-n-butyl phthalate
- **Aldehydes:** acetaldehyde, formaldehyde, crotonaldehyde, benzaldehyde, c3 aldehydes, c4 aldehydes, c5 aldehydes, c6 aldehydes, c7 aldehydes, c8 aldehydes, glutaraldehyde, tolualdehyde

For PAHs, dioxins and PMs, we derived the best estimate of characterization factors as follows.

PAHs: acenaphthene, anthracene, benzo(a)anthracene, benzo(a)pyrene, benzo(b)fluoranthene, benzo[ghi]perylene, chrysene, dibenz(a,h)anthracene, fluorene, indeno[1,2,3-cd]pyrene, naphthalene, phenanthrene, pyrene

To derive the human health characterization factors for PAHs, EPA's emission inventories were used. EPA, in "Appendix H. Estimating carcinogenic potency for mixtures of polycyclic organic matter" of their National-Scale Air Toxics Assessment (1996a), selects four emission sources, namely residential wood combustion, aluminum industry, electric utilities, and wild fires, as the "representative PAH source categories," assuming that these source categories (comprising about three-quarters of all PAH emissions) have PAH emission profiles representative of all PAH sources. The same source categories were used in this analysis, assuming that the "representative" PAH emission profiles are similar to those that may be found for a variety of manufacturing processes. Based on approximately 40 individual PAHs included in their emission inventories, EPA has derived benzo(a)pyrene-equivalent toxicity-weighted total PAH emissions for the four "representative" sources as well as for the "PAH" emissions in general. Their toxicity-weighted emissions, however, do not account for the individual dose levels of different species. This forces the assumption of an equal dose for all chemicals, although this would be unlikely given their different chemical properties (e.g., vapor pressure, octanol-water partition coefficients, half-life, etc). To improve the analysis, cancer and non-cancer characterization factors were developed that take into account both toxicity and dose levels of the potential PAH constituents using intake fractions. Since the number or types of species included in the "PAHs" are not known or present in life cycle inventories, the characterization factors were calculated based on two different assumptions —(1) "PAHs" in LCI include only those species that are known to be toxic (7 carcinogenic PAHs and 9 non-carcinogenic PAHs; i.e., a worst case scenario), and (2) the "PAHs" in LCI include all potential constituents (>40 species) in EPA's emission inventory (U.S. EPA, 1996a). These assumptions were carried through the analysis of human health cancer and non-cancer impacts, contributing to the variability of the impact estimates.

Dioxins (unspecified): 1,2,3,4,6,7,8-heptachlorodibenzofuran, 2,3,4,7,8-pentachlorodibenzofuran, 2,3,7,8-tetrachlorodibenzofuran, 2-butyl tetrahydrofuran, alpha-methyltetrahydrofuran, carbofuran, furan, hexachlorinated dibenzofuran, 1,2,3,4,7,8-, tetrahydrofuran, 2,3,7,8-tetrachlorodibenzodioxin

For unspecified dioxins, we assume the mass ratio of total dioxins to dioxinTEQ to be 60 (total dioxins/dioxins TEQ = 60), based on EPA's assessment (U.S. EPA, 1994) that it is a typical mass ratio for MSW incineration.

Particulate matter (unspecified)

Some life cycle inventory data included pollutants named "unspecified particulate matter". However, only fine particles in the total particulate matter mass are considered harmful. For the U.S., the fine fraction in the background concentrations of PM10 and total suspended particles are typically 60% and 33% of the total mass. Without specific information on size fractions, we treat unspecified particulate matter the same as total suspended particulate matter (i.e., 33% of the total mass in the fine mode).

Appendix F: Human Health Risk Assessment

Risk assessment is the process of estimating the likelihood of harm to an individual or population due to a set of circumstances. Risk assessments calculate the likelihood of one's being in a car accident, or falling in the bathtub; they calculate the likelihood that one might die in an airplane crash, or the possibility that one might outlive their savings. In this analysis, we have performed risk assessments to estimate the likelihood that occupational workers, installers, and the general public might have adverse health effects from exposure to the building materials and the chemicals and processes that produce them. Risk assessment is often a necessary first step to properly manage risk.

Risk assessment is a means of using certain facts known about the conditions of a location, such as the workplace, to define the possible health effects in individuals or whole populations exposed to hazardous materials and situations. Toxicological risk assessment involves four distinct steps: hazard identification, dose-response assessment, exposure assessment, and risk characterization (Commission on Life Sciences, 1983).

- Hazard identification: the determination of whether a particular chemical is or is not causally linked to particular health effects
- Dose Response Assessment: the determination of the relation between the magnitude of exposure and the probability of occurrence of the health effect in question
- Exposure Assessment: the determination of the extent of human exposure before or after application of regulatory controls
- Risk Characterization: the description of the nature and often the magnitude of human risk, including attendant uncertainty.

This section describes how these four steps were completed for each of the compounds involved in the production of the selected PVC and non-PVC building materials.

F.1. Hazard identification

The chemical compounds used in the manufacture of vinyl building products and competing materials were obtained from various reports on building materials and their applications (e.g., American Institute of Architects, 1996), published literature, and on-line data from manufacturers of these products.

The compounds considered in the occupational/end user risk assessment are the following:

The compounds evaluated in the risk assessment are the following:

Pipe	Windows and Siding	Flooring
VCM	VCM	acetaldehyde
EDC	barium	acetone
Acrylonitrile	dibutyltin stabilizers	benzyl alcohol
1,3-butadiene	coke oven emissions	2-butoxyethanol
Styrene	silica	ethylene glycol
iron oxide fume	pyrene	naphthalene
coke oven emissions	benzo(a)pyrene	propionaldehyde
Limestone	aluminum	toluene
Silica	fluoride	trimethylsilanol
Manganese		phenol
Pyrene		formaldehyde
benzo(a)pyrene		furfural
		VCM
		DEHP
		Vinyl acetate
		EDC

F.2. Dose-response Assessment

F.2.1. Exposure limits for the general public

The dose-response assessment defines the relationship between the dose of the chemical taken into the body and the probability that an adverse effect will result from that dose. “A dose response usually requires extrapolation from high to low dose and extrapolation from animals to humans” (NRC, 1983).Two different types of dose-response relationships are commonly associated with exposure to chemicals at high doses or at low doses: threshold and non-threshold (linear low-dose).

Threshold relationships are generally assumed for non-cancer (systemic) effects of toxicants. A threshold relationship assumes there is a dose below which no adverse effect will occur (a dose that the body can accommodate physiologically). The U.S. EPA and other organizations have developed Reference Doses (RfDs) for oral exposure or Reference Concentrations (RfCs) for inhalation exposures for comparison to the estimated average daily dose (defined below). The ratio of the average daily dose and the RfD is called the Hazard Index (HI). A HI greater than unity implies that the estimated average daily dose exceeds the published RfD.

Non-threshold relationships are generally assumed for carcinogenic chemicals that have a linear dose-response curve with no threshold. In other words, a finite risk of a carcinogenic response is

possible with any non-zero exposure. The U.S. EPA has developed a generally conservative (i.e., more likely to overestimate than underestimate potential risks) method for summarizing the dose-response information for carcinogens using a Cancer Slope Factor (CSF). There are many models used to determine the shape of the dose-response curve at very low doses, doses generally below those actually measurable or tested. One such model used by the U.S. EPA is the linearized multi-stage model. The slope factor for humans generally is estimated from the 95 percent upper bound of the slope of the linearized multi-stage model fit to the animal data. The U.S. EPA recognizes that the true carcinogenic potency of a particular compound lies—with 95 percent probability—between zero and the published CSF.

Where available, dose-response values for both ingestion/dermal and inhalation routes of exposure are provided. When values for only one route of exposure were available, these values were extrapolated to the other route. Although this extrapolation is not specifically recommended by any state or federal agency, this practice is an accepted method in the estimation of risks from exposure to hazardous chemicals. Because occupational exposures were the focus of this exercise, the pathways of concern were inhalation and dermal exposures. However, in the interest of simplicity, only the inhalation pathway was considered; it is the primary pathway of exposure to most occupational compounds and the majority of chemicals considered in this assessment were volatile, and/or would exhibit limited dermal uptake (metals, wood dust, silicates). The importance of dermal exposures in the workplace and the contribution of this pathway to risk, particularly cancer risk, are being recognized, however (Jones et al., 2003; Kalnas and Teitelbaum, 2000). Dermal exposures to certain compounds (e.g., solvents) are generally easily managed by using gloves and other personal protective equipment.

It is useful to discuss the relevance of the Hazard Index (HI) and Integrated (Excess) Lifetime Cancer Risk (ILCR) values in this assessment. As mentioned previously, an HI>1 indicates that the RfD has been exceeded by the daily exposure dose of the modeled exposed individual (worker, child, etc.). An HI>1 is generally a departure point for regulatory action by a state environmental protection agency or by the U.S. EPA. For example, a state regulatory agency might require a responsible party to clean up a contaminated property if it has been concluded that exposure to environmental contaminants in the soil, ground water, or air would result in estimated non-cancer risks (e.g., threshold risks) to potential receptors (people on the property) exceeding acceptable limits (e.g., HI of 1). Estimated cancer risks are handled in the same way. The U.S. EPA generally considers a cancer risk of less than 1 in 1 million (1E-06) to not require any action, such as a site cleanup, while those greater than 1 in 10,000 (1E-04) prompt site remediation or some other action. Risks in between these values are handled as appropriate to minimize risks based on site-specific conditions, future uses, and other considerations. State regulatory agencies typically set cancer risk limits of 1E-06 or 1E-05. Workers in some industries, however, may face higher absolute cancer risks because OSHA and other agencies allow workplace chemical concentrations to be much higher than those intended for the general public. *But these agencies often use other management or regulatory tools to control exposure.*

The risk assessment uses RfDs and CSFs gathered primarily from chemical profiles in the U.S. EPA's Integrated Risk Information System (U.S. EPA, 2004a). When limits were unavailable from that source, values from a recent version of the U.S. EPA's Health Effects Assessment Summary Tables (U.S. EPA, 1997c), the Risk Based Concentration Table from Region III of the U.S. EPA (U.S. EPA, 2004b), the Risk Assessment Information System, Oak Ridge National Laboratory (RAIS, 2004), or Minimal Risk Levels from the Agency of Toxic Substances and

Disease Registry (ATSDR) were used. In the absence of values from these secondary sources, sources developed by California EPA (Cal/EPA) were consulted. If values were not available from any of these sources, they were not derived by any other means. There are 7 compounds that currently lack exposure limits (dialkyl tin compounds, two silica compounds, cellulose, wood dust, metal fume, and trimethylsilanol).

The majority of the compounds are VOCs, but metals (including heavy metals and organometallics), silicates, PAHs and combustion products (coke oven emissions), and wood dust are also included. The majority of these compounds are non-carcinogenic; ten compounds are known or believed to be carcinogenic. These are vinyl chloride monomer, ethylene dichloride, di(ethylhexyl)phthalate (DEHP), coke oven emissions, tetrahydrofuran, acetaldehyde, formaldehyde, acrylonitrile, 1,3-butadiene, and benzo(a)pyrene.

The toxicity values for some of the compounds evaluated in this assessment have not been revised in several years (ca 1987 and later). An attempt has not been made to do an extensive review of more recently published toxicity studies for these compounds to determine if the RfDs and/or CSFs are overly conservative or not conservative enough. Promulgation of revised values in the future may change the resultant risk estimates for these compounds; this may in turn, affect the overall interpretation of the relationship of occupational health risks for each building material to its effects on the environment and the general population.

F.2.2. Discussion of general population exposure limits for individual compounds

As discussed previously, the compounds considered in the human health risk assessment are toxic to varying degrees. Many of these compounds are carcinogenic. In reviewing the literature and the toxicity values for the compounds, it became clear that the toxicity values did not necessarily reflect all the research performed for the compound in question. A discussion of some of the hypotheses regarding toxicity of vinyl chloride and DEHP follows.

F.2.3. Vinyl Chloride

High exposures of vinyl chloride monomer in humans are known to result in angiosarcomas of the liver, a very rare cancer of this organ (MMWR, 1997). For example, 23 cases of angiosarcoma were identified in a Louisville, KY plant, representing 30% of the cases of this disease in the vinyl industry (Lewis and Rempala, 2003). Since the establishment of the OSHA PEL of 1 ppm VCM, however, no new cases of angiosarcoma have been identified according to the vinyl industry (Vinyl Institute, 2004). VCM is speculated to be the etiologic agent for other types of cancer in exposed workers; published studies have not supported the link however (Lewis and Rempala, 2003).

A 20% increased risk of lung cancer for every year worked was reported in Italian PVC baggers exposed to PVC dust; cumulative exposure to VCM was not associated with the lung cancer cases (Mastrangelo et al., 2003). Breast cancer was not found to be increased in PVC fabricators in one study (Chiazze et al., 1980); exposure data were not available, but given the date of the study, exposures were likely to be to relatively high concentrations of VCM or residual VCM. Foreign studies indicate that occupational exposures to VCM concentrations lower than those resulting in angiosarcomas also are associated with increased incidences of cancers of the hematopoietic and lymphatic systems (Smulevich et al., 1988; Wong et al., 2002) and malignant melanomas (Lundberg et al., 1993; Heldaas et al., 1987). These tumors have not been reported in

American workers, however. The differences may be due to a spurious association of the lesser-researched cancers with VCM exposure, or they may be the result of different occupational exposure concentrations, or the result of other factors (e.g., differences in genetics, etc).

F.2.3.1. DEHP-Non-cancer

Phthalates, particularly di(2-ethylhexyl)phthalate, are used to make vinyl products flexible. The most commonly used phthalates in vinyl flooring are diisohexyl phthalate (DIHP), butyl benzyl phthalate (BBP), and diisononyl phthalate, while dibutyl phthalate (DBP) and DINP are also found in many consumer products including toiletries, cosmetics, and flexible children's toys. DEHP was used as a model plasticizer in this analysis because historically it was used in flexible vinyl flooring and its toxicity tends to be greater than the other phthalate moieties in most *in vitro* and *in vivo* assays. All phthalates are metabolized by lipases (esterases), present in the mammalian gut, liver, and blood cells, into their respective mono derivatives; in the case of DEHP, the primary metabolite and ultimate toxicant, is mono(ethylhexyl)phthalate (MEHP).

As discussed more fully in subsequent sections, DEHP is not the predominant phthalate used to plasticize vinyl flooring or vinyl tiles. Nevertheless, until recently it was used extensively to plasticize those building materials, and is therefore still present in commercial and residential buildings. Further, current toxicity studies indicate that it is the most toxic of all phthalates used. Therefore, DEHP was used as a surrogate for other phthalates, the understanding that risk estimates derived for this compound are likely therefore to be greater than those derived for any other phthalate. DEHP as a surrogate is a useful screening tool.

The current RfD for DEHP is based on an outdated (1953) study involving exposures to guinea pigs and the sensitive effect was increased liver weight (U.S. EPA, 2004a). Studies published in the last ten years have focused on the effects of DEHP and other phthalates on reproductive organs in both male and female rodents, with particular attention given to neonatal males. It appears that the reproductive organs may be particularly vulnerable as target organs when exposure occurs during early development, as well as exposure in utero (during pregnancy; CERHR-DEHP, 2005). The U.S. EPA is currently reviewing the current body of published data on the toxicity of DEHP and is developing a new RfD value for this compound. A reproductive or developmental toxicity study will most likely be chosen to represent the most sensitive endpoint of non-cancer toxicity; it is possible, although unlikely, that the RfD will be lower than the one currently used (0.02 mg/kg-day), although it could be lower by an order of magnitude (depending on the NOAEL in the study chosen and the composite uncertainty factor selected). Even so, the resultant risk estimates from exposure to DEHP would likely only be in error by one order of magnitude, and these are believed to be overestimates. Therefore, while the limitations of the current RfD for DEHP are acknowledged, they are not believed to introduce an unacceptable degree of error in the resultant estimates.

F.2.3.2. DEHP-Cancer

DEHP causes liver cancer in rodents, which is the result of MEHP-induced peroxisomal proliferation, specifically that of receptor alpha (PPARα), in the rodent liver. Extensive data have been published regarding the mechanism of liver carcinogenesis of DEHP in the rodent model. Humans and non-human primates have a low level of PPARα expression (Palmer et al., 1998; Pugh et al., 2000) and there are species differences in PPARα responsiveness (Mukherjee et al., 1994), suggesting that humans are likely not susceptible to PPARα-mediated liver carcinogenesis (Doull et al., 1999). As discussed previously, the U.S. EPA is currently re-evaluating both the non-cancer and carcinogenic dose-response of DEHP in animals and humans

and is preparing revised RfD and CSF values for this compound. These data are not yet available. The general scientific opinion is that DEHP is unlikely to be a liver carcinogen in humans. For the purposes of this assessment, DEHP has been considered as non-carcinogenic in the human.

One group of researchers has suggested that exposure to PVC contributes to the incidence of testicular cancers in occupationally-exposed males in Sweden (Ohlson and Hardell, 2000). They conducted a retrospective study using self-administered questionnaires in men who had worked in several different industries; there was a statistically increased risk of testicular cancer (both embryonal and seminomas) in PVC workers. The year of first exposure of men with cancer ranged from 1957 to 1979, and the latency period ranged from 11 to 35 years following first exposure. The study authors speculated that phthalates in the plastics might be the underlying cause of the testicular cancer, but had difficulty reconciling this effect with the anti-andronergic effects of DEHP in rodent models (DEHP causes seminiferous tubule atrophy, and decreased testis weight, sperm production, and testicular zinc levels [as discussed in Lovekamp-Swan and Davis, 2003]). Further, in a more extensive study of PVC workers published later (Hardell et al., 2004), the researchers found no connection between exposure to compounds in the manufacture of PVC and testicular cancer. It is interesting to note that the incidence of testicular cancer in Sweden and of other western European countries has risen dramatically over the last few decades and a birth cohort phenomenon has been observed (Bergström et al., 1996). Due to the lack of corroborating data in rodents and supporting studies in humans, these data indicate the need for future analysis before conclusive statements can be made with regard to PVC and testicular cancer. The majority of the data indicate that phthalates induce liver cancer in rodents via a mechanism that occurs only weakly, if at all, in humans. Therefore, DEHP was not considered carcinogenic to humans for the purpose of this analysis.

F.2.4. Occupational exposure limits

Industrial hygiene groups and government agencies that recommend or promulgate occupational exposure limits also review the available scientific data, including human and animal toxicity studies, and determine dose-response relationships similarly to the U.S. EPA. They also determine the relevance of these effects to humans, using supportive data that include pharmacokinetic modeling. Typically, however, they use a different methodology to develop their resultant exposure limits. For example, both the American Conference of Governmental Industrial Hygienists (ACGIH) and the Occupational Safety and Health Administration (OSHA) use a methodology that incorporates a margin of exposure (MOE) or safety factor in their recommended exposure limits. The safety factor may vary but values ranging from 10-100 are typical. In other words, the exposure limit established may be 10 to 100-fold lower than the lowest exposure dose found not to result in an adverse effect in animal models. This exposure dose is called the No Observed Adverse Effect Level (NOAEL). For example, if the most sensitive NOAEL in a series of animal studies was identified as 100 ppm, an exposure limit might be set at 10 or even 1 ppm, depending on the decision of the ACGIH committee (or OSHA). This value typically does not take into account differences in susceptibility among the potentially-exposed human populations. Exposure limits thus are established that are generally believed to be levels that will not pose an adverse health risk for most, but not all workers. It is noted by ACGIH, however, that its TLV values do not represent "…fine lines between safe and dangerous concentrations…(ACGIH, 2004a)." Further, OSHA PELs are enforceable, while ACGIH TLVs are recommended values only. In addition, unlike OSHA, ACGIH technical

committees that recommend TLVs do so based only on a review of the toxicological data available at the time (ACGIH, 2004b); economic and technical feasibility of achieving the recommended limit is not considered. It follows that if an industry cannot either technically or economically reach a recommended TLV value for one or more compounds, then a higher exposure concentration will be present within the manufacturing facilities of that industry. It is expected that companies strive to put good industrial hygiene practices into place in most, if not all, of the industries represented in this report. It is possible to exceed a regulatory guideline value for a limited number of times during the working day as long as the time-weighted average is not exceeded. Therefore, risk estimates developed in this report have relied solely on OSHA PELs in the absence of more realistic exposure data, although it is clear that in many occupational situations these PELs may be exceeded briefly during the day, or exposure may be lower than the PEL. Because the PELs are enforceable, industry practice is to attempt to stay below the limit during most work pracices to allow the spikes of exposure to occur during the rest of each working day and still remain in regulatory compliance. Thus assuming the exposure level is the same as the PEL is conservative.

Some compounds used in current manufacturing or installation processes for PVC and non-PVC building materials do not have promulgated PELs or recommended TLVs. Recommended exposure limits are not proposed in this report. Those compounds lacking both exposure data and promulgated exposure limits have not been carried through the analysis.

It is important to note that although this report attempts to make comparisons between PVC and non-PVC building materials with regard to occupational risk (both cancer and non-cancer) these values are likely to be overestimates of the actual health risks to these workers. There are a number of reasons for this. In some instances, the exposure concentration was assumed to be the full OSHA PEL, and it was assumed this exposure concentration was maintained throughout the entire tenure of the worker; this assumption is conservative, however, as concentrations of compounds in the working environment may vary depending on the particular activities occurring. Because the resulting daily dose estimates were compared to Reference Doses (RfDs) and Cancer Slope Factors developed for the general population, the resulting risk estimates may overestimate the true health risk. The RfDs and CSFs are developed assuming constant exposure to a particular environmental toxicant; however, occupational exposures are generally limited to 8-10 hours per day, 5 days per week. This decreases the overall dose. Further, RfDs and CSFs are developed to be protective of the most sensitive members of the general population, including the very young and the very old who are not typically present in an occupational environment.

Our approach is more conservative, but also allows the comparison of estimated risk values between building materials, as well as to other receptors to which the same methodology is applied, such as residents or those who experience sporadic exposure from upset conditions or incidental leaks (e.g., fenceline exposures). This assessment is just a starting point for comparison of PVC and competitive building materials. It does not include the assessment of acute exposure risks, predominantly because exposure data and appropriate acute limits are unavailable. This gap in analysis is not considered to be significant because one of the major goals of this exercise was to compare occupational risks, which occur over several months to several years, to those of the general population. That comparison necessitates an investigation of chronic health effects.

F.3. Exposure Assessment

The exposure assessment is based on a set of site-specific exposure scenarios that define potential receptor groups (a receptor is considered to be an individual in a unique group such as a worker or a person living close to a manufacturing plant), potential exposure pathways, and concentrations of the toxicants in particular environmental media, such as air. These three components are combined to estimate pathway-specific doses of study chemicals to receptors.

F.3.1. Hypothetical receptors and pathways for exposure to toxicants

Although many workers are involved in the manufacture, installation, end use, and disposal of building products, in some cases a lack of accurate exposure data prevents the estimation of potential health risks from these exposures. For example, people involved in the transport of vinyl chloride monomer from facilities that manufacture the compound to those that use it may be exposed via leaks from the transport trucks. It is not possible to determine what those exposures might be however, with any accuracy, and to attempt to bound the risk range may result in wide-ranging over- or under-estimates of order of magnitude risk estimates. Occupational workers involved in the manufacture of these materials often receive the highest dose of a particular compound, either via inhalation or dermal exposure. These two pathways are considered by many governmental standard-setting agencies to be the most relevant for occupational receptors. This report focuses primarily on inhalation exposure for the following reasons: this pathway tends to contribute the most to resultant health risk estimates due to decreased absorption via the dermal pathway; accurate estimates of dermal exposures are difficult to quantify; and use of gloves and other personal protective equipment, recommended by industrial hygienists and OSHA, mitigate dermal exposure In other words, it is much easier to avoid dermal exposure to occupational chemicals, than it is to avoid inhalation exposure.

1. Occupational Worker -- For the purposes of this assessment, the occupational worker was considered to be either male or female, with a body weight of 70 kg[10] and an inhalation rate of 1.3 m^3/hr[11]. These exposure values are informed by U.S. EPA's Exposure Factors Handbook (U.S. EPA, 1996b) and professional judgment. The assumed body weight is slightly less than currently recommended by U.S. EPA (1996b), but because the toxicity values have been derived based on a 70 kg person, this weight value has been used for consistency. The worker is considered to be actively engaged at work for 8 hours/day, 5 days/week, 50 weeks per year for 15 years. Recent data from the Bureau of Labor Statistics show that within the private sector, manufacturing workers had a median tenure of 5.5 years in the major industry groups (BLS, 2002). Median tenure of primary metal industries was 7.6 years. Therefore, an assumed tenure of 15 years is believed adequate to capture representative exposures of occupational workers.

Occupational exposure to toxicants emitted from flooring materials was also assessed for flooring installers. For this receptor, the inhalation rates, body weights and working tenure are assumed to be the same.

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[10] The assumed body weight is slightly less than currently recommended by U.S. EPA (1996), but is used because toxicity values have been derived based on a 70-kg person.

[11] This value represents an intermediate respiratory rate between rest and heavy exertion and is believed to be a good approximation of the varying activity levels of an occupational worker.

2. End user -- End users comprise all populations that may be exposed to the building materials after installation. All age groups are represented. Although many of these building materials are also used for residences, residential exposures for vinyl products are not considered here as most available exposure data do not differentiate between sources of indoor air contaminants. In other words, the few available studies of indoor air pollutants believed to originate from vinyl products in a residence do not discriminate between resilient flooring and wall coverings and other flexible vinyl products, the latter of which are not considered in this report.

To assess potential end user exposure from flooring materials, a classroom scenario was modeled. The receptors were high-school students (male and female) who were present in the classroom 5 hours/day, 5 days/week for 38 weeks for one year. The inhalation rate was estimated to be 0.72 m^3/hr and the body weight was 61 kg. The inhalation rate represents the 50th percentile ranking of predicted inhalation rates for high-school students (aged 13-17) for slow indoor activity, which was assumed appropriate for a classroom environment (U.S. EPA, 1996b). The body weight represents an average between male and female students in this age range.

F.3.2. Formulas for estimation of doses

Based on the selected exposure scenarios, appropriate formulas were identified to estimate the average daily doses of each chemical to the modeled receptors via inhalation. These formulas are presented in Table F-1.

Table F-1: Formulas used to estimate doses

1. Formulas for Average Daily Dose for a Day of Exposure-ADD(d)

For Inhalation of Volatile Organic Compounds or Dusts:

$$ADD(d)\left(\frac{mg}{kg \times d}\right) = \frac{EPC_{air}\left(\frac{mg}{m3}\right) \times InhR\left(\frac{m3}{hr}\right) \times ED\left(\frac{hr}{d}\right)}{BW(kg)}$$

where:

ADD(d)	=	Average Daily Dose for a Day of Exposure
EPC_{air}	=	Average Concentration of Study Chemical in Air
InhR	=	Average Inhalation Rate
ED	=	Exposure Duration--Hours per Day
BW	=	Body Weight of receptor

2. Formulas for Average Daily Dose Averaged Over a Year of Exposure-ADD(y)

ADD(y):

$$ADD(y)\left(\frac{mg}{kg \times d}\right) = ADD(d)\left(\frac{mg}{kg \times d}\right) \times \frac{D\left(\frac{d}{wk}\right)}{7\left(\frac{d}{wk}\right)} \times \frac{WK\left(\frac{wk}{y}\right)}{52\left(\frac{wk}{y}\right)}$$

where:

ADD(y)	=	Average Daily Dose Averaged Over a Year of Exposure
ADD(d)	=	Average Daily Dose for a Day of Exposure
D	=	Days of Exposure per Week
WK	=	Weeks of Exposure per Year

3. Formulas for Average Daily Dose Averaged Over a Lifetime-ADD(l)

ADD(l):

$$ADD(l)\left(\frac{mg}{kg \times d}\right) = ADD(d)\left(\frac{mg}{kg \times d}\right) \times \frac{D\left(\frac{d}{wk}\right)}{7\left(\frac{d}{wk}\right)} \times \frac{WK\left(\frac{wk}{y}\right)}{52\left(\frac{wk}{y}\right)} \times \frac{Y(y)}{Life(y)}$$

where:

ADD(l)	=	Average Daily Dose Averaged Over a Lifetime
ADD(d)	=	Average Daily Dose for a Day of Exposure
D	=	Days of Exposure per Week
WK	=	Weeks of Exposure per Year
Y	=	Years of Exposure in a Lifetime
Life	=	Years in Lifetime

4. Formulas for Average Daily Dose Averaged Over a Lifetime-(multiple life stage exposure)-ADD(l)

ADD(l):

$$ADD(l)\left(\frac{mg}{kg \times d}\right) = \left[\sum\left(ADD(d)_{ls}\left(\frac{mg}{kg \times d}\right) \times \frac{D_{ls}\left(\frac{d}{wk}\right)}{7\left(\frac{d}{wk}\right)} \times \frac{WK_{ls}\left(\frac{wk}{y}\right)}{52\left(\frac{wk}{y}\right)} \times Y_{ls}(y)\right)\right] \times \frac{1}{Life(y)}$$

where:

ADD(l)	=	Average Daily Dose Averaged Over a Lifetime
$ADD(d)_{ls}$	=	Average Daily Dose for a Day of Exposure in a Life Stage
D_{ls}	=	Days of Exposure per Week in a Life Stage
WK_{ls}	=	Weeks of Exposure per Year in a Life Stage
Y_{ls}	=	Years of Exposure in a Life Stage
Life	=	Years in Lifetime

F.3.3. Exposure point concentrations

The Exposure Point Concentrations for the different compounds to which the receptors might be exposed were obtained from industry, when available, from published peer-reviewed studies (epidemiological, etc.) or agency reports (e.g., ATSDR Toxicity Profiles, CERHR Reports) or were assumed to be equivalent to OSHA PELs. Although not ideal, OSHA PELs were assumed to be upper bound concentrations that were used to model worst-case risk estimates for compounds that lacked accurate estimates of exposure. The EPCs are presented at the beginning of each risk estimate table with explanatory footnotes describing the origin of the values.

Exposure values for occupational flooring installers and end users were obtained from a building materials emissions study prepared by the California Integrated Waste Management Board (CIWMB, 2003) for vinyl sheet flooring, vinyl composition tile (VCT), and linoleum. To assess the impacts from installing cork flooring, emissions data from a published study on cork manufacture was used (Horn et al., 1998).

The emissions study for the California Integrated Waste Management Board (CIWMB, 2003) was prepared to compare the emissions of commercially available rubber-based and non-rubber based flooring materials and those with recycled content. The CIWMB has promoted the use of recycled-content materials for use in sustainable buildings. This study was designed to determine emissions from these products and the resultant air concentrations in modeled school classrooms and in state office buildings. Samples of flooring were obtained, removed from their original packaging and placed in aluminum foil. Samples (6 inch square) were cut and preconditioned for 10 days; the samples were then applied to a steel plate using recommended adhesives. The date of one sample was known; the ages of the other flooring samples were not reported. The report indicated that not all flooring samples could be obtained directly from the manufacturers and therefore, had to be obtained from commercial sources. According to the report authors, "the emissions from undated samples may be more realistic in terms of the actual 'real world' exposures."

After this time, emission factors for the flooring materials were analyzed via laboratory testing in an environmental chamber at 24, 48, and 96 hours. The emission factors were used to estimate concentrations in a standard-size (24 x 40 x 8.5 ft) classroom. The flooring materials were as follows:

- vinyl sheet flooring, felt backed, with adhesive (VOC, ≤5 g/L)
- vinyl composition tile (54 days old), with latex adhesive (listed as "zero-VOC")
- linoleum sheet, jute backing, with styrene-butadiene rubber adhesive (listed as "zero-VOC").

Concentrations of emissions from cork flooring were estimated in the same manner for classrooms using the emission factor data presented in Horn et al. (1998). These concentrations were used to model potential risk to a receptor in the classroom as they were used to model risks to workers exposed while laying the flooring.

As the CIWMB emissions study was primarily concerned with indoor air quality, the laboratory analyses focused only on emitted VOCs. It did not measure levels of phthalates or other compounds in suspended dust that are known to migrate from particulates emitted from vinyl sheet flooring or vinyl composition tile (Øie et al., 1997; Rudel et al., 2003). Further, this assessment has not included exposure to DEHP from these two sources for construction workers (flooring installers) or teenagers exposed in a classroom. Residential (child, teen, and adult) risks from exposure to DEHP in the home were estimated, however, using DEHP concentrations in air and dust in homes on Cape Cod, Massachusetts (Rudel et al., 2003). These exposures were considered for longer periods of time than those for the school scenario discussed above.

F.4. Human Health Risk Characterization

The risk characterization combines information gathered in the dose-response assessment and in the exposure assessment to estimate potential risks that may result from exposures to the varying compounds. As discussed in the previous section, exposures are quantified through the estimation of average daily doses; these doses then are combined with toxicity information to predict potential systemic health effects and cancer effects.

Table F-2: Formulas used to estimate non-cancer health hazards and cancer risks

1. Formula Used to Estimate Chronic Hazard Indices

$$HI = \frac{ADD(chronic)\left(\frac{mg}{kg \times d}\right) \times RAF}{RfD\left(\frac{mg}{kg \times d}\right)}$$

where:

ADD	=	Average Daily Dose for a Chronic Exposure Period
RAF	=	Relative Absorption Factor for the Exposure Route (e.g., inhalation)
RfD	=	Reference Dose for the Exposure Route (e.g., inhalation)

2. Formula Used to Estimate Incremental Lifetime Cancer Risks

$$ILCR = ADD(l)\left(\frac{mg}{kg \times d}\right) \times RAF \times CSF\left(\frac{kg \times d}{mg}\right)$$

where:

ILCR	=	Incremental Lifetime Cancer Risk for Exposure During Some Fraction of a Lifetime (probability)
RAF	=	Relative Absorption Factor for the Exposure Route (e.g., inhalation)
CSF	=	Cancer Slope Factor for the Exposure Route (e.g., inhalation)

Table F-2 above presents the formulas used to estimate the chronic Hazard Index (HI) for systemic health effects. The HI value is the ratio of the product of the average daily dose of a chemical over a period of exposure (either one year or less than a year), its appropriate Relative Absorption Factor (RAF; 1 for inhalation exposures) and the appropriate Reference Dose (RfD). This table also presents the formula used to estimate the Incremental Lifetime Cancer Risk (ILCR), which is the analogous product of the average daily dose of a chemical--over a lifetime--and its appropriate RAF, and the corresponding Cancer Slope Factor (CSF).

The use of RAF of unity for the inhalation exposure pathway is accepted in risk assessment. Inhaled compounds, particularly VOCs, are assumed to be absorbed completely from the lungs. This can overestimate the absorbed dose for some compounds, however, as it has been shown that the uptake of inhaled styrene is 68% (Wigaeus et al., 1983) and that of 1,3-butadiene is approximately 45% in both sexes (Lin et al., 2001). Nevertheless, as research was not performed

to determine inhalation uptake of each compound for each building material, it was appropriate to assume 100% absorption for each inhaled compound.

F.5. Uncertainty Analysis in the Risk Assessment

Uncertainty analysis is an assessment of the consequences of input uncertainties and modeling uncertainties on the resulting conclusions. Uncertainty analysis has been incorporated into this assessment in several ways. In this risk assessment, a deterministic (e.g., non-probabilistic) method was used to estimate conservative risk values, one that is protective of human health. Point values were assumed for concentrations of occupational toxicants. Whenever published or industry-provided exposure data were available to determine high- and low-end exposures, these data were used. When the data were unavailable, OSHA PELs were used as exposure concentrations. Using these values may introduce some uncertainty in the resultant risk estimates, but is protective of human health because it provides an upper bound to the potential risk. Most compounds regulated by OSHA have Action Limits that are one-half the PEL; when ambient concentrations for toxicants reach the Action Limit, the company must initiate a hygiene plan for all employees that may be affected. This hygiene plan includes biological monitoring of the affected personnel, including urine and blood analyses for determination of biomarkers of exposure. Using the PEL in the absence of measured exposure data is a common screening practice. The risk assessment erred on the side of inclusiveness, including any chemicals thought to be of concern. Ones that were omitted were either measured below ACGIH TLV values or lacked toxicity values.

A third source of uncertainty is the use of outdated risk values, such as the RfD for DEHP and the lack of toxicity values for some toxicants. The assessment used Reference Doses and Cancer Slope Factors that are themselves conservative values, developed using uncertainty factors (UFs) that are incorporated to account for limitations in data, extrapolation between animal toxicity studies to the human experience, and variation in human sensitivity to toxicant exposure. Therefore, the processes used in this risk assessment have allowed for the development of an upper and lower bound on estimated risk that is consistent across all building materials within each class.

Appendix G: Non-cancer and Cancer Risk Estimates Normalized per Functional Unit of Building Material

Table G-1: Non-cancer and cancer risk estimates normalized per functional unit of building material

Life Cycle Phase	pwhr/$	HI	Cancer (ILCR)	Retail $/f.u.	Manufact. $ per f.u.	Total HI/f.u.	ILCR/ f.u.
Pipe: PVC							
Resin Low	0.008	36	7.4E-02				
Resin High	0.008	190	3.7E-01				
C2S Low	0.008	0.56	--	1.4	0.93	2.86E-05	3.68E-09
C2S Likely	0.008	1	1.9E-03	1.4	0.93	2.99E-05	4.06E-09
C2S High	0.008	12	3.8E-02	1.4	0.93	1.78E-04	2.60E-08
Construct.		NA	NA				
End use		--	--			--	--
End of life							
Pipe: ABS							
Resin Low	0.008	95	3.8E-04				
Resin High	0.008	1300	2.6E-02				
C2S Low	0.008	0	0	1.2	0.80	6.08E-05	1.62E-11
C2S High	0.008	1400	2.5E-02	1.2	0.80	4.42E-03	5.38E-09
Construct.		NA	NA				
End use		--	--			--	--
End of life							
Pipe: Cast iron							
C2S Low	0.008	240	6.4E-04	7.2	4.80	4.61E-03	8.19E-10
C2S High	0.008	240	7.4E-03	7.2	4.80	4.61E-03	9.47E-09
Construct.		NA	NA				
End use		--	--			--	--
End of life							
Siding: Vinyl							
Resin Low	0.008	36	7.4E-02				
Resin High	0.008	190	3.7E-01				
C2S Low	0.008	0.56	--	1.16	0.77	2.37E-05	3.05E-09
C2S Likely	0.008	1	1.9E-03	1.16	0.77	2.47E-05	3.37E-09
C2S High	0.008	12	3.8E-02	1.16	0.77	1.47E-04	2.15E-08
Construct.		NA	NA				
End use		--	--			--	--
End of life							
Siding: Aluminum							
C2S Low	0.008	3	NA	1.60	1.07	1.28E-05	--
C2S High	0.008	93	6.8E-03	1.60	1.07	3.97E-04	1.93E-09
Construct.		NA	NA				
End use		--	--			--	--
End of life							
Siding: Wood							
C2S	0.008			3.70	2.47		

Life Cycle Phase	pwhr/$	HI	Cancer (ILCR)	Retail $/f.u.	Manufact. $ per f.u.	Total HI/f.u.	ILCR/ f.u.
Construct.		NA	NA				
End use		NA	NA				
End of life							
Siding: Fiber Cement							
C2S	0.008			2.0	1.33		
Construct.		NA	NA				
End use		NA	NA				
End of life							
Window: Vinyl							
Resin Low	0.008	36	7.4E-02				
Resin High	0.008	190	3.7E-01				
C2S Low	0.008	0.38	2.5E-05	150	100.00	1.58E-03	1.98E-07
C2S Likely	0.008	1.7	9.3E-03	150	100.00	2.05E-03	4.21E-07
C2S High	0.008	11	4.6E-02	150	100.00	1.16E-02	2.09E-06
Construct.		NA	NA				
End use		--	--			--	--
End of life							
Window: Aluminum							
C2S Low	0.008	3	--	200	133.33	1.6E-03	--
C2S High	0.008	93	6.8E-03	200	133.33	4.96E-02	2.42E-07
Construct.		NA	NA				
End use		--	--				
End of life							
Window: Softwood							
C2S Low	0.008	2137	--	250	166.67		--
C2S High	0.008	6600	--	250	166.67		--
Construct.		NA	NA				
End use		NA	NA				
End of life							
Flooring: VCT							
Resin Low	0.008	90	0.075				
Resin High	0.008	240	4.1E-01				
C2S Low	0.008	9.8	--	3.20	2.13	1.66E-04	5.12E-09
C2S Likely	0.008	10	4.1E-05	3.20	2.13	1.67E-04	5.14E-09
C2S High	0.008	35	8.2E-04	3.20	2.13	5.09E-04	2.84E-08
Construct.		0.33	1.1E-06			1.38E-06	4.78E-12
End use		0.10	2.3E-08			1.04E-04	2.40E-11
End of life							
Flooring: Sheet vinyl							
Resin Low	0.008	36	7.4E-02				
Resin High	0.008	190	3.7E-01				
C2S Low	0.008	9.8	--	3.60	2.40	1.44E-04	9.47E-09
C2S Likely	0.008	10	4.1E-05	3.60	2.40	1.46E-04	9.49E-09
C2S High	0.008	35	8.2E-04	3.60	2.40	6.34E-04	4.78E-08
Construct.		0.90	--			3.75E-06	--
End use		0.27	--			2.81E-04	--
End of life							

Life Cycle Phase	pwhr/$	HI	Cancer (ILCR)	Retail $/f.u.	Manufact. $ per f.u.	Total HI/f.u.	ILCR/ f.u.
Flooring: Linoleum							
C2S	0.008	ND	ND	4	2.67	--	--
Construct. Likely		2.6	7.1E-07			1.08E-05	2.96E-12
Construct. High		3.3	3.6E-06			1.38E-05	1.50E-11
End use Likely		0.8	1.4E-08			8.33E-04	1.46E-11
End use High		1.0	7.3E-08			1.04E-03	7.6E-11
EOL							
Flooring: Cork							
C2S Low	0.008	4.8E-02	5.6E-8	4	2.67	5.12E-07	3.98E-14
C2S Likely	0.008	4.4E-01	6.2E-07	4	2.67	4.69E-06	4.41E-13
C2S High	0.008	8.4E-01	1.2E-06	4	2.67	8.96E-06	8.53E-13
Construct. Low		0.38	4.4E-07			1.2E-06	1.80E-12
Construct. Likely		3.5	4.9E-06			1.35E-05	2.02E-11
Construct. High		6.6	9.3E-06			2.57E-05	3.86E-11
End use Low		0.11	8.9E-09			1.15E-04	9.27E-12
End use Likely		1.1	9.8E-08			1.15E-03	1.02E-10
End use High		2	1.9E-07			2.08E-03	1.98E-10
End of life							

Pwhr/$ = production worker hour/$; f.u.=functional unit
NA=Not available or not assessed
If values are missing, it is because there was no risk estimated for this building material and use phase (e.g., no exposure data available or no carcinogens measured or assumed)
Shaded values were derived using OSHA PELs for at least one, but not necessarily all, of the compounds in the risk estimates.

Appendix H: Additional Analyses: Risk Assessment of Phthalate Exposure

H.1. Phthalates, Endocrine Disruption and Reproductive Toxicity

Of particular concern to interested stakeholders is the potential for endocrine disruption and reproductive effects following phthalate exposure. DEHP and dibutyl phthalate are reproductive toxicants in both sexes of rodents (CERHR-DEHP, 2005; Lovekamp-Swan and Davis, 2003). Further, the testicular effects in male rats caused by DEHP exposure can result in infertility. By contrast, fertility effects have not been observed in men exposed to phthalates in their working environment, although data are limited. Delay in time to pregnancy was not found in partners of men exposed occupationally to DEHP at concentrations as high as 2.1 mg/m^3 (equivalent to 0.6 mg/kg-day; Modigh et al., 2002). DEHP affects the Sertoli cell in rodent males; these cells are essentially "nurse" cells in the development of mature sperm. Therefore, adverse effects on Sertoli cells can result in decreased sperm number and/or the development of abnormally shaped or functioning sperm. Studies have not been identified that investigated either direct testicular effects or indirect effects on sperm quality or number in occupational workers or in those incidentally exposed to phthalates from the flooring materials analyzed in this report. Nevertheless, pregnancy is the preferred endpoint when assessing fertility, as there is a wide range in the number and quality of sperm in any given male population.

The Center for the Evaluation of Risks to Human Reproduction recently updated its 2000 report on the reproductive toxicity of DEHP (CERHR-DEHP, 2005). The expert panel who authored the report stated in the update several conclusions that were extensions and refinements of those conclusions stated in the 2000 report. They are the following:

- The panel has “minimal concern” that general population exposures adversely affect adult human reproduction. This level of concern is not appreciably altered for adults medically exposed to DEHP or MEHP.
- The panel has “concern” that DEHP exposure may adversely impact reproductive development in male children younger than 1 year of age (expressed as “concern”) and older than 1 year of age (expressed as “some concern”) based on greater susceptibility and uncertainties regarding exposure.
- The panel has “some concern” that oral ingestion of environmental concentrations of DEHP by pregnant or lactating women may adversely affect offspring development. This is downgraded from the previous level of “concern” in the 2000 report based on increased confidence in exposure levels in the population and in effect levels in animal studies.

The CERHR report also discussed recent published studies regarding DEHP reproductive toxicity. The expert panel found that there was not sufficient data available from human studies to indicate that DEHP causes reproductive toxicity in either sex, but that available studies were sufficient to conclude that DEHP causes reproductive toxicity in the following animal models: female rats (given in the diet at 1088 mg/kg-day for multiple generations), female marmosets (given via oral gavage at 500 mg/kg-day for ~15 months during the peripubertal period), in male rats (given via oral gavage or in the diet at 10-113 mg/kg-day during the gestational and/or the peripubertal period), and in adult male mice (given in the diet at 2857 mg/kg-day).

The CERHR report also evaluated developmental studies focused on DEHP exposure that had been published since the release of the draft report. The expert panel found that there was insufficient data in humans to determine that prenatal or childhood exposure to DEHP results in developmental toxicity. The expert panel found sufficient evidence to conclude that DEHP in the diet results in developmental toxicity in rats when exposure occurs either during gestation or during the early postnatal period (effects seen at 14-23 mg/kg-day) and sufficient evidence to conclude that DEHP induces adverse developmental effects following iv exposure beginning at postnatal day 3-5 and continuing for 21 days (effects seen at 300 mg/kg-day).

The majority of the available data do not indicate that DEHP and other phthalates are estrogenic. DEHP was shown not to induce the growth of estrogen-responsive breast cancer cell lines *in vitro* (Harris et al., 1997). It has been shown to decrease serum estradiol levels, lengthen estrus cycles, and prevent ovulations in adult, cycling female Sprague-Dawley rats when given orally at a high dose of 2 g/kg over 1-12 days (Lovekamp-Swan and Davis, 2003). The study authors showed that the granulosa cells of the preovulatory follicles were the target cells in the ovary, and the suppression of luteinizing hormone (LH) was the result of the decreased estradiol production. DEHP and butyl benzyl phthalate can also cause feminization of rat pups dosed in the perinatal period (gestation day 14 to postnatal day 3), but the administered doses were quite large (e.g., 750 mg/kg; Gray et al., 2000). DEHP, BBP, DINP, and five other phthalates were found not to induce a uterine response in ovariectomized immature female rats at oral doses up to 2000 mg phthalate ester/kg body weight (Zacharewski et al., 1998), although high concentrations (10 μM, compared to the 10 nM positive control compound 17 beta-estradiol) of BBP, di-n-butyl phthalate, and di-hexyl phthalate did induce weak responses (≤42% compared to 100% for the positive control) in some of the *in vitro* assays in the same study.

H.2. Phthalate Concentrations in Residential Environments

It is of interest to estimate the total daily dose of DEHP from work and home for an occupational worker. For the purpose of this exercise, it is assumed that the exposure estimate of 0.286 mg DEHP/kg-day in occupational environments (CERHR-DEHP, 2000) is correct. This daily dose is based on a 1 mg/m^3 concentration in the occupational environment (assuming an inhalation rate of 20 m^3/day and 70 kg body weight). The Air Resources Board of California has determined phthalate levels in indoor air samples taken from 125 Southern California homes (CARB, 1994). The average indoor concentration of DEHP was 140 ng/m^3 (daytime), and 100 ng/m^3 (nighttime), while the 90th percentile value for daytime was quoted as 240 ng/m^3 (Sheldon et al., 1993, as cited in CSTEE, 2004). The DEHP values in the Southern California study were comparable, but greater, to the median DEHP value (77 ng/m^3) reported in 120 Cape Cod homes by Rudel et al. (2003).

Phthalate doses were reconstructed using urinary levels of metabolites published from the most recent NHANES exposure study (Silva et al., 2004). Creatinine-adjusted levels of the following metabolites, MEP, MEHP, MBP, and MBzP, for varying age groups were used to calculate exposure estimates based on the method used by Koch and coworkers (2003). Koch's method of dose reconstruction apparently gives greater values than an alternate method due primarily to the estimation of excreted MEHP (CERHR-DEHP, 2005). This assessment provides only dose reconstruction based on the Koch methodology. Both geometric means and 95th percentile values for urinary metabolites were used to estimate potential phthalate exposures.

Table H-1: Combined Intake of Phthalates Based on Urinary Metabolites in Varying Age Groups (NHANES IV Data, 1999-2000)[a]

Urinary Metabolite Levels, µg/g creatinine					Intake Values, µg/kg-day[b]			
Age Group	MBzP	MBP	MEP	MEHP	DI (BBzP)[c]	DI (DBP)	DI (DEP)	DI (DEHP)
Geometric mean urinary metabolite levels								
6-11	40	41.9	92.6	5.19	0.74	0.84	1.70	3.78
12-19	17.3	24.3	142	2.53	0.32	0.49	2.60	1.84
>20	11.8	20.4	179	3.03	0.40	0.74	5.97	4.01
20-39	12.5	20	178	3.3	0.42	0.73	5.93	4.37
>40	11.2	20.7	180	2.84	0.38	0.75	6.00	3.76
95th percentile urinary metabolite levels								
6-11	142	159	625	41.9	2.62	3.19	11.46	30.52
12-19	69.3	88.1	1550	12.1	1.28	1.77	28.42	8.81
>20	57.2	91	2170	17.5	1.92	3.32	72.35	23.18
20-39	61.1	81.4	2661	20.9	2.05	2.97	88.72	27.68
>40	56.8	97.7	2064	12.9	1.90	3.56	68.82	17.09
RfD (mg/kg-day)[d]					0.2	0.1	0.8	0.02
NOAEL[d]					159	125	750	19
MOE[e]					214,865-60,687	148,810-35,112	125,000-8,453	4,347-622

a NHANES data taken from Silva et al., 2004
b Intake values modeled using parameters and equations from Koch et al., 2003, including the following: Fue (adults) = 20 mg/kg-day and DI (µg/kg-day) = UE (µg/g) * CE (mg/kg-day)/Fue *1000 mg/kg * MW diester/MW monoester.
c Daily intake
d RfD and NOAEL values obtained from U.S. EPA IRIS website (www.usepa.gov/iris)
e Margin of Exposure, a ratio of the exposure value and the NOAEL; the values listed above are calculated based on the highest value listed for each metabolite level (geometric mean or 95th percentile)
UE, urinary excretion in µg/g; CE, creatinine excretion per day, mg/kg-day); Fue, molar conversion factor; MW, molecular weight, g/mole

The data above indicate that intake estimates for children range from the lowest geometric mean of 0.32 µg/kg-day for butyl benzyl phthalate to the highest (95th percentile) value of 30.52 µg/kg-day for DEHP. For adults, the lowest geometric mean exposure estimate was 0.38 µg/kg-day for butyl benzyl phthalate and the highest (95th percentile) value was 88.72 µg /kg-day for diethyl phthalate. The lowest exposures for all age groups were consistently those to butyl benzyl phthalate, which is reportedly predominantly used in vinyl flooring, while the highest exposures

were to DEHP in children, and DEP in adults. DEP is primarily used in consumer products such as perfumes, toiletries, and other personal care products; differential use of these products among the age groups explains the differential exposures. The values above were comparable to the ranges estimated by the CERHR expert panel in their own dose reconstruction efforts (CERHR-DEHP, 2005). It is interesting to note that the exposure estimates for DEHP in the table above are much higher than those predicted based on air concentrations of DEHP measured in homes in California or Cape Cod, indicating that other exposure pathways predominate.

The last row of the table above provides the Reference Doses, for the four diester phthalates discussed. For all but DEHP, the estimated intake amounts are well below the RfD values, even when the 95th percentile exposures are considered. The same is not true, however, for DEHP, in which the 95th percentile values are comparable to the RfD, or in the case of 6-11 year old children, the intake estimate actually exceeds the RfD. Because RfD values are only intended to represent a safe exposure limit to within an order of magnitude, many risk assessors and policy makers will discuss safety of exposures in terms of "Margin of Exposure." This value is the ratio of the no observable adverse effect level, or NOAEL (in the most sensitive animal model chosen for the compound), and the estimated exposure level in humans. The NOAELs (taken from the IRIS toxicity profiles) and calculated MOE values for the four phthalates are provided in the last two rows of the table above. Ideally, for non-cancer effects, risk assessors would like to see a MOE value of 100 or greater. As shown, the MOE values are >600. These high MOE values would typically suggest that adverse effects are not expected in individuals exposed at the levels estimated above. However, it must be pointed out that the NOAELs selected for these phthalates are based on older, traditional subchronic toxicity studies that do not capture the dose range used in reproductive or developmental toxicity tests. Recent rodent studies using gestational exposures indicate that the doses at which no adverse effects are noted are lower than the NOAELs above (Zhang et al., 2004; Barlow et al., 2004).

Risk values were estimated using concentrations of phthalates in air and dust obtained in the Silent Spring Household Exposure Study (Rudel et al., 2003). The analyses evaluated potential risks from both the median and maximum air and dust DEHP values reported. The maximum value of DEHP in dust is greater than that determined by Øie et al. (1997) in a sample of 38 homes in Oslo, Norway. In those homes, the mean DEHP concentration was 640 µg/g for total dust (range, 100 to 1610 µg/g) and 820 µg/g (range, 110-2100 µg/g) for organic fraction. Because DEHP values were the largest for all phthalates measured in the houses and because DEHP toxicity is greater than other phthalates, risk estimates were only calculated for this compound. Concentrations of DEHP in air and dust were used to estimate daily doses of the compound, then non-cancer HI values were calculated based on children, teens, and adults inhaling DEHP in air and incidentally ingesting the dust containing the compound. The risk estimates are provided in the table below.

Table H-2: Non-Cancer Risk Estimates for Children, Teen and Adults Exposed to DEHP in Air and Dust from Cape Cod Homes

DEHP	Concentration	Child Hazard Index	Teen Hazard Index	Adult Hazard Index
Air Exposures				
Maximum	1E-03 mg/m3	0.038	0.022	0.0099
Median	7.7E-05 mg/m3	0.003	0.0017	0.00076

Dust Exposures				
Maximum	7700 µg/g dust	3.3	0.077	0.003
Median	340 µg/g dust	0.15	0.0034	0.00013

Exposure Parameters: children, 0-7, teens 8-17, adults, 18+; ingestion rate for dust: children, 135 mg/day; teens, 10 mg/day; adults, 0.56 mg/day. Inhalation rates: child, 0.6 m3/hr; teen, 1.4 m3/hr; adult, 0.9 m3/hr. Body weights: child, 15 kg; teen, 48 kg; adult, 70 kg. Duration in home (hrs): child, 20; teen and adult, 16. Exposure assumed 7 days/week, 50 weeks per year, for a 30-year period. Values from U.S. EPA's Exposure Factors Handbook or using professional judgment.

The above risk estimates indicate that inhalation of DEHP in air does not pose what federal and state environmental protection agencies consider to be an unacceptable risk to any age group modeled, as all non-cancer Hazard Index (HI) values for air exposure were well below 1, even when the maximum DEHP air concentration was considered. All risk estimates from exposure to dust were below 1 also, irrespective of median or maximum DEHP dust concentrations used, except for child risk estimate of 3.3, using the maximum DEHP dust concentration. Adding the HI values from dust and air exposure does not significantly increase the overall risk, as the air values are either comparable or at least one order of magnitude smaller than those for dust exposure. It is noteworthy that the average daily dose for DEHP for the child potentially exposed to the maximum DEHP dust concentration was 0.00077 mg/kg-day, which is two orders of magnitude lower than the intake estimates provided above for the 6-11 age group, both for geometric mean urinary levels and 95th percentile levels. If the intake estimates are accurate, and dust concentrations in Cape Cod homes are representative of those nationwide, then phthalate exposures from other sources are likely to be higher than those from household dust and air. The reader is reminded that these risk estimates are derived using the existing RfD for DEHP, which is not based on reproductive or developmental effects in animal models. A revised RfD for DEHP will most likely reflect the large body of data on these effects in rodents and primates. It is not known if the RfD will remain within the same order of magnitude or may be decreased. Revised estimates of human health risk using an updated RfD may in fact be larger, but are likely to be within one order of magnitude (given the NOAEL and LOAEL values of recent reproductive studies in rodents). Also, these risk estimates are much greater than those expected for exposure to butyl benzyl phthalate or di-n-butyl phthalate in dust. That is because the median and maximum dust concentrations for those phthalates determined in the Rudel study were lower (up to one order of magnitude) than those of DEHP and the RfD values for those phthalates are at least 5 times greater than that for DEHP.

H.3. Phthalates and Asthma

A few studies were identified that focused on the potential role of phthalates in the home and the onset of asthma and other respiratory diseases in children.

Jaakkola et al. (1999) published a link between PVC floors and textile wallcoverings and the incidence of bronchial obstruction in Norwegian children followed from birth to age 2. The adjusted odds ratios (ORs) for bronchial obstruction were 1.89 for PVC flooring and 1.58 for textile wallpaper. PVC-coated wallpaper was not associated with an increased OR. Although the study authors claimed that increased exposure to plasticizers was also associated with the incidence of bronchial obstruction, this analysis was based on a categorization of potential plasticizer-emitting materials in the home, rather than on actual exposure concentrations of different phthalates. Further, the study authors stated that there was a higher percentage of atopic

parents in the group of children with bronchial obstruction. Atopy is an inherited allergic response associated with elevated immunoglobulin E (IgE), which is associated with bronchial asthma and eczema. (It is noted that atopic parents tend to choose flooring surfaces, such as smooth, easy-to-clean materials [such as vinyl], that help control allergies and asthma.) It is not clear that these children were removed from the case group, or that the atopy was controlled for as a confounding factor. Further, atopy was not studied in the case group. Therefore, it is unclear what percentage of the children with bronchial obstruction may have been predisposed toward the condition due to heredity.

Bornehag and coauthors (2004) investigated the association of phthalate esters in dust in homes of Swedish children with asthma, rhinitis, and/or eczema. The study included 175 children with an allergic condition and 177 controls. The study authors report that BBP in the dust of children's rooms was significantly associated with the incidence of rhinitis and eczema, when both the median and geometric mean dust concentrations associated with the rooms of symptomatic children (diagnosed by a doctor) were compared to those values from the rooms of asymptomatic control children. By contrast, DEHP in the dust of children's rooms was associated with asthma, but not rhinitis or eczema. The study authors speculated that the amount of each phthalate present in the gas phase rather than the particulate phase might be affecting the difference in symptoms. For example, they hypothesized that BBP, which they believed should be present in the gas phase of the bedrooms due to its higher volatility, would cause skin and mucosa symptoms and the DEHP, adsorbed to the dust particles, would cause lower airway symptoms. While the vapor pressure of BBP (8.6x10-6 mm Hg at 25C) is 100 times that of DEHP (7.3x10-8 mm Hg at 25C; both values from HSDB, 2005), the low volatility of both compounds indicates that neither is likely to be present in the gas phase at significant concentrations. Limited studies indicate that indoor concentrations of BBP in California homes were 0.035 ng/m^3 (CERHR-BBP, 2000).

The above studies suggest a link between asthma and phthalate-containing materials in the home. Neither, however, discusses exposure of the children to other known risk factors for asthma; further, these other risk factors were not apparently controlled for in these studies. Therefore, these studies do not permit one to quantify the contribution, if any, of the phthalates to the development of asthma in the children studied.

Larsen et al. (2004) showed that a 60-minute exposure of mice to MEHP at concentrations as high as 43.6 mg/m^3 did not result in upper airway irritation, although lower airway irritation (measured by shallow breathing) was observed at concentrations higher than 0.3 mg/m^3. A level as high as 1.7 mg/m^3 did not induce macrophages in the lungs (as measured by immune cells in bronchial alveolar lavage [BAL] fluid) and neutrophils, lymphocytes, eosinophils and epithelial cells were not observed in BAL fluid at concentrations as high as 30 mg/m^3. Based on their analysis of potential MEHP from inhaled DEHP in dust from vinyl flooring, the authors calculated a 200-fold difference between the worst-case DEHP exposures of children from floor dust in the Clausen et al. (2003) study of Danish schools and the NOEL (300 µg/m^3). In light of this safety factor, the authors stated that no airway irritation was expected from MEHP (originating from DEHP exposures) at non-occupational indoor air levels.

Butala et al. (2004) performed a dermal sensitization analysis of several phthalates to determine the ability of each phthalate to induce an allergic response in the mouse model. DEHP, BBP, and other phthalate esters were tested for their ability to induce IgE levels in serum. The phthalates did not significantly induce IgE levels, or the levels of cytokines IL-4 and IL-13, which are

known to stimulate IgE production as part of the sensitization response. Increased liver weight in the dosed mice was observed, which indicated that adequate doses of the phthalates were absorbed through the skin to induce a physiologic response. Trimellitic anhydride, a lung sensitizer, was used as a positive control, and produced a strong induction in IgE, and the levels of the measured cytokines. The study authors indicated these data suggested that DEHP and BBP are unlikely to induce antibody-mediated respiratory allergies.

Recently it has been shown that DEHP, and certain monophthalates, have an adjuvant effect—they do not directly induce, but rather potentiate, or contribute, to the allergic effect induced by other agents (Larsen et al., 2001; Jepsen et al., 2004; Glue et al., 2005). Larsen et al. (2002) evaluated the ability of four phthalates (DBP, DINP, DOP, and DIDP [di-iso-decyl phthalate], when injected subcutaneously in mice, to induce an adjuvant effect when administered following injection of ovalbumin. Phthalates with 8 or 9 carbon atoms as side chains were effective, while shorter or longer alkyl side chains were less so. The relevance of these data to inhalation exposures in man is unknown. Butyl benzyl phthalate (BBP), although not tested by Glue and coworkers, has not been shown to have this adjuvant effect in a murine model (Larsen et al., 2003).

Additional studies regarding asthma and other respiratory diseases in children and adults are needed to investigate the potential relationship between phthalates and asthma. Current data in children suggest a link between phthalate exposure and allergic reactions—but phthalate exposure may be contributing to the overall problem that is induced by some other allergenic agent.

Appendix I: Air Monitoring Data and Fenceline Analysis

The focus of this assessment has been on exposures of occupational workers and those of the general public during the life cycle of PVC and non-PVC building products, from cradle to grave. This section investigates the potential human health risks to residents in neighborhoods adjoining the facilities who manufacture PVC resin or the final building materials. These facilities are allowed by law to emit certain finite quantities of compounds from smokestacks, scrubbers, or in the waste stream. These emissions result in a consistent, but low, concentration of chemicals in the nearby environment. In most situations, these environmental concentrations are unlikely to exceed acceptable regulatory limits. However, critics of PVC manufacture point to "upset conditions" at chemical facilities in Louisiana and Kentucky and the measurement of air concentrations of VCM and EDC that exceed state ambient air quality standards (Subra, 2004).

I.1. Risks based on Air Monitoring in Louisville, Kentucky 1999-2001

In response to growing community concerns about ambient air pollution from Rubbertown chemical facilities in the Louisville, Kentucky area, the Air Pollution Control District of Jefferson County, (now identified as the Metro Louisville Air Pollution Control District, MLAPD), in conjunction with the U.S. EPA, the commonwealth of Kentucky, and the West Jefferson County Community Task Force (WJCCTF) conducted an air monitoring study. Sampling began in 1999; based on the results, additional sampling monitors were installed and sampling was continued in 2000-2001. Air samples were taken from 15 monitors at 12 different sites (one location had two sets of two monitors) from April 2000 to April 2001. The monitoring sites were selected to represent unique locations in residential areas at which it could be expected that a resident may be exposed to airborne contaminants. (WLATS, 2003). Sciences International (Sciences) was hired to perform a risk assessment using the air data to determine if unacceptable human health risks were caused by the levels of environmental toxicants measured by the study (WLATS, 2003). Sciences identified as Chemicals of Potential Concern (COPCs) those compounds that were detected in at least 10% of the samples obtained at <u>each</u> monitor. The monitoring sites are listed in the table below. The task force was primarily concerned about volatile organic compounds; therefore, all monitors analyzed air samples for VOCs. A subset also including monitoring for semi-volatile organic chemicals (SVOCs), metals, pesticides, and PCBs (the risk assessment indicates, however, that because pesticides and PCBs were only sampled on one single day, they were not included in the chronic risk estimations because the single sample may not have represented an actual chronic exposure at each location).

Table I-1: Air Monitoring Stations around Louisville, KY

Monitor Number	Monitoring Station	Location	Type of Station	Monitoring Dates	Monitored Compounds
1	Louisville Firearms Training Center	Algonquin Parkway	Potential max. impact site	1999, 2000, continuing	VOCs, form, metals, SO_2, SVOCs, AR aerosols#
7	Southwick Community Center*	Southern Avenue	General neighborhood exposure site	1999, 2000	VOCs, $PM_{2.5}$, PM_{10}
2	Ralph Avenue/Campground Rd.^	Campground Road	Potential max. impact site	1999, 2000, continuing	VOCs, form, metals, SVOCs, AR aerosols
3	Old Lake Dreamland Fire Dept.*	Campground Road	General neighborhood exposure site; pot. max. impact site	2000	VOCs, form, metals, SVOCs, AR aerosols
6	Otter Creek Park*	Otter Creek Parkway	Background site	1999, 2000	VOCs, form, metals, SVOCs, AR aerosols
5	Univ. of Louisville Shelby Campus	Shelbyville Road	Urban activity control site	1999, 2000, continuing	VOCs, form, metals, SVOCs, AR aerosols
8	Farnsley Middle School	Lees Lane	General neighborhood exposure site	2000, continuing	VOCs
10	New Lake Dreamland Fire Dept.	Cane Run Road	General neighborhood exposure site	2000, continuing	VOCs
9	Chickasaw Park	Algonquin Parkway	General neighborhood exposure site	2000, continuing	VOCs
11	King Elementary*	Vermont Avenue	General neighborhood exposure site	2000	VOCs
4	St. Stephens Baptist Church*	South 15th Street	General neighborhood exposure site	2000	VOCs, form, metals, SVOCs, AR aerosols
12	Cane Run Elementary	Cane Run Road	General neighborhood exposure site	2000, continuing	VOCs

*Sites are no longer monitored. ^ This monitoring site had two sets of stations, A & B.

Codes: VOCs, volatile organic compounds; form, formaldehyde; SVOCs, semi-volatile organic compounds, SO_2, sulfur dioxide, AR aerosols, acid-reactive aerosols, $PM_{2.5}$ and PM_{10}, particulate matter with diameters of 2.5 microns and 10 microns.

The locations of most of the monitoring stations and a wind rose for the region for the time period during which monitoring occurred (taken from WLATS, 2003) are shown in Figure I-2 and Figure I-1, respectively. VOCs which were present in a majority of the monitors at a significantly high rate included several Freons (Freon 113, 22, 12, 11), benzene, toluene, MTBE, carbon tetrachloride, and hexane.[12] Of the primary compounds associated with vinyl chloride manufacture, vinyl chloride monomer was included as a COPC, but only for one monitoring station (Southwick Community Center, which is no longer monitored); it was detected at 8 of the 13 monitoring stations (Louisville Firearms Training Center; Ralph Avenue/Campground Rd A&B; St. Stephens Church; Southwick Community Center; Farnsley Middle School; New Lake Dreamland; and Cane Run Elementary School). However, it was only present at Southwick Community Center in at least 10% of all samples (35.7%); in all other stations, it was detected <10% of the time. (The limit of detection for compounds was 0.1 ppb_v.) The following compounds, also associated with vinyl manufacture, were also not included as COPCs due to their not being detected in at least 10% of the samples in at least one monitoring station during the sampling period: 1,2-dichloroethane, chloroethane, hexachlorobutadiene, di-n-butyl phthalate, and diethyl phthalate. DEHP was included as a COPC for three sites (Ralph Avenue/Campground Monitor A, Old Lake Dreamland, and St. Stephens Church). Interestingly, DEHP was not present at the same location at which vinyl chloride monomer was detected at the highest concentration (Southwick), suggesting that the compounds may be originating from different locations.

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[12] It is noted that Freons such as Freon 113 (also known as CFC-113) are not only human toxicants via direct exposure pathways such as inhalation, but are also chemicals which deplete the stratospheric ozone layer, allowing more ultraviolet light to reach the earth. This thinning of the stratospheric ozone layer increases the risk of cataracts and skin cancer, among other possible human health effects. Because we lack data on the annual quantity of emissions of CFCs from these facilities, and cannot readily estimate the emissions based on this ambient concentration modeling, we are not able to estimate the incremental (global) health risks due to their incremental contribution to the thinning of the ozone layer. However, these impacts might be, on a per-functional unit basis, within the range of the other risks being addressed in this study. This is a topic which deserves follow-on study, starting with an attempt to use updated monitoring to estimate annual emissions quantities (per unit of product output from the facilities). The pathways from emissions to estimated global health risks (in DALYs) have already been modeled and are available for our use, for example within the EcoIndicator 99 method for life cycle impact assessment.

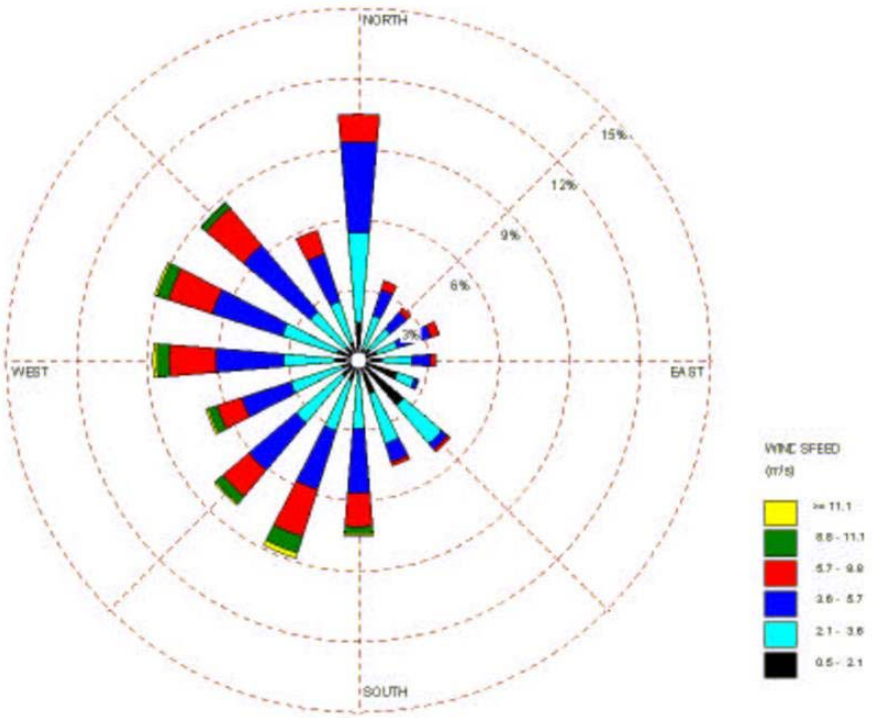

Figure I-1: Wind Rose for Louisville, KY

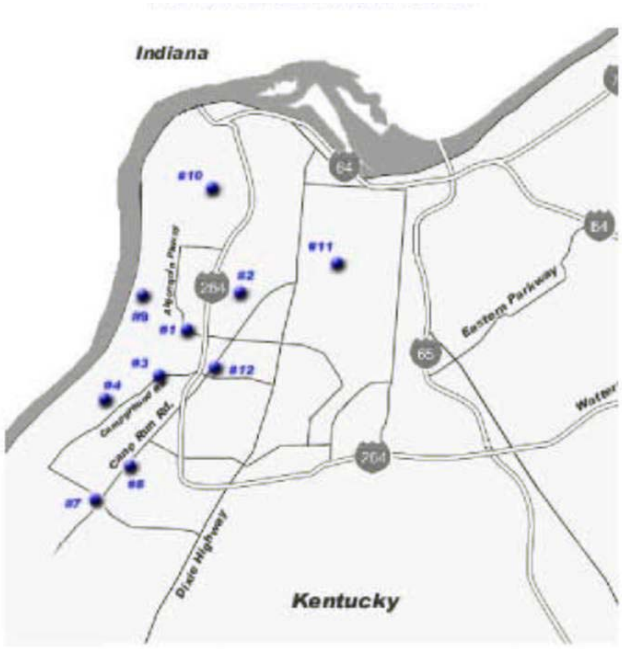

Figure I-2: Locations of Air Monitoring Stations around Louisville, KY

The highest and lowest risk estimates, considering VOCs only, for both non-cancer and cancer risk are presented in the table below, along with the risk estimates for the vinyl-associated compounds and the monitoring stations that comprised these risk estimates. The largest risk estimates were generated when exposure to metals (arsenic, cadmium, chromium, and nickel) were considered; metals analyses were only performed at stations 1-6 (refer to table above). Chromium and nickel were found in all six of the monitored stations in 100% of the samples obtained; arsenic was found in five monitoring stations 100% of the time, and in the last station 92.3% of the time. Cadmium was found in all 6 monitoring stations and in over 38.5% of the samples. Due to the limited nature of metals monitoring, the risks from these compounds will not be considered in this report. Rather, the risk estimates arising from VOCs will be considered, because they are of primary importance in an analysis of risks from exposure to PVC-related compounds and those of the alternatives. (Although mercury is also an important PVC-related metal, it was not monitored by the sampling program at the Louisville monitors).

Table I-2: Median Cancer Risk Estimates from Inhalation of Air around Louisville Kentucky (ca 2000-2001)[a]

Compound[b]	URE[c] $(mg/m^3)^{-1}$	Ralph Avenue (Highest)		King Elementary (Lowest)		Southwick Center	
		Cancer Risk	% of Total	Cancer Risk	% of Total	Cancer Risk	% of Total
Carbon tet[d]	1.5E-02	9.5E-06	5.2	1.1E-05	30.1	7.6E-06	12.7
acrylonitrile	6.8E-02	NA	--	7.5E-06	19.7	7.5E-06	12.4
1,3-butadiene	3.0E-02	5.7E-05	31.1	3.3E-06	8.7	1.8E-05	30.1
Chromium	1.2E+01	4.7E-05	25.5	NA	--	NA	--
Benzene	7.8E-03	9.4E-06	5.1	6.0E-06	15.9	1.1E-05	19.0
Vinyl chloride	8.8E-03	NA	--	NA	--	1.1E-06	1.9
Total Risk[e]		1.8E-04	100	3.8E-05	100	6.0E-05	100

a, Data taken from WLATS, 2003
b, Not all compounds detected and contributing to the site overall risk estimates are presented.
c, URE, Unit Risk Estimate (WLATS, 2003); all values obtained from IRIS Database
d, Carbon tetrachloride, CCl4
e Total risk estimate is for the monitoring station and also includes the risk estimates for the compounds listed above and for compounds not presented in the table.
NA, not applicable to the station (not a COPC for this station)

When *median* concentrations at each station were used to determine chronic cancer risks, the highest risk estimate was obtained at Ralph Avenue/Camp Road Station (Station 2b) and the lowest was obtained at King Elementary. Risk estimates at Southwick Center, the only one for which vinyl chloride was selected as a COPC, totaled 6.0E-05. Vinyl chloride contributed 1.9% of the total risk, with an estimate of 1.1E-06, just higher than Kentucky's cancer risk limit of 1E-

06 (one in one million). Median concentrations of airborne contaminants were not presented in the report available to the TG; the TG did not attempt to reconstruct the median concentrations using data from the WJCCTF database.

Sciences also estimated cancer risk using concentrations that represented the 95% upper concentration limit on the mean (shown in the table below). The risk estimates increased dramatically. For example, the total risk estimate at Ralph Avenue was 6.9E-04, with 1,3-butadiene contributing 72% of the total (5E-04). The lowest risk estimate obtained was at the Otter Creek station (background site) and was 7.6E-05, with chromium (45.6%), formaldehyde (15.2%), and carbon tetrachloride (14.9%) contributing the greatest amount to the risk. The 95% UCL total risk estimate for Southwick Center was 3.9E-04, with 1,3-butadiene contributing 47.9% of the risk estimate, and vinyl chloride representing 1.2% of the risk (WLATS, 2003). Differences between the lowest and highest risk estimates spanned one order of magnitude.

Unlike with the median concentrations, UCL risk estimates include a significant contribution from formaldehyde. This compound is a degradation product of airborne vinyl chloride monomer, via photooxidation. However, formaldehyde is also used extensively in the manufacture of rubber and other products. Therefore, it is not possible to attribute estimated cancer risk from formaldehyde exposure to the manufacture of vinyl products in the Louisville area. It is notable that when UCL air concentrations were used to estimate risks, that risk estimates at background monitoring sites (which are intended to indicate potential exposure at areas not believed to be impacted by the local manufacturing facilities) *exceeded acceptable risk levels common to federal and Kentucky risk guidelines.*

Table I-3: 95% UCL Cancer Risk Estimates from Inhalation of Air around Louisville Kentucky (ca 2000-2001)[a]

Compound[b]	URE[c] $(mg/m^3)^{-1}$	Ralph Avenue (Highest)		Otter Creek (Lowest)		Southwick Center	
		Cancer Risk	% of Total	Cancer Risk	% of Total	Cancer Risk	% of Total
Carbon Tet[d]	1.5E-02	1.1E-05	1.6	1.2E-05	14.9	1.2E-05	3.0
formaldehyde	1.3E-02	2.8E-05	4.1	1.1E-05	15.2	NA	--
acrylonitrile	6.8E-02	NA	--	NA	--	6.6E-05	16.9
1,3-butadiene	3.0E-02	5E-04	72.0	NA	--	1.9E-04	47.9
Chromium	1.2E+01	5.2E-05	7.5	NA	--	NA	--
Benzene	7.8E-03	1.2E-05	1.7	6.8E-06	9.0	3.2E-05	8.3
Chloroform	2.3E-02	4.5E-05	6.6	NA	--	2.0E-05	5.0
Methylene chloride	4.7E-04	NA	--	NA	--	1.7E-05	4.5
Vinyl chloride	8.8E-03	NA	--	NA	--	4.6E-06	1.2
Total Risk[d]		6.9E-04	100	7.6E-05	100	3.9E-04	100

a, Data taken from WLATS, 2003
b, Not all compounds detected and contributing to the site overall risk estimates are presented.
c, URE, Unit Risk Estimate (WLATS, 2003); all values obtained from IRIS Database
d, Carbon tetrachloride, CCl4
e Total risk estimate is for the monitoring station and also includes the risk estimates for the compounds listed above and for compounds not presented in the table.
NA, not applicable to the station (not a COPC for this station)

Table I-4: Median Non-Cancer Risk Estimates from Inhalation of Air around Louisville Kentucky (ca 2000-2001)[a]

Compound[b]	RfC[c] (mg/m^3)	Ralph Avenue (Highest)		King Elementary (Lowest)		Southwick Center[d]	
		Hazard Quotient	% of Total	Hazard Quotient	% of Total	Hazard Quotient	% of Total
Formaldehyde	9.8E-03	0.17	9.7	NA	--	NA	--
Acrylonitrile	2E-03	NA	--	NA	--	NA	--
1,3-butadiene	2E-03	0.95	54.8	NA	--	0.3	62.6
manganese	5E-05	0.3	17.3	NA	--	NA	--
All Others		0.32	18.2	0.19	100	0.18	37.4
Total Risk (HI)[e]		1.7	100	0.19	100	0.48	100

a, Data taken from WLATS, 2003
b, Not all compounds detected and contributing to the site overall risk estimates are presented.
c, RfC, Reference Concentration (WLATS, 2003); all values obtained from IRIS Database except for formaldehyde (ATSDR)
d, Samples taken at this location were analyzed for VOCs only.
e Total risk estimate is for the monitoring station and also includes the risk estimates for the compounds listed above and for compounds not presented in the table.
NA, not applicable to the station (not a COPC for this station)

As shown in the table above, the median non-cancer Hazard Indices (HIs) exceeded a value of 1.0 at two monitoring stations (both monitoring stations at Ralph Avenue had concentrations of compounds that presented HIs >1.0), with the remaining sites having HI values of less than 1 (ranging from a high of 0.93 at Louisville Fire Training Center to a low of 0.19 at King Elementary [shown above]). The majority of the risk was due to measured concentrations of 1,3-butadiene at 6 locations, presumably from the rubber manufacturers in the area (American Synthetic Rubber Corporation and others) (WLATS, 2003). As with the cancer risk, the total non-cancer risk, or Hazard Index (HI) was estimated as the sum of individual Hazard Quotients for each chemical. This approach is recommended and appropriate for assessing total human health risk caused by compounds that may have different exposure pathways, or may affect various target organs via differing mechanisms of action (US EPA, 1989). The total non-cancer risk estimate at Ralph Avenue station was 1.7, and 1,3-butadiene contributed 54.8% of the total (0.95). By contrast, the total non-cancer risk estimate at Southwick was 0.48 and 1,3-butadiene contributed 62.6% of the value (0.3). The contribution of vinyl chloride to the non-cancer risk was not explicitly tabulated in the report.

Acute risks were evaluated in the Sciences report by comparing the maximum concentration of each COPC to an acute Reference Concentration obtained from various sources (ATSDR or CalEPA were cited most frequently). These acute RfCs ranged from 0.22 mg/m3 for acrylonitrile (ATSDR) to 1.3 mg/m3 for vinyl chloride monomer. The resulting Hazard Quotients did not exceed 1.0 for any compound indicating that there were no acute non-cancer health risks posed at these monitoring stations during the sampling period (WLATS, 2003).

I.2. Risks Based on Air Monitoring in Louisville, Kentucky 2003-2005

Data from air sampling rounds taken from January 2003 to November 2005 were used to estimate current potential non-cancer and cancer risks from exposure to vinyl chloride monomer (WJCCTF, 2005). Data were from five monitoring stations that have been maintained in the West Jefferson County: Louisville Firearms Training Center; Ralph Avenue/Campground Rd; Farnsley Middle School; Cane Run Elementary School, and Chickasaw Park. Mean concentrations of VCM and the resultant non-cancer (Hazard Index) and cancer (Integrated Lifetime Cancer Risk) risk estimates are shown in the table below. Both median and mean concentrations are shown; risk estimates are based on the mean concentrations because they were the larger of the two. (Any samples that did not reach the quantitation limit of 0.1 ppb_v were considered to be one half the limit for the purpose of the calculation.)

According to TRI data, two vinyl chloride producing facilities are located in close proximity to each other on Bells Lane, West Louisville, KY. These are the OxyVinyls, LLP and Noveon, Inc. facilities. The Cane Run Elementary School site is located approximately 0.75 miles from these plants, while the LPFTC and Chickasaw Park stations are located 0.5 and 1.5 miles north of the facilities, respectively. The Ralph Avenue and Farnsley Middle School stations are located 0.75 and 1.5 miles south of the vinyl chloride facilities, respectively (Subra, 2004).

Table I-5: Risk Estimates for Exposure to Airborne Vinyl Chloride Monomer, 2003-2005

	Louisville Firearms	Ralph Ave.	Farnsley MS	Chickasaw Park	Cane Run Elem. S.	Risk Limits Kentucky State
VCM (median), ppb_v	0.05					
VCM (mean), ppb_v	0.25	0.06	0.06	0.10	0.17	
VCM (mean), mg/m3	0.0006	1.7E-4	1.6E-04	2.6E-04	4.3E-04	
Hazard Index[a]	6.29E-03	1.64E-03	1.57E-03	2.61E-03	4.35E-03	1.0
ILCR[b]	5.53E-06	1.44E-06	1.38E-06	2.29E-06	3.82E-06	1.0E-06

[a] HI obtained by dividing the mean concentration of VCM (mg/m3) by the Reference Concentration of 0.1 mg/m^3 (U.S. EPA IRIS)

b ILCR obtained by multiplying the mean concentration of VCM by the Unit Risk Estimate of 8E-03 $(mg/m^3)^{-1}$

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The data above indicate that mean concentrations measured over the sampling period (January 2003-November 2005) do not result in a non-cancer hazard index that exceeds 1.0 (state and federal limit); in fact, the HI values are all well below 1.0, indicating that the concentrations of vinyl chloride measured are unlikely to pose a risk of non-cancer effects in individuals who are consistently present at the site (assumed 24 hours per day, 365 days per year). However, all concentrations result in cancer risk estimates that exceed the limit of 1.0E-06; the mean concentration at the Louisville Firearms and Police Training Center, in fact, resulted in an estimate of an excess 5 individuals in 1 million with cancer from vinyl chloride exposure. These

values, however, do not greatly exceed the cancer risk limit. It is not clear whether a tightening of current regulations on emissions would reduce the risk estimates below acceptable limits.

Acute risks were evaluated by comparing the maximum value for VCM noted at the monitoring stations over the 2003-2005 sampling period. None of the concentrations exceeded an acute RfC of 1.3 mg/m3 (as discussed above).

I.3. Risks Based on Air Monitoring in Louisiana

Air monitoring data from the Louisiana Ambient Air Monitoring Program for Vinyl Chloride, tabulated by year from 1999 to 2003 by Sage Environmental Consulting, Inc. were provided by the Vinyl Institute (Sage, 2004). The report includes air monitoring data from the following monitoring stations in Louisiana: Baker (East Baton Rouge Parish); Bayou Plaquemine (Iberville Parish); Capitol (East Baton Rouge Parish); Dutchtown (Ascension Rouge Parish); Hahnville (St. Charles Parish); Lighthouse (Calcasieu Parish); LSU (East Baton Rouge Parish); Marrero (Jefferson Parish); Monroe (Ouachita Parish); Pride (East Baton Rouge Parish); Shreveport (Caddo Parish); South Scotlandville (East Baton Rouge Parish); Southern (East Baton Rouge Parish); and Westlake (Calcasieu Parish). With the exception of Hahnville, Marrero, Monroe, and Shreveport stations, all were within approximately 15 miles of one or more vinyl manufacturing facilities. The Westlake monitors (Lighthouse and Westlake) are within 1 to 3 miles of vinyl facilities (Sage, 2004). Wind roses and maps giving the locations of the monitoring stations are located in Figure I-3, Figure I-4, Figure I-5, Figure I-6 and Figure I-7 (Sage, 2004).

Figure I-3: Wind Rose for Baton Rouge, LA Area

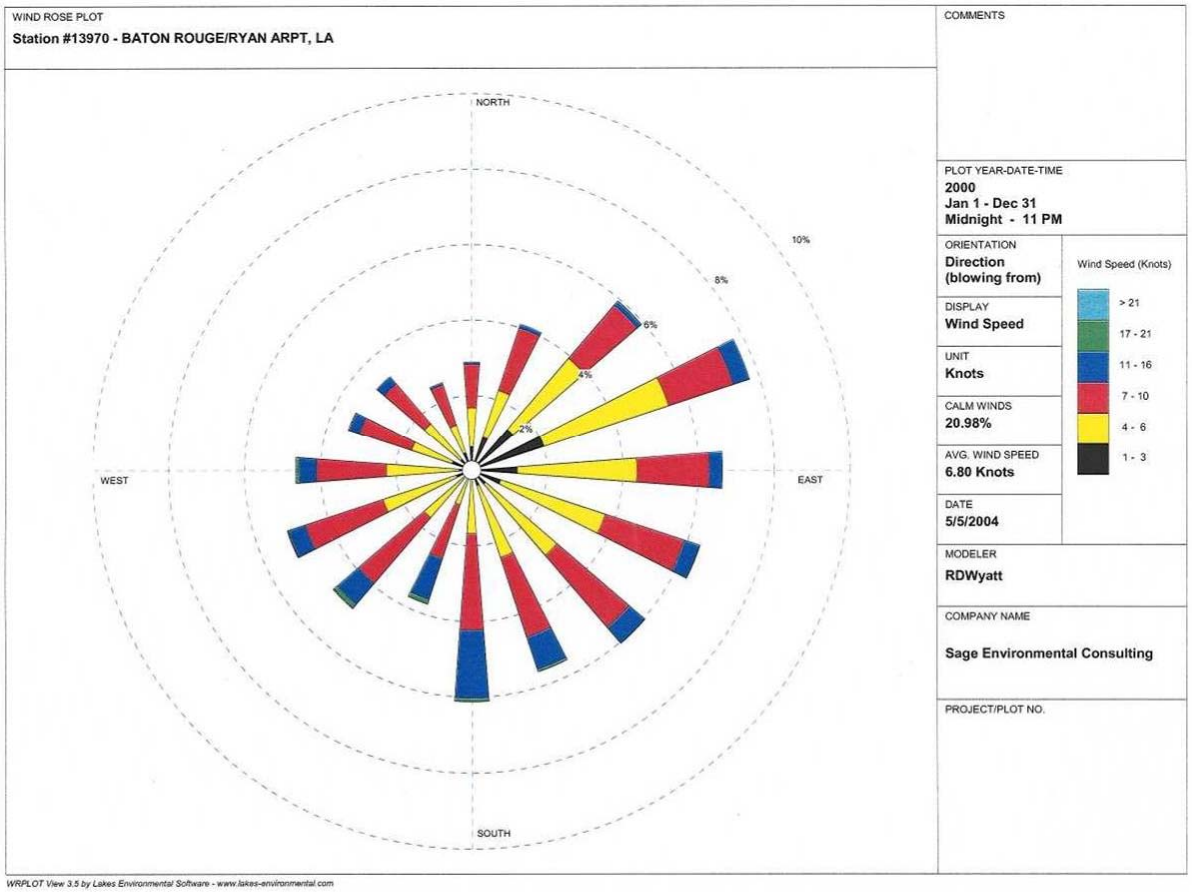


Figure I-4: Wind Rose for Lake Charles, LA Area

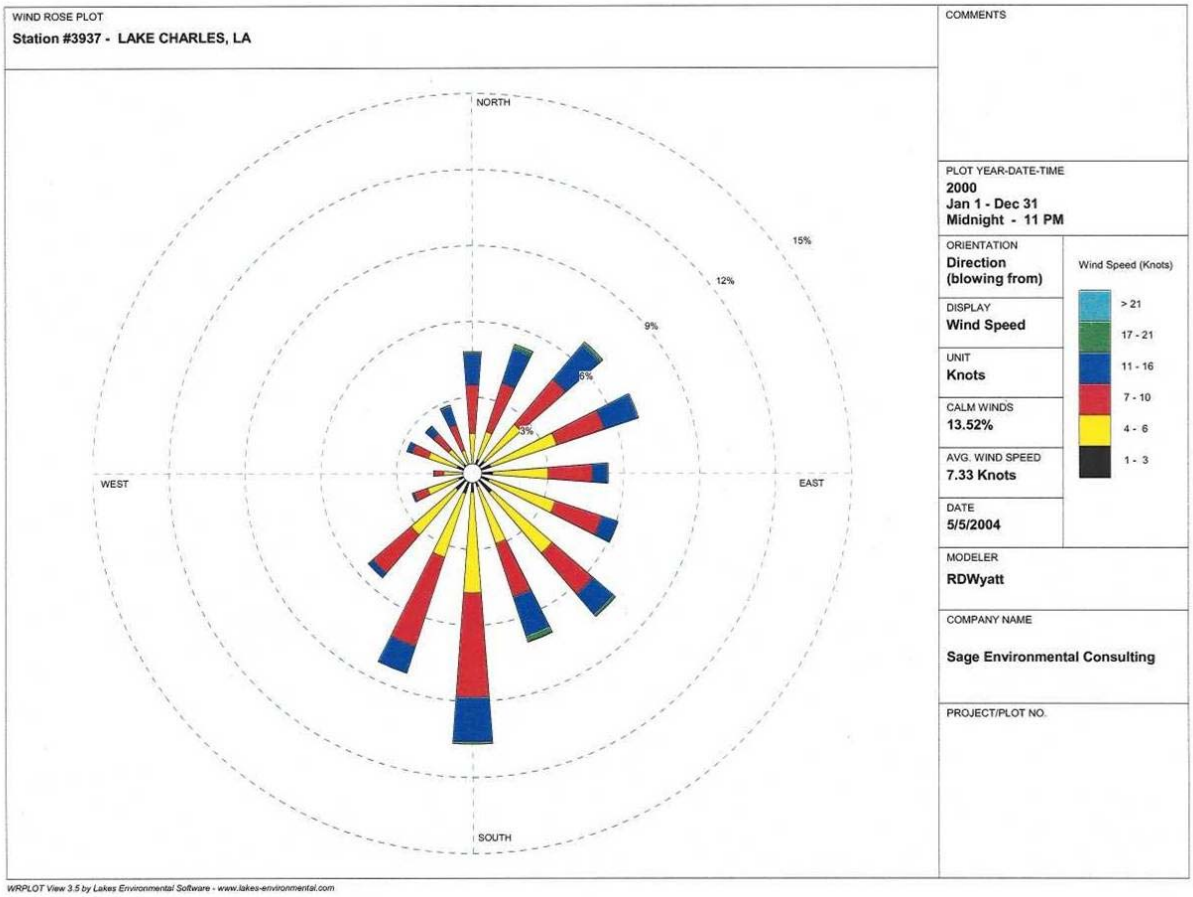


Figure I-5: Locations of Air Monitoring Stations around Northern Baton Rouge, LA

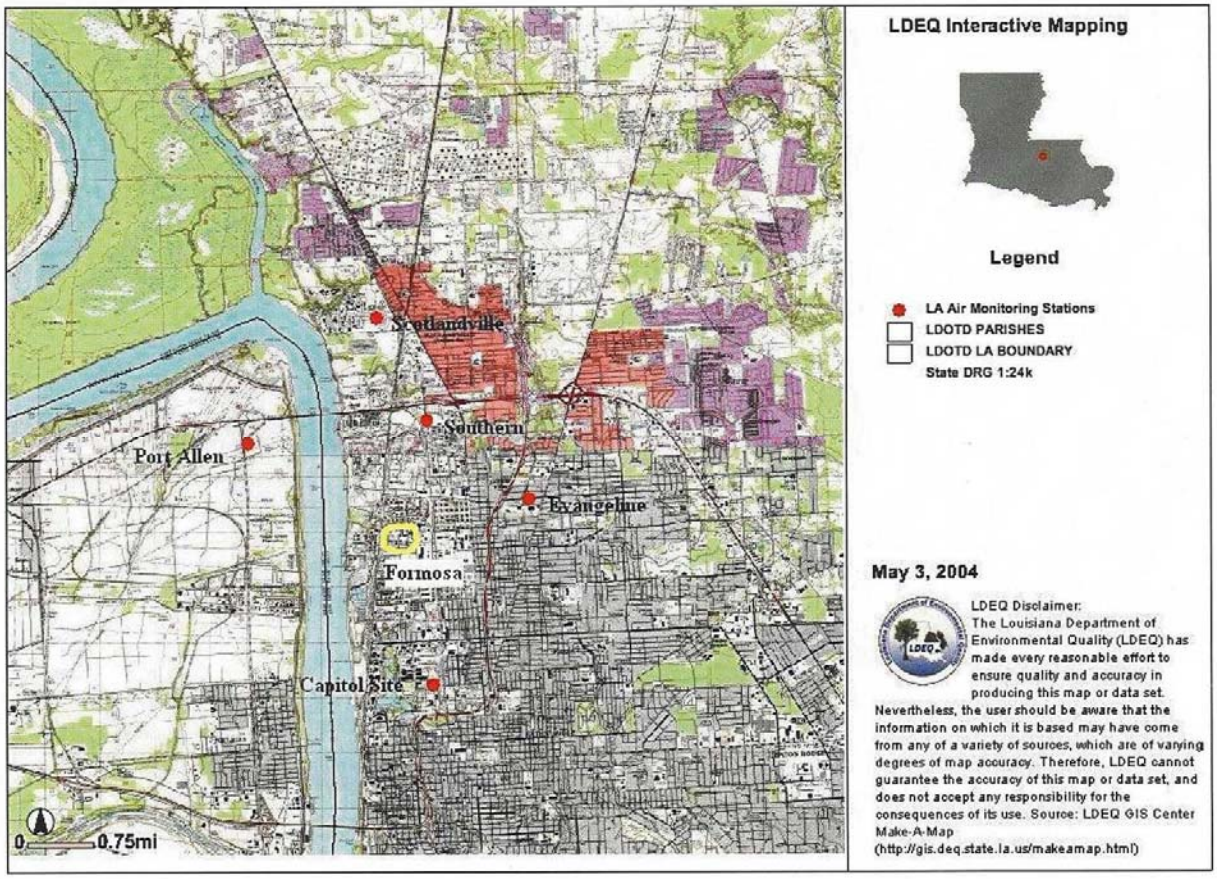


Figure I-6: Locations of Air Monitoring Stations around Southern Baton Rouge, LA

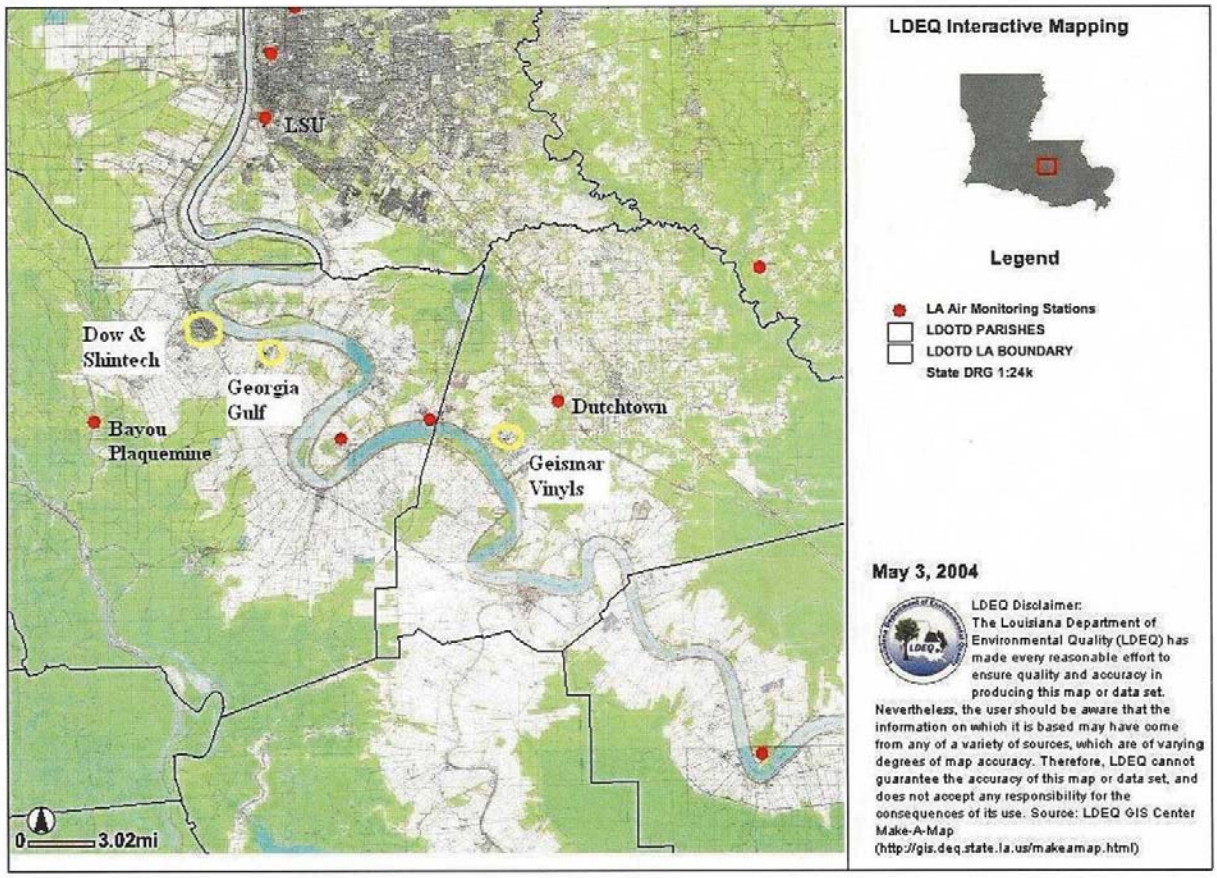
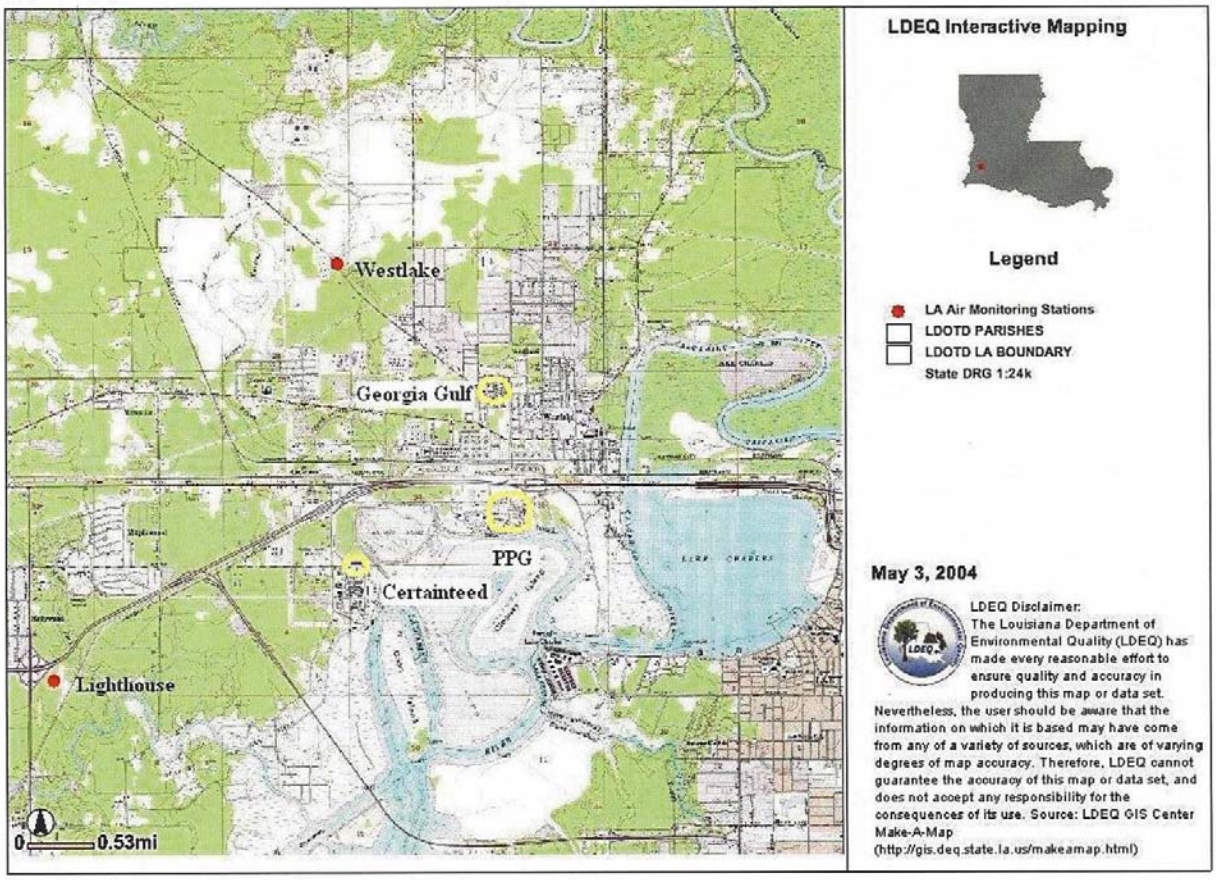


Figure I-7: Locations of Air Monitoring Stations around Lake Charles, LA



Data from the following facilities were considered in this fenceline assessment: Bayou Plaquemine, Dutchtown, Lighthouse, South Scotlandville, Southern, and Westlake, because these stations had the greatest concentrations of VCM monitored during the data sampling. The Lighthouse monitor is located nearby and to the southwest of the Georgia Gulf, PPG and CertainTeed vinyl manufacturing facilities in Westlake/Lake Charles. Assessment of air quality at this monitoring station started in 2000 as part of a parish-wide air toxics monitoring project that was funded by the vinyl industry (Sage, 2004). The Westlake monitoring station is also located close to these facilities (between 1 and 3 miles away, depending on the facility) but to the north. Further, the Southern monitoring station is located near (between 1 and 3 miles away) and to the north of a Formosa facility in Baton Rouge. The South Scotlandville monitoring station is also located slightly more than one mile to the north of the Southern station. The Bayou Plaquemine site is located approximately 6 miles south of the Dow & Shintech facilities and roughly 9 miles southwest of the Georgia Gulf facility. The Dutchtown station is roughly 3 miles northeast of the Geismar Vinyls facility (Sage, 2004). For the Southern, S. Scotlandville, Dutchtown, and Bayou Plaquemine stations, the wind is predominantly NNE, while for the Lighthouse and Westlake stations in southwest LA, it is predominantly from the south.

Monitoring data from these stations are taken on an hourly basis, a 3-hour basis, or a 24-hour basis. For the years monitored, 24-hour samples were generally taken every 5-6 days throughout the year, although some samples were taken as many as 12 days apart. This resulted in approximately 60 air samples for each year that monitoring took place.

The state of Louisiana has established an Ambient Air Standard (AAS) for vinyl chloride of 1.19 $\mu g/m^3$ (0.46 parts per billion by volume, ppb_v; annual average basis). Average yearly concentrations for vinyl chloride are shown below for the stations and the years they were monitored. These values are given in the table below. Risk estimates were generated to determine if the concentrations measured at these stations exceeded acceptable risk values of 1.0 for non-cancer risk and 1E-06 for cancer risk (based on U.S. EPA standards). The concentrations measured and risk estimates for each monitoring station are shown in the table below. The U.S. EPA has developed two different Unit Risk values to calculate cancer risk: 4.4E-06 per $\mu g/m^3$ for continuous lifetime exposure during adulthood, or 8.8E-06 per $\mu g/m^3$ for continuous lifetime exposure from birth. The Sciences report (WLATS, 2003) utilized the latter value, and this assessment has done the same to maintain consistency. This is a more conservative approach and results in higher cancer risk estimates. For example, as shown in the risk estimate table below, a continuous exposure concentration of 0.23 $\mu g/m^3$ results in a cancer risk estimate of 1 in 1 million (1E-06) (using the Unit Risk value of 4.4E-06 per $\mu g/m^3$. Therefore, this same concentration will yield a cancer risk estimate of 2E-06 (2 in one million) when multiplied by the Unit Risk value of 8.8E-06 per $\mu g/m^3$. In other words, the TG has chosen to model that the residents are individuals who have been exposed from birth throughout their lifetime and has calculated the cancer risk assessment in accordance with that decision. Estimates for cancer risk for adult exposure only would be half the values listed in the table below. These data show that continuous exposure to VCM concentrations below the AAS of 0.46 ppbv results in risk estimates higher than 1 in one million.

Table I-6: Mean Concentrations for Vinyl Chloride Monomer at LA Monitoring Stations

Date	**Calcasieu Parish**				**E. Baton Rouge Parish**				**Iberville Parish**		**Ascension Parish**	
Mean Values	Lighthouse		Westlake		S. Scotlandville		Southern		B. Plaquemine		Dutchtown	
	ppbv	mg/m3	ppbv	mg/m3	ppbv	mg/m3	ppbv	mg/m3	ppbv	mg/m3	ppbv	mg/m3
1999	NA	NA	0.115	2.9E-04	0.304	7.7E-04	0.044	1.1E-04	0.024	6.1E-05	0.041	1.1E-04
2000	0.055	1.4E-04	0.083	2.1E-04	0.214	5.5E-04	0.071	1.8E-04	0.032	8.2E-05	0.084	2.2E-04
2001	0.721	1.8E-03	0.105	2.6E-04	0.327	8.4E-04	0.107	2.7E-04	0.049	1.3E-04	0.063	1.6E-04
2002	0.082	2.1E-04	0.114	2.9E-04	0.512	1.3E-03	0.258	6.6E-04	0.036	9.2E-05	0.056	1.4E-04
2003	0.087	2.2E-04	0.099	2.5E-04	0.42	1.1E-03	0.052	1.3E-04	0.017	4.4E-05	0.023	5.9E-05
Average ‘99-‘03	0.24	6.4E-04	0.10	2.6E-04	0.37	9.4E-04	0.11	2.7E-04	0.03	8.6E-05	0.05	1.4E-04

Table I-7: Mean Concentrations of VCM and Cancer Risk Estimates Based on Air Monitoring Data in Louisiana

	Calcasieu Parish				E. Baton Rouge Parish				Iberville Parish		Ascension Parish	
	Lighthouse		Westlake		S. Scotlandville		Southern		B. Plaquemine		Dutchtown	
	Ppbv	mg/m3	ppbv	mg/m3	ppbv	mg/m3	ppbv	mg/m3	ppbv	mg/m3	ppbv	mg/m3
Average '99-'03	0.24	6.4E-04	0.10	2.6E-04	0.37	9.4E-04	0.11	2.7E-04	0.03	8.6E-05	0.05	1.4E-04
	HI[a]	ILCR	HI	ILCR	HI	ILCR	HI	ILCR	HI	ILCR	HI	ILCR
1999-2003	6.4E-03	**5.1E-06**[b]	2.6E-03	**2.1E-06**	9.4E-03	**7.5E-06**	2.7E-03	**2.2E-06**	8.6E-04	6.9E-07	1.7E-03	**1.1E-06**

Cancer Risk Estimates for Continuous Exposure to VCM Concentrations

Risk Estimates Using Unit Risk Value of 4.4E-03 $(mg/m^3)^{-1}$			Risk Estimates Using Unit Risk Value of 8.8E-03 $(mg/m^3)^{-1}$		
1E-04	0.023 mg/m^3		1E-04	0.012 mg/m^3	
1E-05	0.0023 mg/m^3		1E-05	0.0012 mg/m^3	
1E-06	0.00023 mg/m^3		1E-06	0.00012 mg/m^3	

A, HI values were generated by calculating the ratio of the mean VCM concentration and the RfC of 0.1 mg/m^3.
b, ILCR values were calculated by multiplying the mean VCM concentration by the Unit Risk (UR) value of 8.8E-03 per mg/m^3. ILCR values represent excess cancer risk over background. The value 5.1E-6 means there is a risk of 5.1 people in 1 million potentially contracting cancer due to exposure to vinyl chloride at the concentration measured at the Lighthouse Monitor over the time period indicated. Cancer and non-cancer risk values were generated by dividing the average concentration by the RfC (for HI estimation) or multiplying the concentration by the URE (for ILCR estimation) See tables above for RfC and UR values for VCM.

The risk estimates above indicate that average concentrations from all stations except for Bayou Plaquemine resulted in cancer risks (ILCR values) that exceeded a limit of one in one million; The risk estimates ranged from a low of 1.1 in one million (Dutchtown) to a maximum of 7.5 in one million (South Scotlandville). As discussed above, these values do not greatly exceed the federal risk limit and are based on the most conservative estimate of cancer risk recommended by the U.S. EPA. It is not clear that increasing regulation on emissions will reduce the risk estimates below acceptable limits. The HI values were all well below one, indicating that non-cancer systemic effects are not expected from these exposure concentrations.

I.4. Discussion of Fenceline Risk Estimates: How Do They Compare and What do They Mean?

Risk estimates for fence-line exposures in West Jefferson County, KY and two areas in LA were generated in the same manner used by Sciences International, Inc., in its 2003 report. Specifically, the average air concentration at any air monitoring station of interest was divided by the Reference Concentration (for non-cancer risks) or multiplied by the Unit Cancer Risk (for cancer risks) to provide the resulting risk estimate. This method for estimating risk assumes that a person is in the vicinity of the air monitoring station 24 hours per day, 365 days per year, for 70 years (an assumption that is generally considered to be overly conservative). It is useful to point out that the Sciences report only used air data from the sampling year of April 2000 to April 2001; this time frame coincided with the monitoring of air stations established by the West Jefferson County Task Force and others by the U.S. EPA. After April 2001, the EPA monitors were taken off-line but the remaining air monitoring stations continued collecting samples.

In our risk estimates, air samples from 2003-2005 were used to take advantage of more recent data. The three-year period was used to avoid under or over-sampling in any particular time frame (e.g., to avoid under-representing the actual risk if air samples were higher in 2003 than 2005). Samples from the years of 2001-2003 were not used for the sake of keeping the task manageable. Given the fact that the majority of air samples from any given year are below the detection limit (0.1 ppb_v), it is not believed that truncating the data caused a bias toward under representing the actual health risks. For example, a review of the data indicates that more non-detect VCM concentrations were obtained at Farnsley Middle School, Chickasaw Park, and LFPTC stations in 2002 than in subsequent years.

As is shown above, the risk estimates for fenceline exposures in Kentucky are slightly above the risk limit of 1 in one million cancers (values range from 1.38 to 5.53 in one million). Those at all but one of the eight Louisiana stations are also higher than applicable state risk limits.

These cancer risk estimates cannot be directly compared to those generated for the occupational worker. Although both fenceline and occupational risk estimates were generated using the same methodology and the same promulgated risk values from the EPA and other agencies, the risk estimates for fenceline residents are *pure* risk values, in the sense that they haven't been modified by some weighting factor or adjusted based on a scale set by either the TG or another body. The risk estimates simply represent the numerical approximation of people who might develop cancer (above the background cancer rate) if they inhale for a lifetime the concentrations of vinyl chloride that have been measured at the monitoring stations over the last few years (assuming those concentrations don't increase or decrease dramatically during the exposure period). In contrast, those for occupational workers have been normalized to the total amount of building material manufactured per year. This normalization was necessary in order to be able to

compare one type of building material to another with regard to risks during the cradle to gate period. However, when considering potential risks to residents who live near a manufacturing facility, it is not appropriate to normalize the risk values to some outside factor. To do so would run the risk of artificially decreasing the overall risk estimate or spreading it out amongst individuals who may live hundreds or thousands of miles away and who do not breathe the same air.

One limitation to this quantitative fenceline analysis is that it is only available for the vinyl industry. If monitoring stations exist around manufacturing plants for the non-vinyl alternatives considered here, they were unknown to us. Articles in the available literature speak to the potential hazards from living adjacent to aluminum and cement plants, but there are no readily available air monitoring data (and thus no quantitative estimates of the amount of material people may breathe in a day or a year) for the materials used at these facilities. It is necessary to state also that, aside from ABS plastic and crystalline silica released from concrete plants, the majority of compounds used in the manufacture of the other non-vinyl building materials are non-carcinogenic; therefore, a numerical comparison of estimated health risks from fenceline exposures to all building materials considered would be chiefly limited to one of non-cancer risks.

Nevertheless, the analysis shows that potential risks to fenceline residents are not negligible, and should be accounted for in future life-cycle analyses. For example, the cancer risks for Kentucky and Louisiana air monitoring stations were estimated at ~1-8 excess cancer deaths in one million exposed individuals. It is useful to compare occupational risks to fenceline risks on a functional unit basis. In order to do this, one needs to start with the risk estimates for occupational workers, and adjust those using the ratios of exposure concentrations and populations in each group. Therefore, for VCM, the ratio of the lowest concentration of VCM for occupational worker:resident is 827 (based on 0.13 mg/m^3 being the lowest [non-zero] concentration estimated for a VCM worker) and the ratio of the highest exposed worker (2.6 mg/m^3) to a resident is 16,538. These ratios then give us risks for the potential exposed resident when we adjust for differences in exposure duration (hours per day and days per week). The resulting values for potentially-exposed residents then range from 10^{-10} to 10^{-8}, on a functional unit basis. This range is comparable to the occupational risk estimates of workers for most of the building materials modeled in this report; the similarity of the risk estimates underscores the need to model both occupational and bystander risk in future life-cycle assessments.

Case reports and articles regarding potential health hazards of the vinyl industry in Louisiana and Kentucky have not focused on VCM-induced disease, however. Populations identified as recently experiencing high exposures of compounds suspected to be released from vinyl manufacturers include Mossville residents and individuals living in Myrtle Grove Trailer Park, Plaquemine, Louisiana. Mossville residents have exhibited high blood levels of dioxin, compared to the rest of the country. Recently, it was found that rates of soft-tissue tumors were increased in three out of four regions of Calcasieu Parish, although not were found within the boundaries of Mossville (LDHH, 2002). Some researchers have linked dioxin exposure with the incidence of soft tissue sarcomas in humans, but other studies have not found an association (Bertazzi et al., 2001; Bodner et al., 2003; Pesatori et al., 2003). Residents in the Myrtle Grove Trailer Park were exposed to VCM-contaminated water from a community well from sometime after April, 1994 to March 31, 2001. The ATSDR (Agency for Toxic Substances Disease and Registry) performed a health consultation of the trailer park well system based on a request from

the residents. Agency representatives analyzed water samples from the well for vinyl chloride monomer and other contaminants. VCM levels exceeded federal safe drinking water standards from 1997-2001. Based on its analysis of the water contamination, as well as modeling exposures from drinking water, bathing, and other household activities, the agency determined that the exposures were too low to cause adverse health effects in either children or adults (ATSDR, 2002). Nevertheless, the risk values above, in combination with the case reports, indicate past and current exposures to vinyl-associated compounds are occurring and risk estimates in some cases exceed *risk levels generally considered acceptable in federal and state standards.*

Appendix J: Peer Review

The TG actively sought out peer review of the revised draft report in an effort to verify validity of methodology and to gain constructive criticism of the approaches used. Sections of the report have undergone a peer-review process by recognized experts in the appropriate fields. The sections on phthalate toxicity and exposure were reviewed by Dr. Russ Hauser from Harvard School of Public Health, and Dr. Thor Larsen from National Institute of Occupational Health, Denmark. The section on fenceline risk was reviewed by Dr. Jon Levy from Harvard School of Public Health. Analyses estimating dioxin emissions were reviewed by Dr. Ulrich Quass from Müller-BBM GmbH, Germany. Mercury exposure was reviewed by Dr. Glenn Rice from Harvard School of Public Health. In some cases, comments from the peer review resulted in additional analyses; these are discussed in the appropriate appendices.

J.1. Mercury

Dr. Glenn Rice, Harvard School of Public Health:

Dr. Glenn Rice is an author of "Economic Valuation of Human Health Benefits of Controlling Mercury Emissions from U.S. Coal-Fired Power Plants" (2005). The study estimates the health benefits of reducing mercury emissions from coal-fired power plants in the United States via reductions in methylmercury concentrations in fish, the consumption of which is the primary pathway of human exposure to methylmercury.

He suggested the following:

(1) Check EPA's 1997 Mercury Study Report to Congress and more recent inventories developed by OAQPS for mercury to confirm this emissions estimate

- We checked this source (EPA, Regulatory Impact Analysis for the Final Clean Air Interstate Rule; http://www.epa.gov/interstateairquality/rule.html) and found that mercury-related health effects were not quantified in the EPA's analysis.

(2) Instead of using the QALY estimates from a simple environmental model, in which a decrease in mercury deposition would result in a proportional decrease in marine fish methylmercury levels, model the impacts as a range, citing the freshwater fish impacts as a low end and the freshwater + marine as a high end value.

- Previously, we used 280 QALYs per ton of mercury emission for the lower bound and 800 QALYs per ton of mercury emissions for the upper bound. To follow Dr. Rice's advice, we assumed that only freshwater fish consumers are affected for the lower bound, while for the upper bound freshwater and marine fish consumers are both affected. As a result, the range of effects became larger, from 7.3 QALYs per ton of mercury to 805 QALYs per ton of mercury. Consequently, the lower bound of mercury effects was about a factor of 40 lower than the previously estimated. However, this didn't change the overall conclusion since the mercury-related health effects are relatively small compared to other impacts such as dioxin-related cancers and PM-related premature deaths and respiratory effects.

J.2. Dioxin Emissions

Dr. Ulrich Quass, Müller-BBM GmbH, Germany. Author of European Dioxin Inventory Stage I (1997) and Stage II (2000). The full report of Dr. Quass is available upon request. He focused mainly on the topic of uncontrolled waste burning processes. His comments are summarized below:

(1) Dioxin emission factors related to the production phases were checked briefly and in general were found to be plausible in comparison to published data.

(2) The authors of the report are aware of the complexity of the dioxin emission analysis and provide where needed a detailed analysis of the likely influence factors and their impact on the results. They also clearly indicated assumptions made and provide plausibility checks or proofs if available. Thus the general methodological approach is scientifically sound.

(3) The regression analysis approach chosen to evaluate the lower and upper bound emission factors generates the impression of higher accuracy than supported by the actual experimental results (Lemieux, 2003).

- In response, we conducted sensitivity analyses: (1) omit data point 1, 12 and 13 in Figure D-2, (2) omit data point 2 in

Figure D-2 only. The slope of the regression line for (1) is 0.07 (P-value >0.3). The slope of the regression line for (2) is 0.32 (P-value of 0.005). Since we use the upper and lower 95% confidence limits of the slopes to represent the upper and lower bounds, the regression lines generated by omitting the extreme values are within the range of our upper and lower bound values that we have estimated originally. Therefore, to clarify our approach, we revised the figure to show the actual range of confidence-limit based slopes that we used in our analysis.

(4) The approach of using EPA's dioxin emission rate for landfill fires (1,126 g TEQ/year) and UNEP's emission factor of 1000 ng/kg to estimate the amount of burned materials is "closed loop." Since both input values refer to the same Swedish research results published in the 90s, the mass of landfill waste burned in the U.S. would just have been

calculated by use of the specific waste generation rate in Sweden multiplied with the ratios of the respective capita numbers.

- In response to this comment, we have tested the suggested approach using the Swedish data. Our original calculation gives a very close estimate to the alternative method (see Appendix D, Life Cycle Assessment Emission Factors, section on Emission Factors for Disposal Phase.

(5) The fraction of burned waste in landfill is estimated by taking into account the probabilities of underground and surface fires as well as the temporal development of the landfill mass due to the annual input and losses by fire. The calculation appears to be kind of methodological "overkill," as at the end the result further used is just a range of percentages of burned waste related to overall waste generation. The resulting range of percentages (0.8 to 2.0% of mass annual input into landfills) must therefore be assessed as a guess and is likely to be too small.

- We have decided to leave our analysis as it is in order to avoid too crude calculations.

J.3. Phthalate Exposures

Comments were provided by Dr. Russ Hauser, Associate Professor of Environmental and Occupational Epidemiology at Harvard School of Public Health and by Dr. Søren Thor Larsen, researcher at Denmark's National Institute of Occupational Health. As part of his research interests, Dr. Hauser is conducting studies on the effects of phthalate exposures on men's reproductive health. Dr. Larsen studies the effects of phthalate exposure on asthma and other respiratory conditions. Copies of their full comments are available upon request.

Dr. Hauser had specific suggestions to clarify the discussion of metabolism of phthalates and adverse effects of phthalate exposure in humans and animal models. He also suggested that the sections be updated with recent research on phthalate toxicity as reviewed in the 2005 update to the DEHP report by the Center for Environmental Risks to Human Reproduction.

- The sections on phthalate exposure and toxicity have been completely rewritten to focus predominantly on the CERHR expert panel's assessment of DEHP toxicity. It is the opinion of the PVC TG that the CERHR 2005 report provides a useful synthesis of DEHP exposure and toxicity in both humans and animal models.

Dr. Larsen's comments were predominantly focused on clarifications in the discussion of the role of DEHP in the onset of asthma and other allergic responses. All comments were incorporated into the revised report.

J.4. Fence-line Exposures

Comments were provided by Dr. Jonathan Levy, Mark and Catherine Winkler Associate Professor of Environmental Health and Risk Assessment at Harvard School for Public Health. A complete copy of Dr. Levy's comments is available upon request. Dr. Levy's comments focused primarily on additional analyses that could better inform the reader and he raised a question regarding the incorporation of individual risk into an analysis of total population risk. Dr. Levy's comments are summarized below.

(1) It was suggested that additional information regarding screening criteria, limits of detection, and the decision for selecting compounds to measure at each monitoring site be included.

- The text was rewritten to increase clarity regarding the selection of compounds of concern in the original risk assessment of the Louisville, KY residential areas.

(2) Additional information regarding the risk assessment should be provided including potency factors of compounds, a discussion of how the non-cancer risk assessment was done, and adding more detail about assumptions used in the risk assessment, including providing the median exposure values because median cancer risk estimates are shown.

- Additional text was provided regarding summing non-cancer Hazard Indices, which is in accordance with U.S. EPA risk assessment guidelines, and cancer potency values for the carcinogens assessed in the risk assessment. Median concentrations for the initial risk assessment in the Louisville, KY area were not provided in the original report (WLATS, 2003) and the TG did not have access to the raw data. Therefore, it was not possible to provide median exposure concentrations.

(3) A suggestion was made to include wind rose data and maps with locations of the air monitoring stations.

- Maps of monitoring stations and appropriate wind roses were added to the text.

(4) A suggestion was made to assess acute risks to determine if sporadic high exposure concentrations pose an unacceptable risk.

- Acute risks were assessed in the subsequent analysis of current fenceline risks for Louisville, KY areas.

(5) A comment was raised regarding the utility of individual risk when LCA prefers the evaluation of population risk.

- An assessment of population risk for cancer endpoints was discussed in the original report. Some stakeholders believed that the original analysis of population risk was not specific to residents who might live adjacent to, or downwind, of PVC facilities and other industrial plants. Therefore, in the fenceline analysis, it is appropriate to present the data in terms of risk to an individual receptor so that these stakeholders can understand what the exposures mean for the health of each individual person who might be living in those areas.

Appendix K: Sensitivity Analysis of Aluminum Windows

K.1. A comparison of LCA results for aluminum windows with and without thermal breaks

The two figures below show that the relative performance of wood, aluminum and vinyl window frames do not change when thermal breaks are added to the aluminum windows; their life cycle health and environmental impacts remain worse than those for PVC frames.

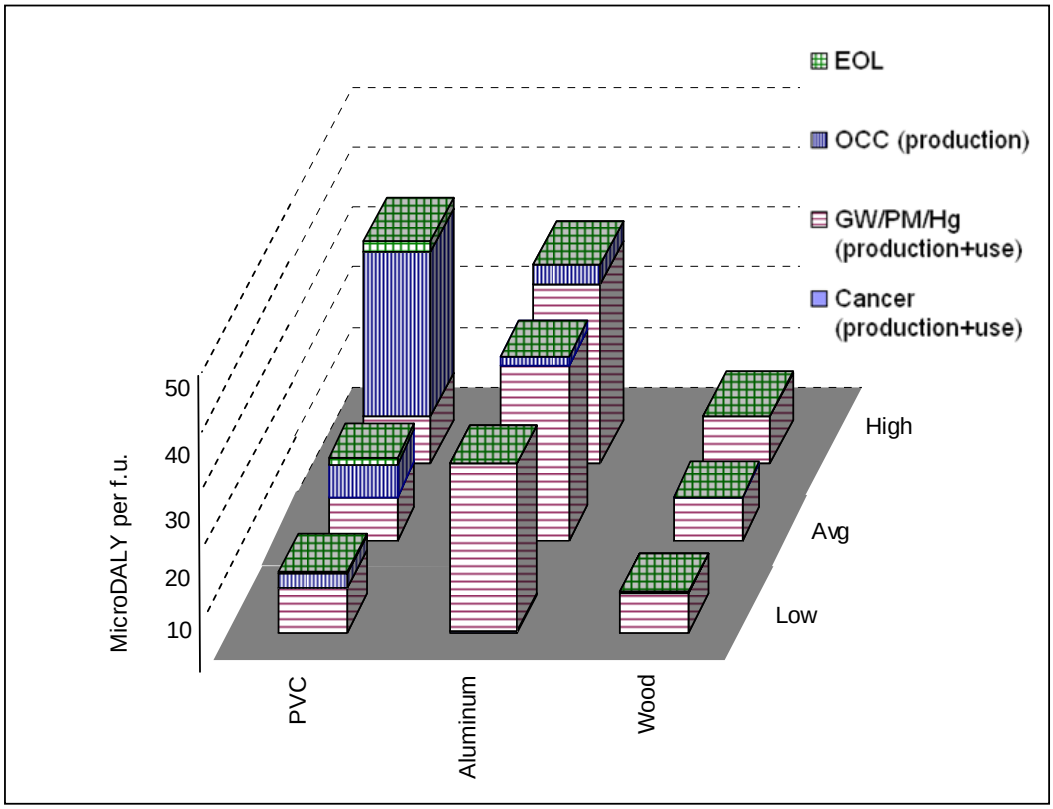


Figure K-1: High, average and low estimates of human health impacts (microDALY per functional unit) by life cycle stage – Window frames including aluminum frames WITHOUT thermal breaks

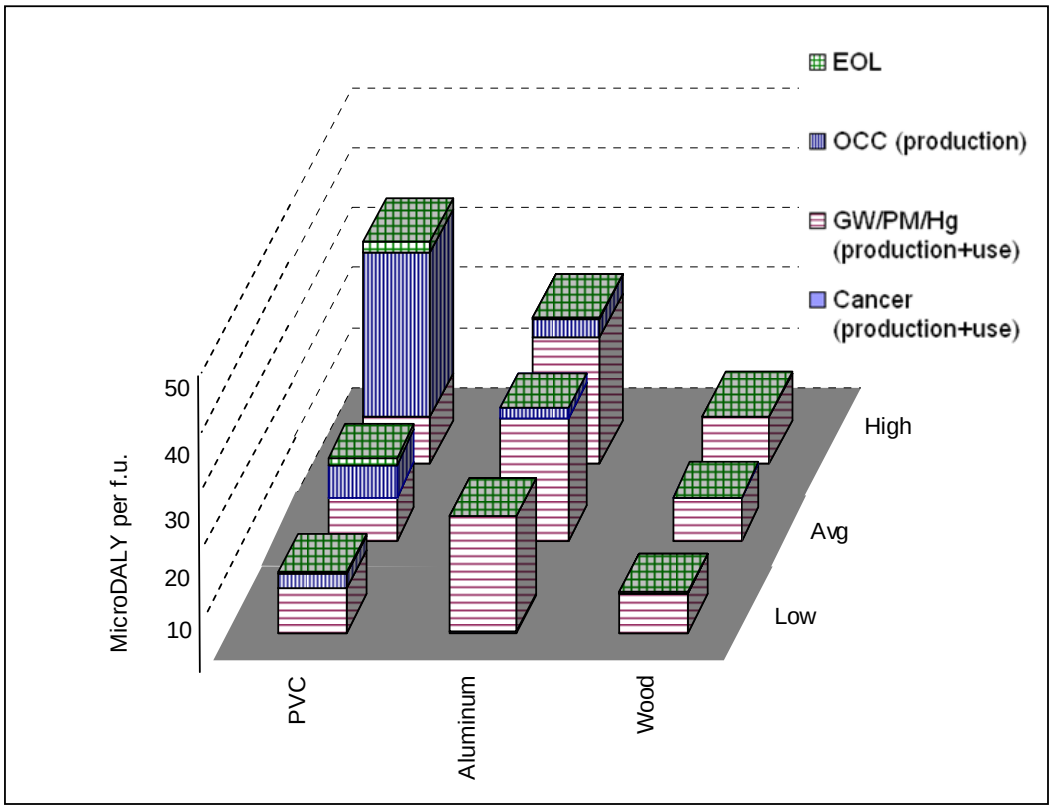


Figure K-2: High, average and low estimates of human health impacts (microDALY per functional unit) by life cycle stage – Window frames, including aluminum frame WITH thermal breaks

Appendix L: Task Group Biographies

Kara Altshuler

Dr. Altshuler has twelve years of experience in consulting with specific focus in areas of human health risk assessment, toxicological analyses and support, regulatory assessment, outreach, international environmental guidelines, and peer review. Dr. Altshuler has worked for such companies as ICF International, Sciences International, and Alceon Corporation.

Dr. Altshuler has developed significant updates to toxicity reviews for manganese and chloroethane for ATSDR, has written updates to IRIS toxicity profiles for dichloroacetic acid and copper, and has written numerous documents on toxicity of compounds in water for the EPA and WHO. She has performed toxicity reviews of compounds with potential to come into contact with food for the FDA's Center for Food Safety and Nutrition, and has managed a team of scientists and support personnel tasked with assisting the Office of Pesticide Programs' Antimicrobial Division in its risk assessment of pesticides under new and renewing applications. Dr. Altshuler was a co-author of a three part paper series on children's environmental health for the U.S. EPA's Office of Childrens' Health Protection. Dr. Altshuler has developed numerous risk assessments for compounds developed as alternatives to Halon 1301 and other ozone-depleting compounds (ODC) for U.S. EPA's Significant New Alternatives Policy Program (SNAP). Also in that capacity, she has developed risk-based exposure limits for occupational workers, end-use consumers, and the general public for these ODC replacements.

Scot Horst

Scot Horst began Horst, Inc., a sustainable materials consulting firm, in 1994 which specializes in developing and working on innovative environmental programs relating to materials technologies and testing. This work has ranged from environmental verification work on bio-based technologies with the Civil Engineering Research Foundation to extensive work with the cement industry including a blended cement carbon dioxide offset program with the Climate Trust in Oregon.

In 1999 Mr. Horst co-founded 7group, a multi-service green building consulting LLC, where he serves as President. As a LEED Accredited Professional he has worked on over 30 LEED projects. Horst was a member of the Materials and Resources TAG and currently sits on the Technical Scientific Advisory Committee and the LEED CI Core Committee. He is a LEED faculty member and, as a partner in 7group, writes credit interpretation rulings and reviews certifications for the USGBC.

Horst also serves as Vice-President of Athena International, the U.S. non-profit affiliate of the Canadian Athena Sustainable Materials Institute. In this capacity he chairs the PVC Task Group. Horst is involved with a broad range of work related to Life Cycle Assessment (LCA), including regional database development and LCA education.

Nadav Malin

Nadav Malin is vice president of BuildingGreen, Inc. and serves as editor of *Environmental Building News,* a monthly newsletter on environmentally responsible design and construction and coeditor of the *GreenSpec* product directory. He is chair of the Materials and Resources Technical Advisory Group for the U.S. Green Building Council's LEED™ Rating System, a LEED Trainer, and a LEED Accredited Professional. He also serves on the LEED Technical and

Scientific Advisory Committee's PVC Task Group. He also represents BuildingGreen on the team that has been contracted by the State of California to develop and Environmentally Preferable Product Database for schools.

Malin has written on environmentally preferable products for the AIA/ Wiley *Handbook of Architectural Practice* and was a principal author of the Applications Reports for the AIA's *Environmental Resource Guide* that compares the environmental value of different building materials in various applications. He has written numerous articles for publications including *Architectural Record* and *The Construction Specifier.* He consults and lectures widely on sustainable design, with a particular focus on green materials. In addition to running LEED training workshops, he has taught seminars for various USGBC chapters, CSI chapters, state AIA chapters, and private architecture firms. He also manages the U.S. Department of Energy's High Performance Buildings Database project, and leads the content development team for Web and software resources at BuildingGreen.com.

Greg Norris

Greg Norris founded and directs Sylvatica, a life cycle assessment (LCA) research consulting firm in Maine, USA (www.sylvatica.com). Norris is Program Manager for the United Nations' Environment Program's (UNEP) global Life Cycle Initiative, directing the Program on Life Cycle Inventory Analysis. He teaches graduate courses on LCA and Industrial Ecology at the Harvard School of Public Health, where he also advises graduate students from HSPH and visiting research fellows from abroad. He consults on LCA and sustainable consumption to UNEP, to Federal and state agencies in the U.S., and to the private and non-profit sectors. Norris is founder and executive director of New Earth, a global foundation for grass-roots sustainable development (www.newearth.info). Norris has developed several software tools to assist analysis and decision-making related to Life Cycle Assessment and sustainable enterprise. Recent research integrates socio-economic pathways to human health within the LCA framework and develops a human need-based approach to sustainable consumption analysis. Norris is Adjunct Research Professor at the Complex Systems Research Center, University of New Hampshire; he is a Program Associate in the Center for Hazardous Substance Research at Kansas State University, and an editor of the International Journal of Life Cycle Assessment.

Yurika Nishioka

Yurika Nishioka received her Ph.D. from the Harvard School of Public Health in 2004. Her research focused on applying and combining quantitative models in the following domains: energy consumption in the industrial, commercial, and residential sectors; inter-linkages of industrial sectors using life cycle assessment methods, with process-based models as well as economic input/output models; pollution dispersion modeling, and epidemiological models to estimate health impacts of pollution exposures. She has published her research in a variety of journals, and presented the work at many professional conferences. She is continuing to advance her research on the potential impacts of energy-saving technologies; an association of U.S. manufacturers has funded her to expand the modeling and results to address the impacts of energy conservation measures on greenhouse gas emissions, taking into account the full life cycle (economy-wide) industrial interactions, using a dynamic economic/emissions modeling framework. She has assistant-taught several graduate-level courses at Harvard and MIT.

Appendix A: Sources Cited

The sources included in this appendix have been cited in *Assessment of Technical Basis for a PVC-Related Materials Credit in LEED®* or the accompanying appendices. Additional references can be found in the database at **http://www.usgbc.org/DisplayPage.aspx?CMSPageID=153**.

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