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

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Preface

This report has been prepared under the auspices of the US 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 impaneled 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 Appendix D of this report.

TSAC has developed a nine-step process for preparing positions on technical issues, and one of the most important elements of this process is obtaining input from the various stakeholders on an issue. It is as part of this public input process that this draft report is being released for public review, with the following objectives:

- to present the analytical methodologies and results and the information resources being used;
- to present the findings and conclusions drawn from the analysis to-date; and.
- to receive public comment on this work prior to issuing a final report.

To the extent that the public comments identify new data, new or different methods of analysis, or possible new interpretations of the analytic findings, this input will be seriously considered by the PVC TG and TSAC as the assessment of this issue is finalized.

The technical analysis provided by TSAC will ultimately be 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.

Malcolm Lewis, TSAC Chair

Executive Summary

In the year 2000, during the process of developing the LEED Commercial Interiors rating system, draft credit language was developed providing credit for the avoidance of PVC materials. The US Green Building Council decided that further technical knowledge was needed to determine the soundness of this 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, the USGBC's Technical and Scientific Advisory Committee (TSAC) established a PVC Task Group, charging it 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."

General Approach

In response to its charge, the PVC Task Group identified several building applications in which PVC-based materials have a significant market share, and investigated whether for those applications the available evidence indicates that PVC-based materials are consistently worse than the alternative materials in terms of environmental and health impacts, or vice versa. Applications representing a diversity of uses of PVC in buildings were chosen for study. The applications and materials that were studied are:

Siding: vinyl, aluminum, wood, and fiber-cement

Drain-waste-vent pipe: PVC, ABS (acrylonitrile-butadiene-styrene), cast iron

Resilient flooring: sheet vinyl, vinyl composition tile (VCT), linoleum, cork

Windows: vinyl, aluminum, wood.

To compare the impacts of alternative materials choices two assessments, a life cycle assessment (LCA) and a risk assessment, were completed for each material used in each application. The detailed analytical process used is summarized below.

Such assessments can never be better than the data used as inputs to the process. To assess the quantity and quality of the data publicly available on these topics, a Web-based relational database was created to map the available information resources onto a matrix representing the knowledge that would be needed to reach a conclusion regarding the charge. This database consists of two main components: 1) a three dimensional matrix of life cycle cells, in which each cell represents the intersection of a material, an impact category, and a life cycle stage; and 2) a list of information sources, including stakeholder submissions, papers identified by literature searches, and sources provided by individual Task Group members. Each source is linked to any and all cells in the matrix to which it contributes information. The quality and nature of the knowledge conveyed is also characterized. A total of almost 2,500 references were reviewed as part of this analysis and are cited in the database. The database is available on-line at pvc.buildinggreen.com. It represents an important aspect of the work done by the PVC Task Group, and is a part of this report by reference.

Life Cycle Assessment

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 actual 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 chosen include those listed in EPA's TRACI (Tool for Reduction and Assessment of Chemical and other Impacts) method for life cycle impact assessment, enhanced by others considered important to stakeholders. The impact categories and units of measure used in this study are:

Environment/Resource
- Acidification (+H millimoles)
- Ecotoxicity (g 2,4-D-equivalent)
- Eutrophication (g N-equivalent)
- Fossil Fuel depletion (Surplus MJ)

Combined human and ecosystem health
- Ozone Depletion (g CFC-11 equivalent)
- Global Warming (kg CO_2 equivalent)
- Photochemical Smog (g NOx equivalent)

Strictly human health
- Human Health: Particulates – HHPM (microDALY-Disability Adjusted Life Years)
- Human Health: Cancer – HHC (microDALY)
- Human Health: Other – HHO (hazard index)

For each of the four building applications, a *functional unit* is defined to allow for meaningful comparison of the materials. The functional units for siding and flooring are 1 ft2 of material with a 50-year service life. The functional unit for pipe is 1 linear foot of schedule 40 or comparable pipe with an inner diameter of 3 inches with a 50-year service life. The functional unit for windows is a two panel Swiss window of 1.3 meters by 1.65 meters with 23 square foot in area, over a 50-year service life.

Risk Assessment

Risk assessment attempts to quantify potential risk for developing adverse health effects following exposures to environmental toxicants (compounds which have the potential for causing toxicity in living things). This quantification is done by comparing doses of the toxicants in a person (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.

In order to determine occupational risks from exposure to compounds used or made in the manufacture of PVC and non-PVC building materials, a human health risk assessment was conducted for occupational workers involved in the manufacture of the building materials. In addition, for resilient flooring materials only, potential risks from installation and use were modeled, based on a school classroom environment. Exposure data were not available for the construction and end use phases for other building materials (e.g., exposures to solvent cements or welding fumes) or there were limited or no exposures during these life cycle phases. Exposure of end users in a home environment was not assessed because comprehensive data of compounds emitted from the flooring materials was not available. Risk estimates also were not generated for the general population; exposures of this group were assessed in the LCA analysis.

Risk estimates were developed for both non-cancer and cancer effects. Summary risk estimates for non-cancer and cancer effects for each type of exposed individual and building material were then normalized for the functional unit for each building material. This normalization process allows for the comparison of the health risks for each building material's functional unit.

Morbidity was assessed in the occupational and general population exposure assessments by evaluating disability-adjusted life years following exposure. In addition, a brief evaluation of other types of morbidity associated with manufacture and use of the building materials was performed.

Integration of LCA and Risk Assessment – Findings

Combining the conventional life cycle assessment and risk assessment methods allows, perhaps for the first time, the comparison of risk-assessment data on occupational exposures with life cycle inventory information on health risks to the population at large. Environmental impacts were derived solely from the LCA studies. Health effects derived from these two separate assessment methods were merged by converting the results into units of measurement for human-health impacts (hazard index and disability-adjusted life years) that are common to both systems. This comparison shows that occupational cancer risks are far from negligible in relation to health risks to the general population, and should be considered more widely as part of comprehensive life cycle methodologies.

Siding

This study found that wood siding performs best for most impact categories, while fiber cement performs the worst among the options for most impact categories. Aluminum had the largest ranges of values primarily due to various recycled contents. Assuming that a standard recycled content for aluminum siding is 30%, it tends to be worse than wood siding in most categories, but better than fiber cement in most categories. This group of materials was found to be more influential in relation to human health, ecological effects, fossil fuel depletion, global warming and smog, and less influential on acidification and ozone depletion.

Aggregated human health impacts of carcinogens as well as particulate emissions show that vinyl siding leads to the highest total mortality risks, while wood siding leads to the lowest, though the differences are small. Almost all mortality risks are driven by population exposure to particulate matter for all siding types except vinyl, for which the occupational risks account for 29% of the total mortality risk. As for non-carcinogenic, toxicological human health impacts, hazard indices indicate that the non-lethal risks for the occupational exposures are much higher than those for the population exposures via the environmental pathways.

Pipe

For the piping options, the LCA results show that, for many impact categories, ABS pipe tends to perform relatively well. Unlike the situation with siding, neither of the other materials (PVC and cast iron) performs consistently poorest across the impact categories. All impact categories were found to be of affected at a similar level by this group of materials except for acidification and ozone depletion, which are less affected.

Aggregated health impacts of carcinogens as well as particulate emissions show that PVC and cast iron pipes lead to higher total mortality risks than ABS pipe. For all pipe types, occupational risks as a fraction of the total mortality risks vary from 22% for cast iron to 24% for vinyl to 29% for ABS. As for non-carcinogenic, toxicological human health impacts, hazard indices indicate that the non-lethal risks for the occupational exposures are much higher than those for the population exposures via the environmental pathways.

Flooring

For the flooring options, the LCA results show that cork flooring performs very well in most categories. Unlike siding, we do not observe one material option performing consistently poorest among the options across the impact categories. Vinyl has the highest values in some impact categories, VCT in others, and linoleum in others. In general, linoleum tends to perform better than the two vinyl options on a majority of impact categories, though it performs poorest in some impact categories. This group of materials was found to be most influential in relation to eutrophication and ecological effects, and least influential on acidification and ozone depletion impacts.

Findings on aggregated health impacts of combined cancer and particulate effects are affected by a wide variation in published particulate results for VCT. Our sample of results contains seven low values and three values which are 50 times greater; thus, the comparative conclusion concerning which material has the highest aggregated health impacts depends on whether we choose the mean or median value for VCT as a basis of comparison. The median provides a measure of central tendency that is more robust, less susceptible to outliers, than the mean, and for this reason we have tended to use median results as our basis of comparison in this report. Note that the selection of median versus mean only changes the aggregated human health rankings of materials for flooring, not for pipe, siding, or windows. Cork has the lowest combined mortality impacts. For non-carcinogenic, toxicological health effects, hazard indices indicate that the non-lethal risks for the occupational risks during the manufacturing stage tend to be higher than the population non-lethal risks for all flooring alternatives.

Windows

The results for windows are remarkably similar across the product alternatives. This is because the total life cycle results are not dominated by the frame material, but rather by the glazing and the usage phase energy use. This group of materials was found to be most influential in relation to ecological effects, global warming, and smog, and least influential on acidification impacts and ozone depletion impacts.

Aggregated health impacts of carcinogens as well as particulate emissions show that the comparative performance of the window alternatives is entirely driven by differences in the energy efficiency of the studied alternatives. Thus, among our three frame types studied, the aluminum-framed window had the worst energy efficiency, while the wood and PVC framed windows were virtually identical in efficiency. The proper conclusion to be drawn from this analysis is that from a life cycle human health perspective, the key issue is window energy efficiency rather than frame material. For the non-lethal, toxicological health effects, the occupational exposures lead to higher hazard indices than the population risks associated with outdoor environmental emissions for all window alternatives.

Human Health Risk Summary

Several findings emerge when results from all four product groups are reviewed:

- Mortality – for the four product groups evaluated, the total life cycle mortality results are dominated by particulate exposures, rather than cancer. Looking at cancer risks only, for all the product alternatives studied, the occupational cancer risks dominate the population cancer risks posed by the life cycle releases of carcinogens to the environment.
- Non-cancer, toxicological health effects – for all four product alternatives studied, the occupational non-lethal risks dominate the population non-lethal risks posed by the life cycle releases of toxicants to the environment. While these non-cancer risks do not appear to be particularly important relative to the mortality risks posed by particulate emissions noted above, LCA tends to focus only on population risks and to neglect occupational risks; the integrated approach used in this study brings a needed balance to the analysis.

Note that the approach to estimating occupational risks, and to normalizing those risks to the functional units of product produced, has been based on the best available data. It is recommended that the data coverage and basis for addressing occupational exposures be improved and the analyses and findings in this report be updated accordingly.

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.

Air Monitoring Data

An assessment of data from Kentucky and Louisiana air monitoring stations was performed. Data were limited for concentrations of ethylene dichloride (EDC) in Kentucky, precluding any substantive conclusions. In general, however, the concentrations were low. Annual average air concentrations of vinyl chloride monomer (VCM) at Louisiana monitoring stations indicated that two stations had averages that exceeded the state's Ambient Air Standard (AAS) for this compound. Averages taken over all available monitoring years (1999-2003 or 2000-2003), however, were below the AAS. These values do not indicate a significant human health hazard. Air data taken closer to ground level, via the Trace Atmospheric Gas Analyzer (TAGA) truck, over 5 days in 1999, indicate concentrations that were significantly higher than the AAS, however. Because these data are obsolete, additional data would be necessary to determine if potential health risks exist to residents nearby and downwind of vinyl facilities.

Cancer incidence rates in Louisiana are generally comparable to or lower than, national averages. Cancer death rates, however, particularly in white males, are higher than national averages. Data indicate late diagnosis and lack of access to cancer screening and preventative health care contribute to the increased death rates in Louisiana. The majority of Louisiana parishes have liver cancer incidence rates that are well below the national and state average.

Phthlate Exposure

Data regarding the role of phthalates in endocrine disruption and the induction of asthma in children were reviewed. Occupational exposure to phthalates is much greater than those in the ambient environment. Studies show that infants exposed to high levels of di(2-ethylhexyl)phthalate (DEHP) did not develop any adverse effects with respect to sexual development or organ function later in life. In general, animal studies indicate that levels of phthalates required to affect sexual development or maturation are much higher than the levels to which the average person is exposed. In vitro studies show that DEHP is not estrogenic. Two studies have shown a link between asthma and rhinitis in children and high levels of phthalates in the home environment. Data are lacking from these studies regarding the presence of other risk factors and their potential contribution to these respiratory conditions. Additional data are needed before conclusions can be made regarding phthalate exposure and the onset of asthma in children.

PVC in Fires

Studies on PVC fires indicate that with proper protective equipment, firefighters are not at increased risk for exposure to combustion products from this plastic. Limited data indicate that backyard burning may be a very significant source of localized dioxin concentrations. Dioxin levels emitted from barrel burning varies significantly with the chlorine content of the fuel source.

Conclusions & Recommendations

The Task Group draws the following conclusions from the research described in this report:

1. Using current data for LCA and risk assessment, our analysis of the chosen building material alternatives shows that PVC does not emerge as a clear winner or loser. <u>In other words, the available evidence does not support a conclusion that PVC is consistently worse than alternative materials on a life cycle environmental and health basis.</u> With the possible exception of cork flooring, neither is any material dominantly best across all impact categories. Instead, there are some impact categories in which each material performs poorly and others in which it performs well. This result holds up even though we have subjected both the LCA and the risk assessment portions of the analysis to sensitivity analyses. <u>Therefore, the *current body of knowledge* as analyzed in this report in Section 3 as it relates to the Task Group's charge from TSAC does not support a credit in the LEED rating system for eliminating PVC or any particular material.</u> Further, with respect to a PVC-related credit, the available evidence indicates that for some product categories, such a simple credit could steer designers to use materials which performed worse over their life cycles with respect to the bulk of the impact categories.

2. Section 4 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.

3. The Task Group's charge includes "…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." Based on its analysis of current quantifiable data, the Task Group concludes that material-based credits that discourage

the use of specific materials are unnecessarily "blunt instruments." Rather, human and environmental health would be better-served by developing credits based on issues or impacts such as those on human health and ecosystems.

4. While credits that promote or discourage the use of specific materials across a range of applications may be suspect, the analysis in the report has shown that within a given product category predicted health and environmental impacts of material alternatives can differ widely. Thus, within product categories more and less desirable materials can be identified.

The foregoing conclusions represent the Task Group's advice to the LEED Steering Committee in response to its charge. Additionally, based on its work on this issue the Task Group has developed certain policy recommendations that would build upon the analysis techniques developed in this report and would help to fill the gaps in knowledge identified herein. These recommendations are separate from the conclusions and should be weighed with other factors as they are considered by the Steering Committee. Specific recommendations include the following:

- The literature is overwhelmingly PVC focused, and additional research needs to be done on the risks associated with alternative materials to move towards comprehensive comparative analysis. 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, lead the Task Group 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.
- In the long-term, the Steering Committee is encouraged to consider developing issue-based credits in conjunction with other developments under way for LEED version 3.0. For such issue-based credits to be technically sound, they should be informed by both LCA and risk assessment. The Task Group is willing to share and develop examples of this approach.
- In the shorter term, the Steering Committee should consider allowing Innovation and Design credits in current versions of LEED for projects that fill data gaps and provide credible evidence that is in keeping with the findings of this report and that furthers the body of knowledge toward lowering health impacts of building materials. The Task Group recommends beginning this effort by working toward eliminating a class of pollutants and/or particulate emissions associated with building materials. This information can be used to support the development of scientifically sound credits for future versions of LEED and will provide incentive for project teams and others to research the issues that could make such credits scientifically robust. These ID credits and related CIRs should be reviewed against the integrated methodology outlined herein in order to maintain consistency with the findings of this report.

1 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 Commercial Interiors rating system, draft credit language was developed with a requirement to avoid PVC materials. The US Green Building Council decided that further technical knowledge was needed to determine the soundness of this requirement. Pending resolution of this issue, all related Credit Interpretation Rulings and potential Innovation and Design credits were put on hold.

This report is supplemented 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.

1.1 Task Group Charge

The U.S. Green Building Council's Technical & Scientific Advisory Committee has 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."

1.2 Background

The formation of the Task Group was precipitated by a series of events beginning in 1999. In December of that year, a meeting was held at the Pocantico retreat center in Tarrytown, New York, at which the 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 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 LEED Technical & Scientific Advisory Committee, 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 extensive 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. These individuals also exhibited an unbiased and objective approach to the charge. Each of the Task Group members has been approved by the USGBC Board of Directors. The Task Group includes the following members:

Scot Horst, Chair	The Athena Institute
Kara Altshuler, PhD	ICF Consulting
Nadav Malin	BuildingGreen, Inc.
Greg Norris, PhD	Harvard University School of Public Health, Sylvatica

The Task Group was assisted greatly by Yurika Nishioka, PhD, Harvard University School of Public Health.

1.3 Objectives

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

Task Group: The Task Group is not aiming to definitively resolve all issues relating to PVC or competing materials. Rather, it is the objective of the Task Group to provide a report to TSAC that presents 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.

1.4 Scope

Understanding the environmental pros and cons relating to PVC building materials and competing alternative materials is a mammoth undertaking. In order to prepare a timely report, 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 listed here:

1.4.1 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 create a level playing field for the review of PVC materials and products with alternatives, and in order to avoid the need for an historical review of all pros and cons, only current manufacturing practices are considered.

1.4.2 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.

1.4.3 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 small number of representative applications were selected for analysis. These are siding, drain/waste/vent pipe, resilient flooring, and windows. Potential implications of this streamlining measure are discussed below in Section 2.6.

1.4.4 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-based and non-PVC building materials. Morbidity, however, is defined as “the presence or incidence of disease,” but does not provide an indication of the adversity of one disease compared to another. For example, many of the compounds used to manufacture the 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 windows, is associated with respiratory diseases and cancers of the lung and bladder.

Concern has also been raised regarding morbidity in end users of these building materials. Indoor air quality is increasingly being raised as an important factor to human health. Several factors contribute to reduced indoor air quality. These include biological contaminants such as molds, viruses, bacteria, dust mite allergen, animal dander and pollen; chemical pollutants including tobacco smoke, emissions from office equipment, furniture, and building materials; and particles such as dust and dirt

The most important type of morbidity is cancer, as it more likely results in the affected individual's death. Cancer risk has been assessed for the occupational workers in the manufacture of PVC building materials and the alternatives, as well as in the installation and end use of all flooring materials evaluated. Further, cancer incidence rates and death rates for Louisiana residents, including parish-specific rates have been evaluated (See Section 3.4). Air monitoring data for carcinogens emitted by vinyl manufacturers in Louisiana and Kentucky have also been reviewed in relation to their potential to cause cancer in the general population (Section 3.4).

Not all of the building materials analyzed are significant contributors to the development of cancer, because they do not involve or emit cancer-causing agents, or because exposures to carcinogens during one or more life cycle stages have been minimized or controlled. Therefore, types of non-cancer morbidity increase in importance. These other types include those discussed above and others. It is difficult to make comparisons between the building materials in terms of the types of morbidity they induce because many occupational exposures involve multiple chemicals, dose-response data are inadequate, or the data across studies represent endpoints that are not directly comparable.

This report addresses morbidity in general terms in Section 3.4, but does not attempt to make quantitative comparisons regarding the various contributions toward morbidity from the various building materials.

1.5 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. *The distribution of this draft report represents step seven in the 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 and shows the specific topic to which each source contributes.

1.6 Report Structure

This report is intended to function as a decision support tool for the LEED Steering Committee as to 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." 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 five main sections, supplemented by appendixes and an online database.

The rest of the report is organized as follows:

Section 2 defines report scope and methodology. 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 provides a review of different types of evidence. Section 3.1 is a discussion of the availability of evidence and reports that provide information that is not quantitative in nature. Section 3.2 is a quantitative analysis of life cycle and risk related evidence. This section specifically 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 (where relevant) the usage phase of the product alternatives.

Section 4 contains discussion of data gaps and information that the Task Group considers important but not covered in the quantitative analysis in Section 3.

Section 5 contains conclusions that the Task Group has made from the evidence presented.

Appendix A Acronyms

Appendix B Life Cycle Analysis Modeling Assumptions

Appendix C References

Appendix D Toxicants Assessed in the Human Health Risk Assessment

Appendix E Task Group Biographies

2 Methodology

For this report, the Task Group has mapped the current literature on PVC-based materials and alternatives to assist the LEED Steering Committee in their decision making. The work began with the creation of a reference table and a matrix of life cycle considerations. These two documents have become a relational database that is publicly available for review and consideration (http://pvc.buildinggreen.com). The literature includes many types of reports and stakeholder documents that range from studies to opinion pieces. The literature has been reviewed, compiled and synthesized in an integrative assessment using a life cycle framework.

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.

2.1 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.

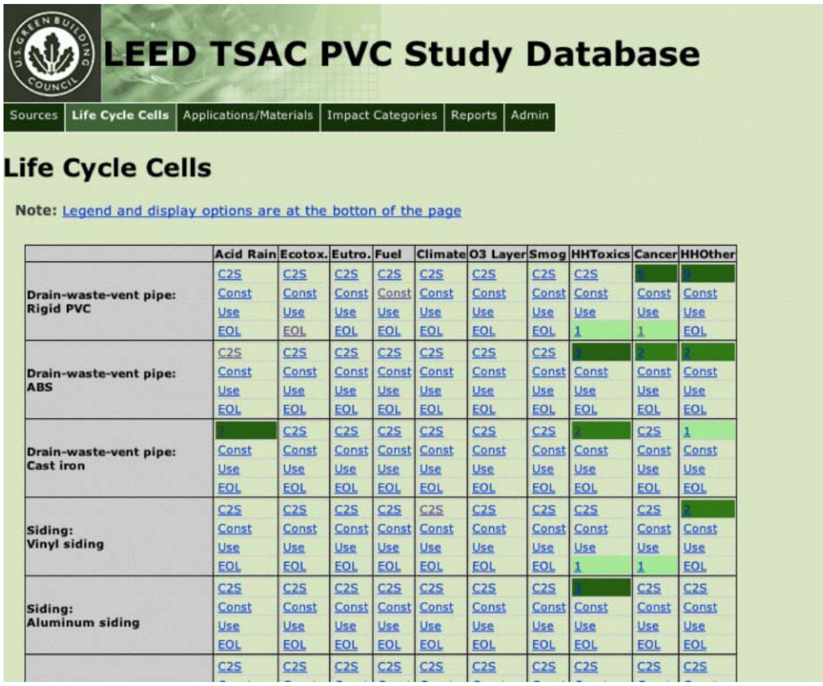

Figure 1. Life Cycle Cells page of the PVC Study Database

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.

2.2 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 C.) 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.

2.2.1 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 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.

2.2.2 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.

2.3 Life cycle assessment framework

The LCA framework refers to the manner in which the literature has been organized. The framework ensures that all phases of product life cycles are considered, that a wide set of environmental and health impacts are considered and that products are compared on the basis of options that deliver an equivalent function to the user.

When a report or study has information that relates to one or more materials, impact categories, and/or life cycle stages, that report is linked by the corresponding cells in the database. Reports found in Section 3 reflect these links. Decision makers and stakeholders are also welcome to use the database, which can be accessed at http://pvc.buildinggreen.com for further review and reporting.

The impact categories chosen include those listed in EPA's Tool for the Reduction and Assessment of Chemical and Other Environmental Impacts (TRACI) method as well as others considered important to stakeholders (Bare et al., 2003). The impact categories and their units are:

Environment/Resource

- Acidification (+H millimoles)
- Ecotoxicity (g 2,4-D-equivalent)
- Eutrophication (g N-equivalent)
- Fossil Fuel depletion (Surplus MJ)

Combined human and ecosystem health

- Ozone Depletion (g CFC-11 equivalent)
- Global Warming (kg CO_2 equivalent)
- Photochemical Smog (g NOx equivalent)

Strictly human health

- Human Health: Particulates – HHPM (microDALY)
- Human Health: Cancer – HHC (microDALY)
- Human Health: Other – HHO (hazard index)

2.3.1 Assumptions for upstream impact analysis

The cradle-to-site impacts of each building material or product were estimated using the methodology and data sources of process-based life cycle inventory (LCI) analysis. Process-based LCI models describe the production supply chains of products as a total system of unit processes, interconnected 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™ was used to calculate life cycle impacts of various products based on its internal LCI data and characterization factors (PRé Consultants, 2004). The information and assumptions in SimaPro were checked and enhanced with information from the U.S. Department of Commerce's Building for Environmental and Economic Sustainability (BEES) software (NIST, 2002). With the exception of cancer and non-cancer impacts, TRACI characterization factors were used in the impact analysis.

Cancer risk was evaluated as a product of dose and potency (or cancer potency factors). For the cancer potency factors, values published by the USEPA and California EPA (Cal/EPA) were primarily used (Hertwich EG, 2001). For the dose calculations, intake fractions developed by Debbie Bennett at the Harvard School of Public Health were used (Bennett DH et al., 2003). 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.

Absolute cancer risk has been estimated for exposures during manufacturing, installation/construction and product use. TRACI already provides estimates of absolute risk from inhalation of primary and secondary particulate matter. The use of intake fraction estimates enables the estimation of *absolute* cancer risk from life cycle releases to the environment. Both the cancer risk from environmental pathways and the cancer risk through indoor exposures can be converted to estimates of life years lost (see section 2.3.2). These estimates of lost life years can then be compared and aggregated with the mortality and morbidity impacts of particulate inhalation, 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.

Note that the human health particulates and human health cancer impact categories for LCA releases can be, and were, integrated with occupational risk assessments for cancer, since mortality is the common endpoint. In contrast, the human health non-cancer impact category is highly variable and non-specific in terms of its endpoint; the resulting "hazard indices" reflect the ratio of estimated doses over no-effect doses, but the effect of interest is varied, and is not the only cause for concern regarding chronic exposures, especially at doses appreciably higher than the no-effect dose. Switching from the TRACI approach to non-cancer impacts, to an intake fraction-based approach, enables direct comparison and integration of the pollution-based (population risk) impacts with the indoor exposure-based (occupational risk) analyses.

As discussed later in Chapter 3, the newly-achieved and demonstrated ability to integrate occupational 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 very noteworthy conclusions for the LCAs and risk studies reported here. It is found that the occupational impacts are not at all trivial by comparison with the population risk impacts in these life cycles. These findings indicate that occupational exposures should not be ignored in LCAs in the future.

2.3.2 Description of modeling methods

Using CalTOX™, a multimedia total exposure model, Bennett has modeled intake fractions of 308 chemicals for an average person in the U.S.[1] In her model, population intake fraction is calculated as follows:

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[1] Bennett et al. (2000) 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

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

Using the population intake fractions, a cancer risk associated with X mg of pollutant *i* emitted over a 50-year period is calculated as follows:

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

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*.

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

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

where,

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

For cancer and non-cancer characterizations, TRACI uses human toxicity potentials (HTPs) developed by Edgar Hertwich. 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 do not include metals in our human health impact analysis for the general population. Although HTP values are available for metals, there are large uncertainties associated with the exposure levels of metals using a fugacity model.

To compare the magnitude of cancer risks with the particulate matter-related risks, cancer risk was converted into disability-adjusted life years (DALYs), which is the sum gf 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

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1,2,3,4,7,8-HEXACHLORINATED DIBENZOFURAN, which is not available in Bennett (2000). 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.

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 P, 1998). Using his figure, the total DALYs associated with the upstream emissions of air toxics are estimated as the product of the total cancer risk and 13.1 (DALYs per cancer incidence).

For the non-carcinogenic effects, the information on the disease categories associated with a pollutant are 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.

2.3.3 Treatment of chemical groups

The TRACI as well as the *iF*-based characterization factors 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 and phthalates.

With the exception of human health cancer and non-cancer characterization factors for PAHs, 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. Although this assumption may not be realistic, 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,
- **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
- **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 (1996)" 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 includes 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 (1996). These assumptions were carried through the analysis of human health cancer and non-cancer impacts, contributing to the variability of the impact estimates.

2.4 Data gap identification and significance assessment

When possible, values have been extracted from the studies to give numerical values to life cycle and risk considerations. When values are not obtainable, they are listed as data gaps. In such instances, this report flags gaps in existing knowledge with the understanding that missing knowledge does not mean that impacts, either positive or negative, are not occurring. These flags might provide a guide for future research, but are also potential clues for information that affects the charge of the Task Group, specifically as it relates to the availability of evidence required to make a reasoned decision. The data gaps so identified are summarized in Section 4 of this report.

2.5 Risk Assessment

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. Hazard identification is a procedure in which the chemical compounds present at the location are analyzed for their contribution to health risk based on their incidence and frequency. Dose-response assessment involves the determination of the relation between the magnitude of a human's exposure to the compound and the likelihood of a resultant adverse health effect. Exposure assessment is the determination of relevant exposure pathways and potentially exposed groups of individuals, and the quantification (when possible) of potential exposure. Risk characterization is the estimation of potential systemic (non-cancerous) and cancer risks associated with exposure to the compounds under consideration. 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.

2.5.1 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., *Environmental Resource Guide*, 1997; 1998), published literature, and on-line data from manufacturers of these products.

2.5.2 Dose-response assessment

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. Two different types of dose-response relationships are commonly associated with exposure to chemicals at high doses or to environmental contaminants: 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. The U.S. EPA and other organizations have developed Reference Doses (RfDs) 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 associated 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). This CSF 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 cancer risk, particularly, are being recognized, however (Ward et al., 2003). Data are limited with regard to adequately preventing dermal exposures; the use of whole-body protection and/or gloves to mitigate dermal exposures may also be associated with certain adverse effects such as heat stress and dermal reactions. In addition, there is a general lack of data on the ability of these personal protective measures to prevent dermal exposure (Ward et al., 2003).

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 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.

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 data were unavailable from that source, values from a recent version of the U.S. EPA's Health Effects Assessment Summary Tables (U.S. EPA, 1997), 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 toxicity data (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; eleven compounds are known or believed to be carcinogenic. These are vinyl chloride monomer, ethylene dichloride, di(ethylhexyl)phthalate (DEHP), cadmium, 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.

Discussion of General Population Exposure Limits for individual Compounds

DEHP--Non-cancer

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 (IRIS, 2004 for DEHP). 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; NTP-CERHR, 2000). 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.

DEHP-Cancer

DEHP has been shown to be a liver carcinogen in rats and mice. Extensive data has been published regarding the mechanism of liver carcinogenesis of DEHP in the rodent model. The mechanism has been shown to be mediated through peroxisomal proliferation in the liver (specifically via activation of peroxisomal proliferator-activated receptor alpha, PPAR-α), a phenomenon which does not happen to a measurable degree in the human (Doull et al., 1999; NTP-CERHR, 2000). 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 majority of the scientific community does not consider DEHP to be a human carcinogen. For the purposes of this assessment, DEHP has been considered as non-carcinogenic in the human.

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 (typically 10-100). In other words, the exposure limit established is 10 to 100-fold lower than the lowest exposure dose found to not 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 PEL 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. Exceedances of OSHA PELs, as well as ACGIH TLVs, however, do occur and have been documented. Therefore, risk estimates developed in this report have relied solely on OSHA PELs in the absence of more accurate exposure data, although it is clear that in many occupational situations these PELs may be exceeded frequently.

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. To provide an upper bound estimate of risk, particularly when exposure data for an industry are missing, OSHA PELs were used as starting points for the risk estimates. These assumed exposure concentrations will be high for those industries that use ACGIH TLVs as exposure limits or for those processes that have exposure controls (e.g., increased ventilation, emission and dust control mechanisms, etc.) in place. 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. Cessation of exposure following work hours and on weekends allows for the recovery of the exposed individual, particularly from central nervous system effects that are typically caused by many organic solvents (e.g., headache, dizziness, tingling). 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 that use this same methodology, such as residents or those who experience incidental 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 are extremely limited. 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.

2.5.3 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.

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 many cases a lack of accurate exposure data prevents the estimation of potential health risks from these exposures. For example, it is believed that 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 and an inhalation rate of 1.3 m^3/hr (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). These exposure values are informed by U.S. EPA's Exposure Factors Handbook (U.S. EPA, 1996) and using professional judgment. The assumed body weight is slightly less than currently recommended by U.S. EPA (1996), but because the toxicity values have been derived based on a 70 kg average 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. The tenure of the occupational worker is informed by recent data from 2002 from the Bureau of Labor Statistics which shows 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. Exposure values for this receptor 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. Due to the various manufacturing times of the materials, and the conditioning period to which flooring samples were exposed for this study, it is likely that the actual measured concentrations of off-gassed VOCs are lower than those to which installers are actually exposed. However, the published data do not provide any basis for an adjustment by which these values might be adjusted upward to more accurately reflect installation conditions.

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).

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 m3/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, 1996). The body weight represents an average between male and female students in this age range. Exposure data from the California emissions study (CIWMB, 2003) and from Horn et al. (1998) were also used for this receptor.

As the CIWMB emissions study was primarily concerned with indoor air quality, the laboratory analyses focused only on VOCs that were emitted. 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. Although this is a gap in the overall assessment, it is not considered to be a significant one for the following reasons. Clausen and coworkers (2004) studied DEHP migration from vinyl flooring and its uptake into dust over a 1.3-year period; they reported steady-state DEHP levels of 1 μg/m3. Using the default exposure values for teenagers discussed above, this exposure concentration is equivalent to an average daily dose for one year (ADDyear) of 3.08E-05 mg/kg-day. Comparison of this value to the current RfD for DEHP gives an HI of 0.0015, which is negligible. (A revised RfD would have to decrease by three orders of magnitude for this exposure to significantly increase a teenager's non-cancer risk.) By contrast, exposure of an infant to DEHP in an intensive care medical situation may be as high as 0.38 mg/kg-day (averaged over one year; based on data from Rais-Bahrami et al., 2004). For more discussion on morbidity from exposure to phthalates and susceptible subpopulations, see Section 3.4.

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 2.

Table 2. 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

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.

2.5.4 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.

Formulas

Table 3. 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(sub)	=	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 3 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 at this time to assume 100% absorption for each inhaled compound.

Estimates of Systemic Health Hazards and Cancer Risks

Estimated doses and resultant risk estimates for systemic and cancer endpoints are presented in the same format for each exposure scenario in tables found in Appendix C. In general, the EPCs for each receptor are repeated at the beginning of the table. The appropriate ADD(day) value for each pathway and life stage of the receptor (if applicable) is then presented. Following this information are estimated systemic risks in the form of HI values and cancer risks in the form of ILCR values. The HI and ILCR values are summed for each relevant exposure pathway. At the end of the table the site total HI and ILCR values are presented, along with the proportion of each risk value contributed by each compound. The last page of each table contains the relevant exposure parameters and conversion factors applicable to the receptor. These values are then normalized to the functional unit of the building material, based on the amount of material produced per year, the cost of the manufacturing labor time (person work hour per $) and the manufacturing cost of the item per ($ per functional unit). These normalized risk values are presented in a summary table in Section 3.

2.5.5 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. According to OSHA regulations, when ambient concentrations for toxicants that equal half the OSHA PEL level are reached in an occupational environment (a value called 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. Therefore, using OSHA PELs as high-end exposure values affords a level of security in potentially overestimating risk.

The risk assessment erred on the side of inclusiveness, including any chemicals we thought would 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.

2.5.6 Limitations

Reasonable care has been used in the performance of all analyses in this report. The risk assessment conclusions contained herein represent the professional interpretation of the TSAC PVC Task Group. The conclusions of the risk assessment are to be interpreted within the context of inherent limitations in the data available at the time it was developed and current analysis techniques.

2.6 Normalization

In Chapter 3, results of the integrated LCA and Risk Assessment of impacts associated with each product group are normalized in two ways.

First, the results are internally normalized within each impact category, with respect to the central value of LCA results for the PVC alternative for that impact category. The normalized values are unit-less, which is the ratio of an LCA result to the central value of the PVC alternative. This internal normalization allows a comparison of the range of impacts relative to the PVC alternative within each impact category. However, internal normalization does not allow a comparison of impacts across impact categories. For this, use of an “external” reference system as the normalization factor is helpful. An “external” normalization method sheds light on the relative influence of the product choice upon the total annual impacts of the reference system for each impact category. In this analysis we use the total annual per U.S. capita flows as the basis for normalization.

2.7 Materials and Alternatives

In its task of assessing the availability and quality of evidence concerning a PVC-related credit in LEED, the Task Group set out to compare PVC-based building materials with typical competing materials for various applications. Through this exercise the Task Group hoped to learn if it was possible to document any clear trends in terms of environmental preferability between the PVC and non-PVC materials.

When the different applications for PVC-based building materials were viewed, it became apparent that it was not feasible, with the Task Group’s available time and resources, to undertake a comprehensive comparison of the various options for all applications. A choice was made to identify a small number of categories for analysis that would be representative. In selecting these categories, the ones that represented the largest quantities of PVC use in buildings were of selected. Likewise, it was important to include a range of material and product types, including those intended for both interior and exterior and use, both rigid and flexible materials.

The Task Group recognized that it was making a compromise by focusing on a representative set of applications and materials rather than undertaking a comprehensive survey. The expectation was that, if a clear trend was established showing that either the PVC or the non-PVC materials were environmentally preferable, that trend would most likely have to be corroborated by investigating additional applications. If no trend was found, however, it would be unnecessary to study a broader set of applications.

The applications selected for study were siding, drain-waste-vent pipe, resilient flooring, and windows. 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. The specific materials studied for each of these applications are described briefly below. Detailed modeling assumptions for each material are provided in Appendix B.

2.7.1 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.

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 and wood battens and 70 years for fiber-cement.

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.

Aluminum Siding

Aluminum siding is assumed to consist of 99% aluminum and 1% PVC coating. 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.

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. Wood siding is occasionally left unfinished, but is typically coated with a clear, semitransparent, or opaque coating. Each of these differ in formulation, but the products in use today are primarily water-based. The available European life cycle inventory sources only contain information about alkyd (oil-based) paint, however. Due to the lack of appropriate information, primer and stain is assumed only for the analysis in BEES. Installation waste of 5% is assumed.

Fiber-cement Siding

For lack of data on fiber cement siding, fiber-cement roofing was modeled for this comparison. This substitution was checked using internally comparable Swiss data, which showed that ecological impacts for fiber cement roof slate tends to fall *in between those of* the two other options, fiber cement corrugated slab (lowest impact for all but one essential tie) and fiber cement facing tile (always highest impact). A range of components in fiber cement siding was modeled. As an upper bound, the following ratio of constituents was modeled: the siding is assumed to consist of Portland cement (90%), sand as filler (5%), and wood chips as organic fiber (5%). Galvanized steel nails were assumed for the installation, with a 5% installation waste. As with wood siding, no paint is assumed. Most fiber-cement products require paint, so this is clearly an underestimation of impacts. A 70-year useful life was assumed. The mass of the material is 1.79 kilogram per functional unit.

2.7.2 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 delivery distance to the site (during the installation phase) is assumed to be 500 miles (805 km) for all materials. The life span of the pipe products has been assumed to be 50 years. All three pipe materials will typically last longer than 50 years, but there was no information by which to determine a significant difference in lifespan between them.

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

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. At this point, health and environmental impacts associated with the solvent cement used for installation is not included in the analysis, but it recognized that exposure to this cement could represent a health risk to plumbers. Installation waste is also not accounted for.

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. As with ABS, solvent cements and installation waste were not included in our analysis.

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. As with the other piping materials, neither installation processes and materials nor installation waste were included in the analysis.

2.7.3 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. The delivery distance to the site (during the installation phase) is assumed to be 500 miles (805 km) for all materials.

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. 5% process and installation waste was assumed.

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. Styrene butadiene flooring adhesive was included in the models, as was an installation waste factor of about 1.6%.

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%.

Cork

Cork flooring consists primarily of cork waste from wine stopper production with a binder. The raw material transportation from Portugal is included. 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 for the adhesive was assumed, but no installation waste for the flooring itself.

2.7.4 Windows

There is a wide variety of window types available, including many that combine wood with PVC or aluminum (as exterior cladding). To keep the task manageable, the comparison was limited to windows that have frame materials made entirely from PVC, aluminum, or wood. The functional unit that was used as a basis for comparing these windows is a two panel Swiss window of 1.3 meters by 1.65 meters with 23 square foot in area, over a 50-year service life. All three window types were modeled based on the life cycle inventories from LCExplorer (Norris et al, 2002).

Vinyl Windows (assumptions for vinyl and aluminum windows)

The mass of vinyl windows is estimated at 2.0 kilograms per functional unit. The primary framing components are PVC (60%) and galvanized steel (32%). The remaining components include steel stainless (3.6%) for the fittings, EPDM (1.7%), PVC-NBR (1.1%), aluminum (0.9%) and zinc (0.3%) for the framing, and screws (0.3%).

Aluminum Window

The mass of aluminum windows is estimated at 1.8 kilograms per functional unit. The components of aluminum windows include aluminum (70%), polyamide glass fiber (12%) and EPDM (7%) for the framing, as well as zinc (5.3%), stainless steel (0.9%), aluminum casting (1.6%) and galvanized steel (0.5%) for the fitting with the remainder including HDPE, epoxy resin and brass.

Wood Windows

The mass of wood windows is estimated at 2.1 kilograms per functional unit. Wood windows consist primarily of the framing containing timber (83%), aluminum (2.9%), EPDM (2.1%) and silicone (0.8%). The remainder includes the fitting components such as steel sheet (3.7%), zinc (0.3%), and screws (0.1%) as well as the surface coating such as acrylate top coating (3.4%), primer (1%), and polyester powder (0.1%). There is also a small portion of spruce strips, polyvinyl acetate adhesive, and beach wood.

For the usage phase, the assumption is that wood frames are painted every 9 years on the exterior surface and 7 years on the interior surface. Based on the Athena model, the amount of oil-based paint used on the wood window framing for each application is assumed to be 0.38 kilogram per linear foot for previously unpainted surfaces and 0.32 kilogram per linear foot for painted surfaces.

3 Evidence

3.1 Characterization of evidence

The evidence used by the Task Group in the compilation of this report is described and characterized here, along with detailed analyses of both the qualitative and quantitative information that was used.

3.1.1 Sources in the database

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 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.

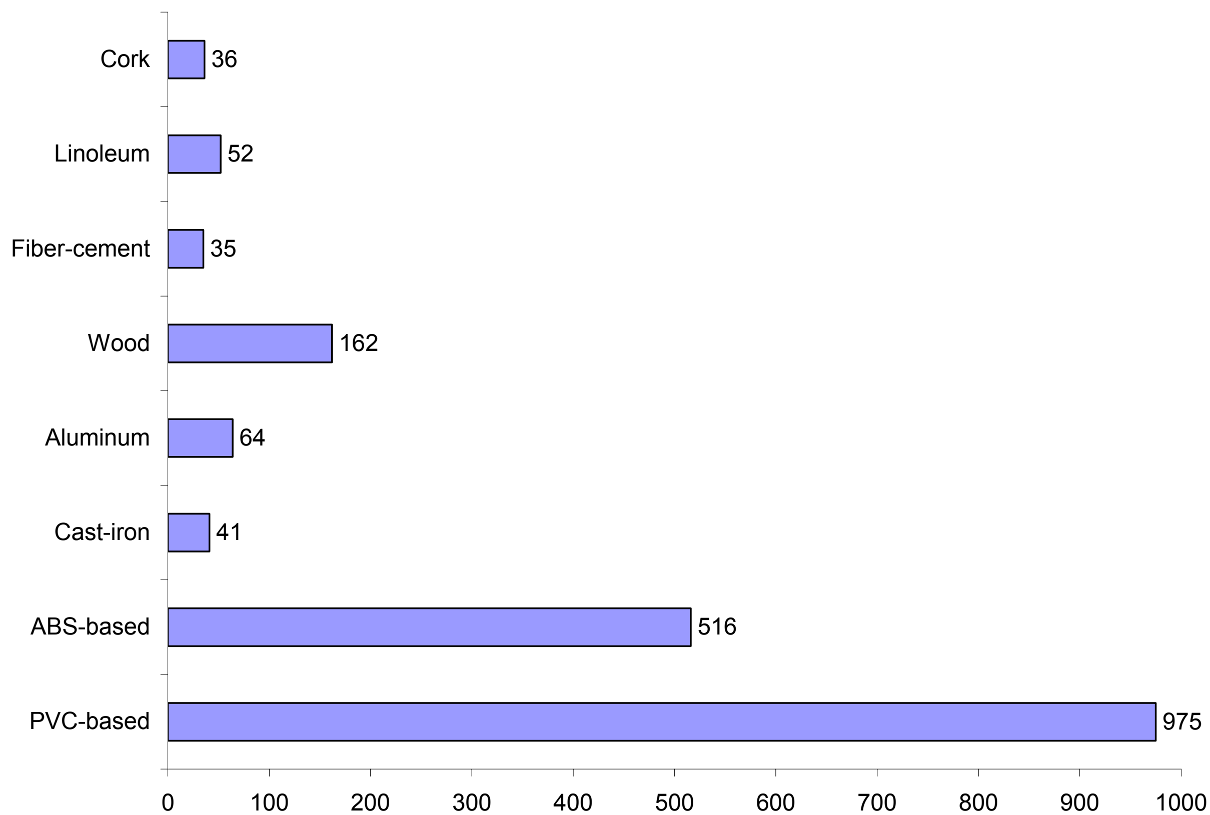

Figure 2. Information sources by material

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 3.

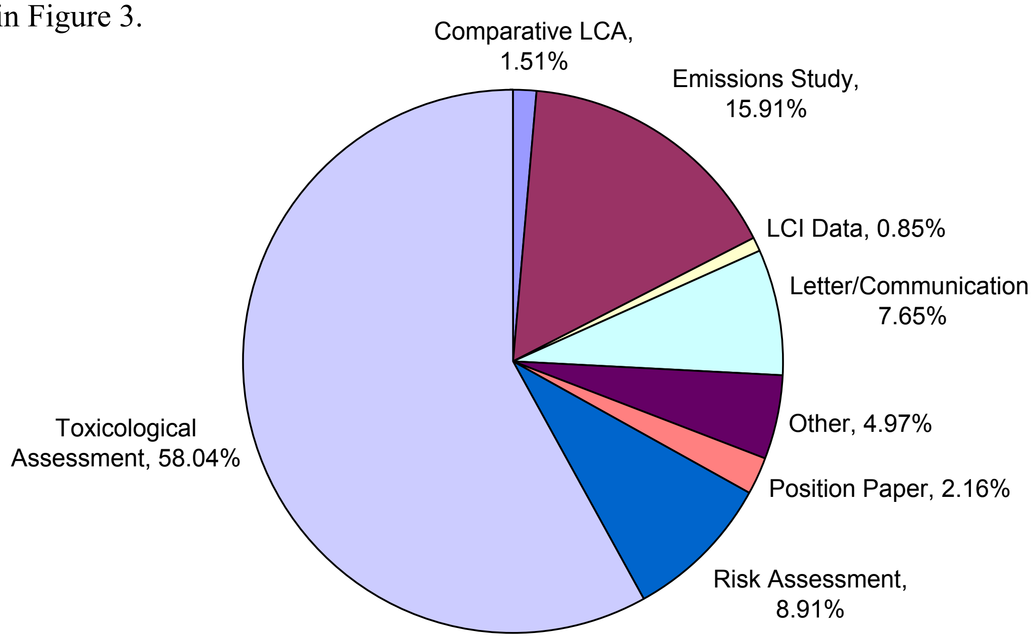

Figure 3. Sources by type

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.

3.1.2 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 4). Of the PVC-related submissions, more than half (~58-59%) are related to toxicology (see Figure 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 (NPT-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.

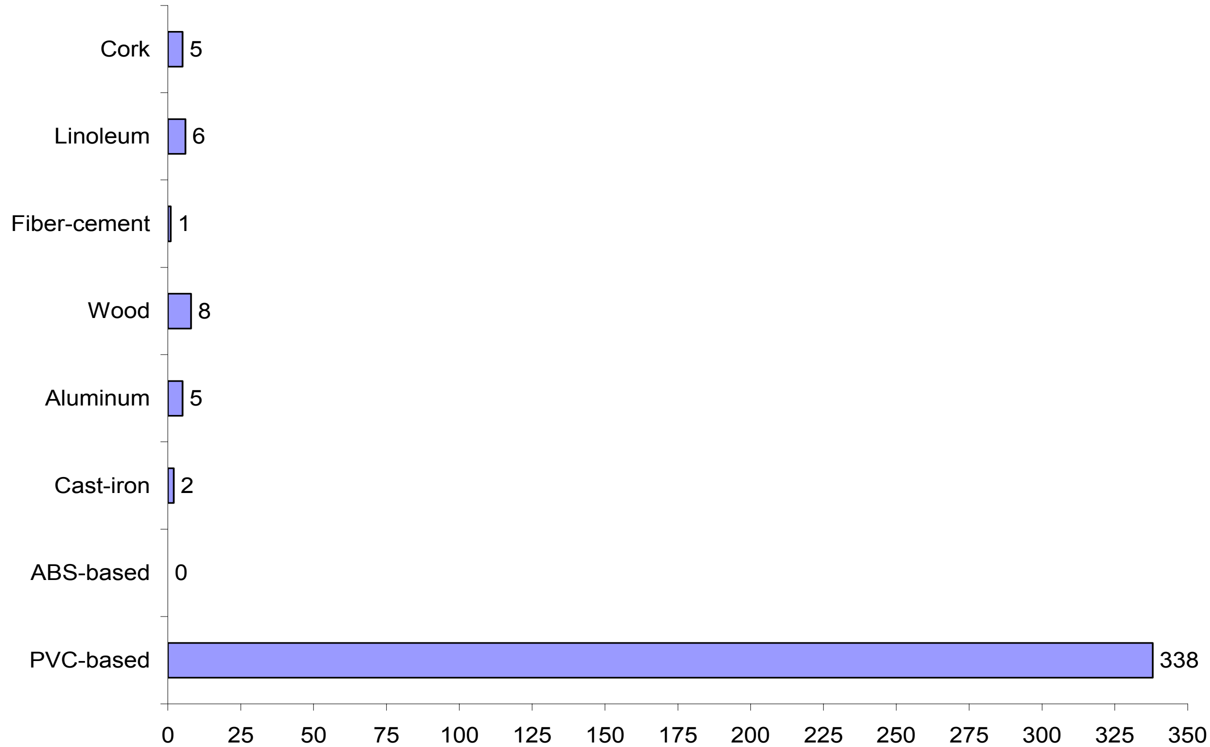

Figure 4. Stakeholder submissions by material

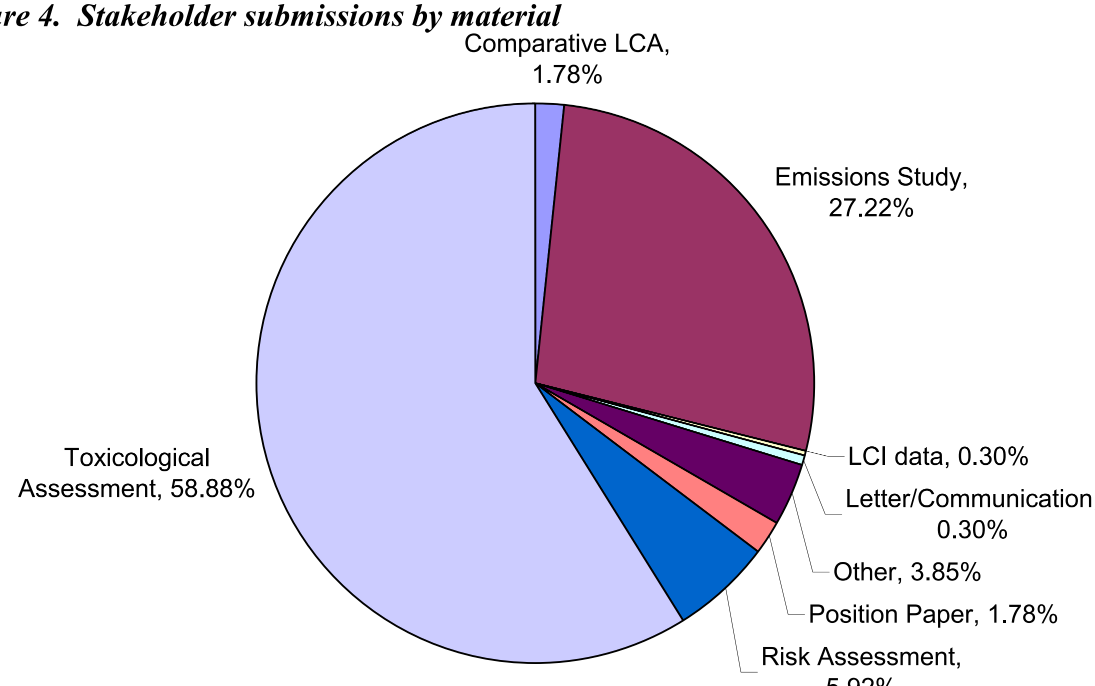

Figure 5. PVC stakeholder submissions by type

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.

3.2 Qualitative evidence

Sources in the database include a huge amount of information that, for various reasons, could not be used in a quantitative analysis. Some key points that emerged from that information are described here.

3.2.1 Non-quantifiable effects of key pollutants

Many of the sources on VCM documented its ability to induce cancer of the liver and other organs, which are quantified below, but several investigated the relationship between exposure to vinyl resin and respiratory effects in exposed humans, such as asthma and pulmonary disease, as well as allergic responses. The majority of articles on phthalates dealt with liver cancer in rodent models and peroxisome proliferation, as this area of research has been widely investigated. A handful focused on endocrine effects; the scientific articles on this topic tended to represent studies using animal models, while opinions and position papers also involved the extrapolation of these effects to reproductive endpoints in humans exposed to these compounds. Many extensive reviews of different phthalate esters by NTP and other federal agencies are also included in the database.

Many of these sources provide quantitative data regarding exposure of different human population groups to compounds in PVC and/or ABS. Other studies provide insight into subcellular changes caused by the compounds in the building materials, such as genotoxicity, effects on apoptosis and cell signaling. These articles inform theories on mechanism of action of toxicity and suggest future areas of research. Additional data regarding biomarkers of exposure in humans provide information on distribution of these compounds and potential pathways of exposure.

In general, the articles on phthalates indicate that ingestion or intravenous exposure to these compounds results in adverse effects to the liver and the reproductive organs in animals (reviewed in NTP-CERHR, 2000). Parent DEHP is not the primary toxicant in humans; rather it is the monoethyl hexyl phthalate (MEHP) metabolite of DEHP which is believed to mediate the adverse effects observed in animal studies. DEHP is primarily metabolized by lipases in the gut. Other cells in the body can also metabolize DEHP to MEHP, but it occurs more slowly (NTP-CERHR, 2000). Although primary exposure of the general population is believed to occur through the diet (following leaching of DEHP from plastics involved in the storage and preparation of food; NTP-CERHR, 2000), individuals are also exposed to DEHP via inhalation of particles from a variety of consumer products including flooring materials, as well as via intravenous exposure from medical plastics such as intravenous tubing and blood bags. DEHP introduced into the circulation can be metabolized by enzymes present in the plasma component of the bloodstream.

Studies in animals indicate that rodents develop adverse toxicological effects from DEHP exposure that occurs at high concentrations, e.g., with most studies having NOAELs at ~150 mg/kg/day or higher. The exception to this is rodent developmental studies with NOAELs ranging from 3.7-14 mg/kg/day (NTP-CERHR, 2000). Nevertheless, significant differences in metabolism across species indicates that humans may not metabolize DEHP (or other phthalates) as readily as do rodents. For example, young adult marmosets, which are primates, showed no testicular effects following 13 weeks of oral DEHP doses at 2,500 mg/kg/day, in contrast to effects in other animal models such as rats, mice, guinea pigs, and ferrets. Pharmacokinetic data indicate that metabolic processes in humans and other primates may saturate at lower concentrations than do those of rodents, indicating that blood levels of phthalate metabolites in humans may never reach the levels attained in rodents that are associated with the severe adverse effects (NTP-CERHR, 2000).

Individuals receiving intensive care in hospitals including parenteral nutrition, dialysis, iv fluids, and blood transfusions may receive large doses of DEHP from leaching of the flexible plastics. Of particular concern are infants and young children exposed in this manner. Rais-Bahrami and coauthors (2004) evaluated pubertal development in adolescents who as neonates were exposed to DEHP via extracorporeal membrane oxygenation (ECMO) support. It was estimated that the infants may have been exposed to a range of DEHP from 42-140 mg/kg over a treatment period of 3-10 days (although another author estimates the dose range as non-detectable to 34.9 mg/kg/treatment period). Thyroid, liver, and kidney function were measured using standard serum enzyme analyses and were determined to be normal. Reproductive hormones (luteinizing hormone, follicle-stimulating hormone, estradiol, and testosterone) were within normal ranges, as were testicular volume and phallic length. These data indicate that high exposures to DEHP in infants do not result in adverse effects to organ function and parameters of sexual maturity. This study did not address effects on the Sertoli cell, however, which is a target cell for DEHP. Additional studies on sperm motility and condition in adults who were exposed as infants to DEHP may help elucidate the sensitivity of humans to exposure to this compound. The data currently suggest that humans are not as sensitive to the reproductive effects of DEHP exposure as rodents.

3.2.2 Backyard burning

Burning of household trash in the rural United States is recognized as a significant source of exposure to PBTs, particularly dioxins and furans. The U.S. EPA estimates that 40% of rural Americans burn their trash, based on phone surveys taken in five central Illinois counties (Two Rivers Region Council of Public Officials and Patrick Engineering, 1994, as reported in U.S. EPA, 2000). These data were consistent with questionnaires given to 760 residents of northeast Minnesota and northwestern Michigan. Although the majority of residents reported that they had garbage hauling service, 27.5% reported using burn barrels to burn household garbage and other materials. Of those individuals who admitted burning trash, 39% did so weekly and 30% did so once or twice per month (Zenith, 2000, as reported in U.S. EPA, 2000).

Laboratory burn estimates for emissions of dioxins and furans using representative household waste (based on New York State residents who do not typically recycle; Lemieux, 1997) 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. In fact, according to Lemieux's 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 (Lemieux, 1997).

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. Statistical analyses on the emissions data indicated that knowledge of temperature trends (burn rate parameters) in the test burns allowed the best predictions for emissions of total TEQs. These data suggest that it would be very difficult to determine which factors of barrel burning (e.g., waste orientation and/or composition, and burning practices) in a particular rural location were contributing most significantly to total dioxin/furan emissions. Clearly, reducing PVC content in household waste would reduce dioxin and furan emissions, but that change alone would not eliminate emissions, and it is unclear how significant the effect would be.

The U.S. EPA (2000) estimated annual nationwide dioxin emissions from backyard burning in its Draft Dioxin Reassessment using the following assumptions:

- ❖ Of the rural population in the U.S., 40% are assumed to burn their household waste in a barrel;
- ❖ Each citizen generates 3.72 pounds of solid waste, on average per day (616 kg/person-yr);
- ❖ Of households disposing of waste by burning, ~63% of it is burned (e.g., 388 kg/person-yr);
- ❖ In 1994 (data that were used for 1995 reference year), 52.7 million people lived in non-metropolitan areas (U.S. DOC, 1997, as reported in U.S. EPA, 2000).

Annual nationwide emissions from this source were estimated using the following equation:

$$E_{TEQ} = EF_{TEQ} \times P \times F \times W$$

Where,

E_{TEQ} = Annual TEQ_{DF} emissions (g/yr)

EF_{TEQ} = TEQ_{DF} emission factor (g TEQ_{DF}/kg of waste)

P = Non-metropolitan population of U.S. in reference year (1995)

F = Fraction of non-metropolitan population assumed to burn household waste (0.4)

W = Mass of household waste burned/year per capita (388 kg/person-yr).

Estimated nationwide emissions for 1995 were 595 g I-TEQ_{DF}.

These values are an order of magnitude higher than emissions calculated for PVC and EDC/VCM manufacturing facilities, produced by Vinyl Institute (1998) and audited and agreed to by the U.S. EPA (2000). For example, estimates of total TEQ_{DF} by the Vinyl Institute for the 1995 reference year are the following (based on VCM, EDC, resin, and PVC products produced in that year):

Wastewater solids:	12.10 g I-TEQ_{DF}
Wastewater:	0.32 g I-TEQ_{DF}
Stack gas:	31 g I-TEQ_{DF}
Products:	3.83 g I-TEQ_{DF}

Total equivalents sum to 47.25 g I-TEQ_{DF}

Data published by the Great Lakes Binational Toxics Strategy's Burn Barrel Subgroup (GLBTS, 2004) indicates that >90% of burned materials are paper and cardboard wastes. Therefore, it seems prudent to conduct outreach activities focused on increasing paper and cardboard recycling and persuading states to ban backyard and barrel burning in all counties. Currently, eight states (California, Connecticut, Maine, Massachusetts, New Hampshire, New Jersey, New Mexico, and Vermont) have enacted state-wide bans preventing barrel burning of household trash (ENN, 2004). New York may join this number (ENN, 2004), but is being resisted by farming groups. The state currently bars communities with 20,000 people or more from barrel burning. Similar laws are in force in other states.

It is important to note that according to the U.S. EPA's most recent Dioxin Inventory (2000), it cannot be assumed that the sources that make the largest contributions to overall dioxin emissions also make the same relative contributions to human exposure. For example, it is accepted that human exposures to dioxins are primarily via food products, dairy products and meat in particular. Cows and other domesticated feed animals bioconcentrate the PCDDs/PCDFs in their fatty tissue after ingesting plant materials that have dioxin residues from emission sources. Therefore, the geographic locations of the major U.S. providers of beef, pork, chicken, and fish and location downwind of dioxin sources should be considered for further analysis.

Backyard burning as a potential source of dioxin emissions was identified through the data gap in the 1994 Dioxins Emissions Inventory which showed a significant difference in total emissions in the U.S. and the exposure estimates for the U.S. population. Similarly, other potentially significant sources of emissions, such as iron sintering, are still being identified. Additional research on reducing total emissions, through command and control technology, is needed to ensure that exposure of the general public is minimized.

3.2.3 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. Markowitz (1989) and Markowitz et al. (1989) reported on a population of firefighters exposed to burning PVC in a 1985 fire. The retrospective cohort study surveyed 80 exposed firefighters and 15 nonexposed firefighters for their responses to an 81-symptom checklist. Both 5-6 weeks after the fire, as well as 22 months following the fire, the exposed subjects reported significantly more frequent and severe respiratory symptoms. At the 5-6 week timepoint, several of the symptoms reported were related to exposure to hydrogen chloride, while others were related to smoke inhalation or were psychosocial. At 22 months, ~18% of exposed firefighters reported receiving a diagnosis of asthma and/or bronchitis from a physician. This study may have suffered from an inadequate number of control subjects.

These studies conflict with two others regarding firefighters potentially exposed to PVC. Tashkin et al. (1977) reports on a group of 21 Los Angeles firefighters who were exposed to PVC combustion products in a fire; 19 were initially diagnosed with hypoxemia (deficiency of oxygen in arterial blood). Analyses on respiratory function performed one month following the exposure revealed no deficits in respiratory function and no reported respiratory symptoms, when compared to 20 non-firefighter controls, matched for age, sex, race, cigarette smoking, and geographic and climatic differences. Musk et al. (1982) reports that 951 white Boston firefighters were studied over 6 years (1970-1976) for respiratory function and other symptoms. While current cigarette smoking was associated with increased prevalence of bronchitis and other symptoms, longitudinal changes in respiratory function were not correlated with any exposures in the active firefighters. The study authors concluded that measures taken by the Boston Fire Department, including the use of protective respiratory apparatus, appeared to be protecting the firefighters.

A study of Croatian firefighters also suggests that exposure to environmental pollution may contribute to acute and chronic respiratory symptoms, but odds ratios for chronic respiratory symptoms were significant only for duration of work exposure and for smoking (Mustajbegovic et al., 2001).

Upshur et al. (2001) studied short-term effects in a community exposed to PVC combustion products from a 1997 fire in a plastics recycling plant, Plastimet, Inc., in Hamilton, Ontario. The plant contained several thousand tons of auto parts composed primarily of PVC plastics and polyurethane. According to a concomitant report written by the Ministry of Environment and Energy (1997), at least 400 tons of PVC were burned. The fire was large and difficult to control and lasted 80 hours (July 9-12, 1997). An atmospheric inversion occurred two days following the start of the fire, prompting the evacuation of the local community to minimize exposures to contaminants whose levels were increased in the immediate vicinity.

The study authors reviewed data from a health survey conducted in the affected community, as well as records of emergency room visits and hospital admissions for 14 days prior to and 12 days following the date of the fire in 1997 and the previous year. Recorded cases of chloracne reported to family physicians in the area of the fire were also tabulated. A total of 163 people responded to the questionnaire about themselves and 325 additional people, for a total of 488 residents. There were a total of 222 general concerns reported on the questionnaires, with the highest number of respondents concerned about eating garden produce (17%), the ash/dust fallout (13%), and air quality (12%). Only 4% had concerns about smoke, and 2% had concerns about safety. Approximately 62% of the respondents had no symptoms during the days of the fire, while 17% of the residents complained of throat irritation. Other symptoms reported in decreasing order of frequency were the following (percentages not reported): headache, breathing difficulty, nausea, abdominal pain, vomiting, diarrhea, eye irritation, dizziness/lightheadedness, mouth irritation, and nose irritation. Symptoms were most prevalent two days after the fire started and decreased subsequently. Only 2% of the affected individuals sought medical attention from local hospitals; local physicians were not surveyed. There were no notable increases in hospital visits or admissions during or immediately following the fire; further, no cases of chloracne were reported. No fatalities occurred from the fire in firefighters, emergency service providers, or the local community. These data indicate that a large PVC fire can result in reported symptoms in a locally-exposed community, but symptoms tend not to be serious, do not require medical attention, and are not long-lasting. Other studies with reported quantities of PVC combustion products are necessary to determine adequate dose-response relationships for short-term respiratory symptoms following accidental PVC fires.

The MOEE (1997) report for the fire presented monitoring data taken around the fire site. Trucks with Trace Atmospheric Gas Analyzer (TAGA) capabilities were dispatched the second day of the fire and analyzed the air for VOCs. Hydrogen chloride levels were measured as high as 0.930 mg/m3 over 20 hours of plume tracking during the fire. The health effects reported by the neighbors of the facility were most likely attributable to hydrogen chloride exposure. Levels of this compound dropped to <0.020 mg/m3 on the last day of the fire, and to a maximum instantaneous peak of 0.0034 mg/m3 on July 29. Dioxin levels were measured at ground level at two different times near the site, and at an air monitoring station 1 kilometer southwest of the site. A sample of dioxins in the fire plume was taken atop a building across from the fire. In general, the dioxin levels were 2.8-19.3 pg TEQ/m3. The sample atop the building across from the fire measured approximately 1100 pg TEQ/m3, but represented plume conditions to which the local residents were not exposed. Two weeks after the fire, dioxin levels in air were ≤0.05 pg TEQ/m3. Dioxin levels in soot measured 1 to 5 km from the fire ranged from 0.1 to 2.9 ng TEQ/m2, which was much lower than the acceptable clean-up value (25 ng/m2) for Quebec and New York. Sixteen samples of tree vegetation were sampled for dioxin levels; those distant from the fire were below detection limits, while two near sites had levels of 28 and 32 pg TEQ/g. Most of the sixteen samples exhibited trace levels of dioxins (<1 pg TEQ/g). Background dioxin levels at other urban areas ranged from <1 to 11 pg TEQ/g. Dioxin levels at two sites with higher contamination measured 8 days following the fire and after two rain events were decreased by ≥73%, indicating rapid declines in foliar dioxin contamination.

This study provides particularly useful information for studying the effects on the nearby environment, air quality, and adverse effects in the surrounding population following a large PVC fire. The data indicate that mild, but clinically-relevant, short-term effects are noted, most likely as the result of inhalation of hydrogen chloride. Dioxins were detected in the air and landed on nearby surfaces in the soot, but were not present on the ground at concentrations higher than regulated clean-up levels. Dioxins were also noted on the foliage of trees surrounding the location of the fire, but quickly decreased to background urban levels within approximately one week, following two rain events. These data indicate that a local increase in dioxin did not persist and should not contribute to any long-term health effects in the nearby population (MOEE, 1997).

3.3 Quantitative evidence: LCA and risk assessment

The Task Group has identified 10 life cycle impact assessment (LCIA) categories (see Table 4) and two occupational categories. Occupational risks, both non-cancer and cancer, are additive with the composite values for human health cancer and non-cancer, respectively; in some plots and tables they have been reported separately for purposes of explicit comparison.

Table 4. Life cycle impact assessment categories

Category	Abbreviation	Units of Measure
Human Health Cancer	HHC	Disability-adjusted life years
Human Health Mortality from Particulate Matter	HHPM	Disability-adjusted life years
Human Health Other (non-cancer effects)	HHO [HH NonC]	Hazard Index
Acid Rain	Acid	millimoles of hydrogen ions
Eutrophication	Eutro	grams nitrogen equivalent
Ecological Effects	Eco	grams of dioxin 2,4-D equivalents
Fossil Fuel Consumption	FF	surplus megajoules
Global Warming	GW	kilograms of CO2 equivalents
Ozone Depletion	O3	grams of CFC-11 equivalents
Smog	Smog	grams of nitrous oxide equivalents

In order to assess occupational risks, the major toxic chemicals that a worker would be exposed to were considered. These were also typically the predominant compounds in a building material on a mass basis. In general, non-cancer and cancer risk estimates are aggregate values that assumed additivity of toxic effects. (It is possible for some compounds to have antagonistic or synergistic effects, rather than additive effects. An analysis to determine if that is the case for these building materials has not been performed.) For example, exposure to vinyl chloride monomer and ethylene dichloride was assumed for workers making vinyl resin for all PVC products. Exposure doses for these compounds were obtained from literature, or agency exposure limits were assumed. As discussed previously, those exposure doses were then compared to either the Reference Dose or the Cancer Slope Factor for each compound to provide the resultant Hazard Index (HI, for non-cancer effects) or Integrated Lifetime Cancer Risk (ILCR, for cancer effects). The individual HI and ILCR values were then summed to provide an aggregate value for each type of effect. These values were then normalized to the functional unit of each building material manufactured per year.

The ILCR is essentially the chance of developing a cancer over and above one's background risk for developing the same cancer; if the ILCR is 5.2E-05, then the additional risk of developing cancer from manufacturing a particular building material is estimated to be 5.2 in 100,000. It is important to note that many of these cancer risk estimates may never be observable in the affected populations (e.g., occupational workers). For example, let's say the cancer risk for manufacturing PVC pipe is 1 in 10,000. If fewer than 10,000 people are involved in manufacturing either PVC resin or pipe, there will be insufficient statistical robustness in epidemiological studies to detect an increased incidence of cancer in the affected population.

3.3.1 Overview regarding risk assessment estimates

As one reviews the risk assessment contribution to the final LCA values, it is important to keep in mind which processes contribute the most to the overall risk estimates. 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 for EDC.

Individuals involved in the manufacture of plastic resins are potentially exposed to many other compounds besides individual resin monomers. For example, they can be exposed to 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. These solvents are used similarly in the manufacture of both PVC and ABS resins, and data regarding which one or ones are particular to either type of plastic resin were not located. 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 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.

It was not possible to develop risk estimates for fiber-cement siding and wood siding and windows 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 windows and 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., 1993; 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. Non-cancer and cancer risks, normalized to the functional unit of each building material, are provided below in Table 5 and are discussed below for each building material.

Siding

Risks for vinyl siding per functional unit at the low and high end varied by one order of magnitude. 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. 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).

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.

Piping

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 cadmium and 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. (1994) reported data that also indicated low emissions from thermal processing of ABS resin; therefore, only resin manufacture was assumed to contribute to the low C2S risk for this piping. For C2S high 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. Nevertheless, the non-cancer and cancer risks per functional unit are found to be comparable across the different piping materials.

Flooring

Risk estimates for vinyl flooring and vinyl composition tiles 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 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.

Windows

The risk estimates for aluminum windows reflect the same assumptions made for aluminum siding: the differences in risk reflect controls of exposures to coke oven emissions and PAHs. Risk estimates for vinyl windows 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. In general, the risks were comparable across the two window types, for both non-cancer and cancer risks. Health risks from manufacture of wood windows could not be calculated, as no dose-response data for nasal cancers were available.

Table 5. Non-cancer and cancer risk estimates normalized per functional unit of building material

Material	pwhr/$	HI	Cancer (ILCR)	Retail $/f.u.	Manufact. $ per f.u.	Total HI/f.u.	ILCR/ f.u.
Pipe: PVC							
Resin Low	0.008	140	7.4E-02				
Resin High	0.008	720	3.7E-01				
C2S Low	0.008	0.36	--	1.4	0.93	1.06E-04	3.68E-09
C2S Likely	0.008	1.1	2.2E-03	1.4	0.93	1.08E-04	4.12E-09
C2S High	0.008	11	3.9E-02	1.4	0.93	5.7E-04	2.62E-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		--	--			--	--

Material	pwhr/$	HI	Cancer (ILCR)	Retail $/f.u.	Manufact. $ per f.u.	Total HI/f.u.	ILCR/ f.u.
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	140	7.4E-02				
Resin High	0.008	720	3.7E-01				
C2S Low	0.008	0.36	--	1.16	0.77	8.75E-05	3.05E-09
C2S Likely	0.008	1.1	2.2E-03	1.16	0.77	8.93E-05	3.42E-09
C2S High	0.008	11	3.9E-02	1.16	0.77	4.73E-04	2.17E-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		
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	140	7.4E-02				
Resin High	0.008	720	3.7E-01				
C2S Low	0.008	0.38	6.8E-03	150	100.00	5.74E-03	3.61E-07
C2S Likely	0.008	1.7	9.3E-03	150	100.00	6.21E-03	4.21E-07
C2S High	0.008	11	4.6E-02	150	100.00	3.28E-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							

Material	pwhr/$	HI	Cancer (ILCR)	Retail $/f.u.	Manufact. $ per f.u.	Total HI/f.u.	ILCR/ f.u.
Window: Softwood							
C2S Low	0.008	2137	--	250	166.67	1.42	--
C2S High	0.008	6600	--	250	166.67	4.4	--
Construct.		NA	NA				
End use		NA	NA				
End of life							
Flooring: VCT							
Resin Low	0.008	200	0.075				
Resin High	0.008	770	4.1E-01				
C2S Low	0.008	9.8	--	3.20	2.13	2.78E-04	5.12E-09
C2S Likely	0.008	10	4.1E-05	3.20	2.13	2.80E-04	5.14E-09
C2S High	0.008	35	8.2E-04	3.20	2.13	1.05E-03	2.48E-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	140	7.4E-02				
Resin High	0.008	720	3.7E-01				
C2S Low	0.008	9.8	--	3.60	2.40	3.44E-04	9.47E-09
C2S Likely	0.008	10	4.1E-05	3.60	2.40	3.46E-04	9.49E-09
C2S High	0.008	35	8.2E-04	3.60	2.40	1.65E-03	4.78E-08
Construct.		0.90	--			3.75E-06	--
End use		0.27	--			2.81E-04	--
End of life							
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-04	5.6E-10	4	2.67	5.12E-09	3.98E-16
C2S Likely	0.008	4.4E-03	6.2E-09	4	2.67	4.69E-08	4.41E-15
C2S High	0.008	8.4E-03	1.2E-08	4	2.67	8.96E-08	8.53E-15
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

Material	pwhr/$	HI	Cancer (ILCR)	Retail $/f.u.	Manufact. $ per f.u.	Total HI/f.u.	ILCR/ f.u.
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.

3.3.2 Siding

Integrated LCA and Risk Assessment

A graphical summary is presented below, showing the cradle-to-site and use-phase LCA plus occupational risk results for all impact categories, for each of the siding alternatives. In order to display the results for all impact categories on one graph, the results have been internally normalized separately within each impact category, with respect to the median LCA results for the PVC alternative for that impact category. Note that the value axis presents a unit-less result, which is the ratio of an LCA result to the median PVC LCA result for that impact category. Also note that the value axis is logarithmic, meaning that equal vertical distance represents a factor 10 difference.

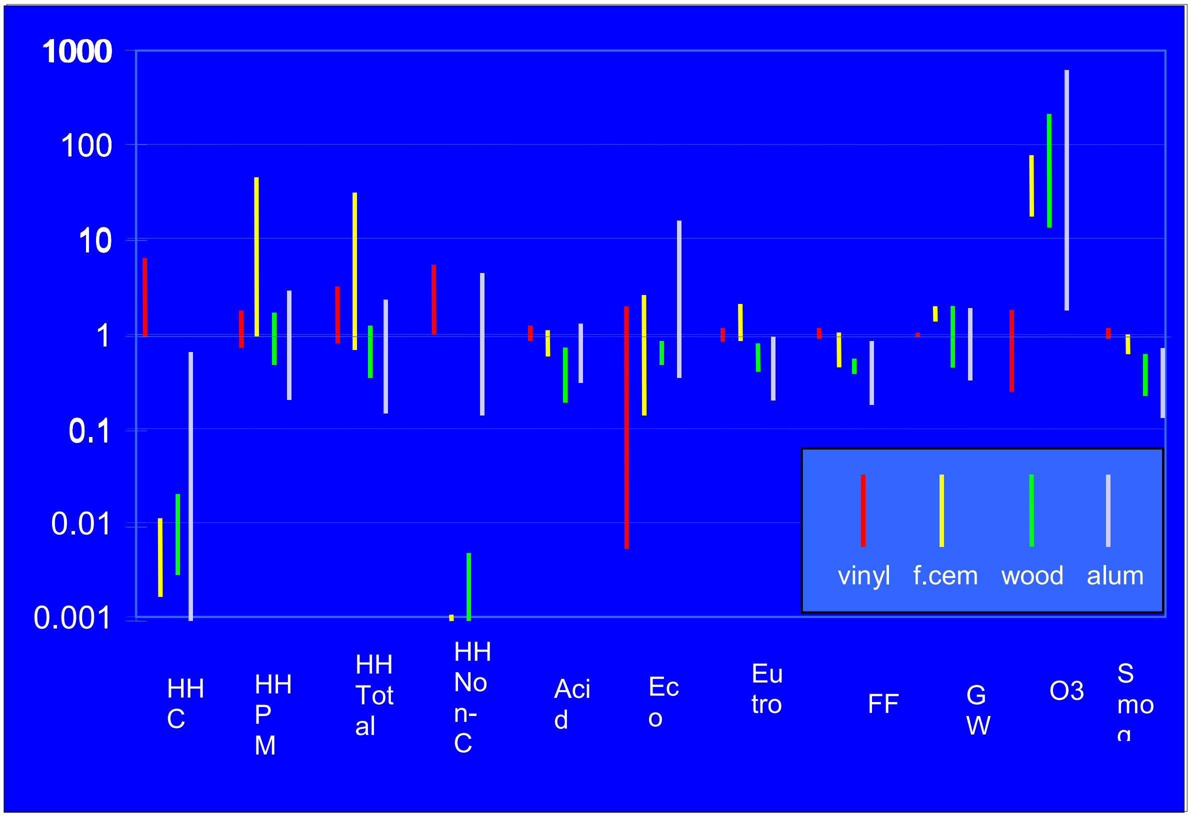

Figure 6. LCA results for siding alternatives, internally normalized by vinyl siding (occupational risks included in human health impact categories)

Several separate conclusions can be drawn from the results in the graph. First it can be noted that different assumptions and data sources for the LCA modeling can lead to variations in the results for a given material and impact category that range over more than a factor of ten (an order of magnitude). This serves as a helpful reminder that making comparisons among products based on LCA results requires the use of common databases and assumptions.

In terms of the relative performance of the different material options, the following general conclusions can be drawn. First, wood siding performs quite well; its maximum values are typically below the maxima for all other material options for most impact categories, and the midrange of its values is often the lowest of the four materials' midrange values. Note that an important caveat is our assumption of no painting to the wood siding.

Another general conclusion is that fiber cement tends to perform the worst among the options, for most impact categories; the maximum value of its range tends to be at or among the highest within each impact category. The value ranges for fiber cement tend to be higher than those for PVC for most impact categories. For human health impacts of fiber cement, the highest values are based on the BEES data that assume the Portland cement content by 90%. Although other data sources such as MSDS do not specify Portland cement for the content, those high values have been kept since use of Portland cement for fiber cement siding can be reasonably assumed.

Finally, the value ranges tend to be largest for aluminum siding. This variation is primarily due to the range of possible recycled content values, which among our LCA data points span the spectrum from 100% recycled content to 0% recycled content. The findings indicate that if one could obtain aluminum siding with 100% recycled content, it would be among the better performing options for most impact categories; however, at 0% recycled content, it tends to be among the 2 worst options across the impact categories. A baseline standard recycled content for aluminum siding is reported to be 30%, tending to place it worse than wood siding in most categories, but better than fiber cement in most categories.

The results in the previous plot were normalized with respect to the median result for PVC in each impact category. A different way to normalize is with respect to the total annual impacts for some other "reference system." A common reference system chosen for such purposes is the total annual emissions per capita for the nation of interest. In our case we use data for the U.S. as the basis for normalization. The results of such an "external" normalization method shed light on the relative influence of the product choice upon the total annual impacts for the reference system on the different impact categories. For example, in the plot below, we note that the selection of siding alternatives has slightly higher influence on human health (focus on total), ecological effects, fossil fuel depletion, global warming and smog and a relatively lower degree of influence on acidification and ozone depletion impacts. Note also that the "human health non-cancer" is missing from this plot, as the information required for external normalization of that category is not available.

Figure 7. LCA results for siding alternatives, externally normalized by BEES normalization factors for U.S. flows per capita (occupational risks included in human health impact categories)

Population *vs.* Occupational Risks

The figure below compares the results of the upstream (LCA) mortality health risks with the occupational risks per functional unit. Results for two different pollutant categories are presented in the figure: pollutants linked to cancer risk and primary and secondary particulate emissions. For the cancer and particulate categories, we present results in units of micro DALY per functional unit. For the non-lethal toxic impacts, the results are expressed using the unitless "hazard index" (HI) discussed in earlier sections of the report. Recall that a DALY is a "disability-adjusted life year" lost; by far the majority of these impacts are mortality impacts (life years lost) rather than morbidity impacts (years lived with a chronic illness or disability).

Several conclusions can be drawn from the results in the figure below. First, by comparing the median LCA cancer impacts (labeled "C:") and the median LCA particulate matter impacts (labeled "PM"), it can be noted that for the pollutant releases to the environment over the life cycle, the population particulate risk to human health exceeds the estimated population cancer risk to human health (compare the first four blue diamonds with the next 4 blue diamonds).

Next, it is possible to note that the occupational cancer risks from the vinyl siding production are within an order of magnitude of the population particulate health risks for vinyl siding and all of the alternatives. Indeed, this is true not only for the high-end occupational cancer risk estimate, but for the median and low-end estimates as well.

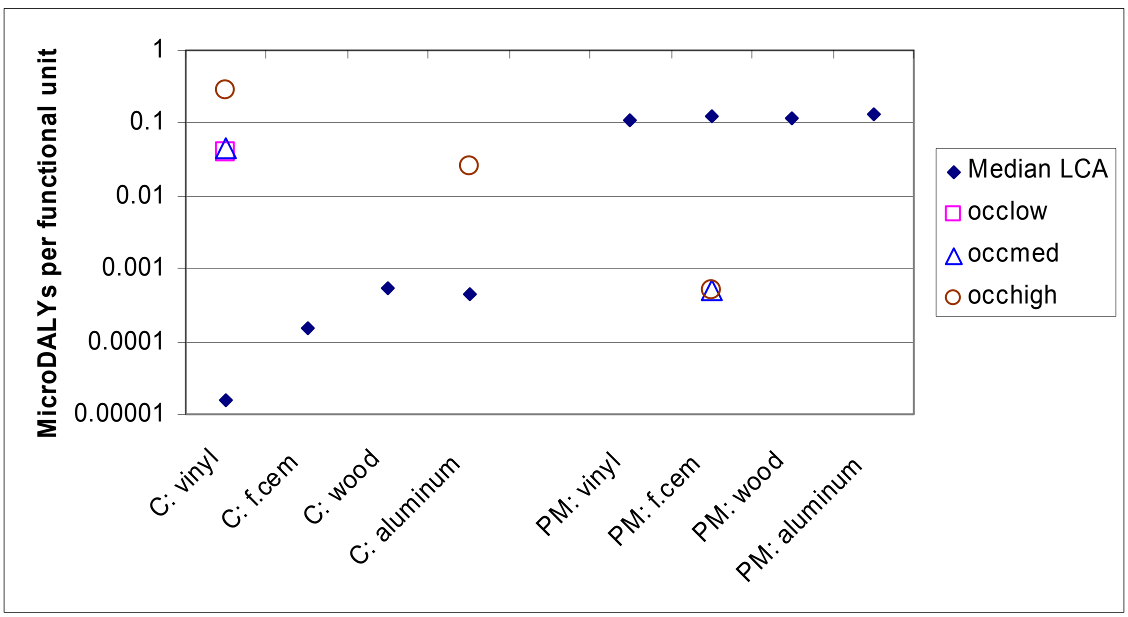

Figure 8. Population and occupational cancer and PM-related DALYs per functional unit – siding

The results for non-carcinogenic human health impacts are plotted below. For the non-lethal health effects, it is quite clear from the figure that the occupational exposures lead to expected total hazard indices which dwarf by several orders of magnitude the expected sum of hazard indices due to human exposures via environmental pathways.

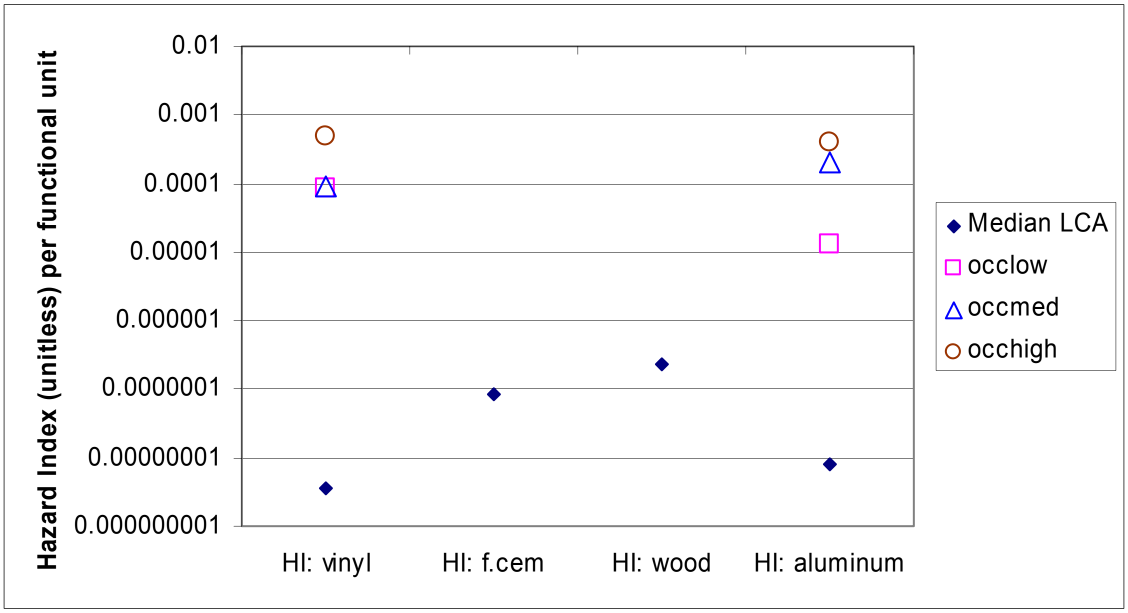

Figure 9. Population and occupational non-cancer hazard index per functional unit – siding

Lastly, it is instructive to compare the alternatives on the basis of total mortality risk, summing population and occupational exposures, and summing the impacts of carcinogens as well as particulate emissions. The results are displayed in the figure below. Because "stacked bars" are being plotted, the axis in this figure is not logarithmic, but linear. Note that median occupational risks have been aggregated with the median LCA (population) risks in these figures.

Based on the figure below, among the siding alternatives, vinyl siding leads to the highest total mortality risks, while wood siding leads to the lowest. However, the differences are small and may be considered insignificant given the relatively large uncertainty levels associated with these results. Almost all mortality risks are driven by population exposure to particulate matter for all siding types except vinyl, for which the occupational risks account for 29% of the total mortality risk.

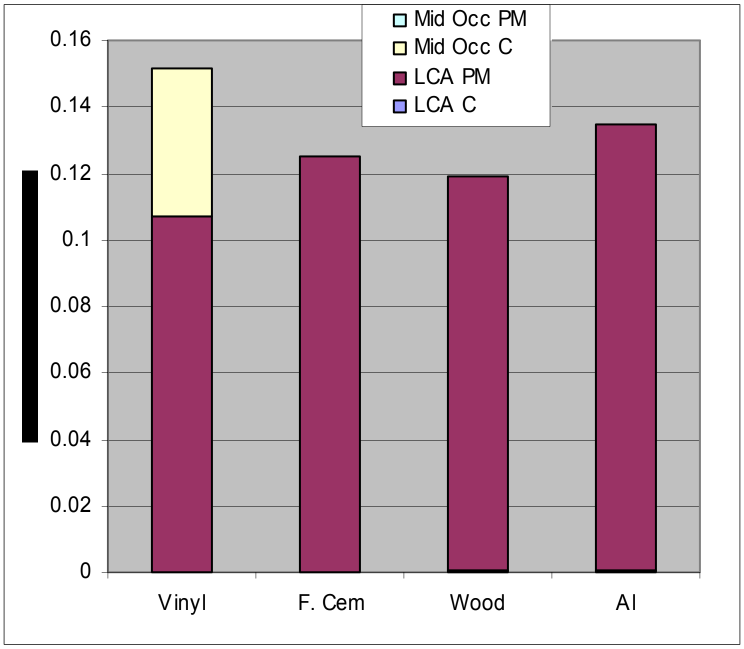

Figure 10. Aggregated population and occupational cancer and PM-related health risks (DALYs per f.u.) – siding

3.3.3 Piping

Integrated LCA and Risk Assessment

A graphical summary is presented below, showing the cradle-to-site and use-phase LCA plus cradle-to-site occupational risk results for all impact categories, for each of the pipe alternatives. In order to display the results for all impact categories on one graph, the results have been normalized separately within each impact category, with respect to the median LCA results for the PVC alternative for that impact category. Note that the value axis presents a unit-less result, which is the ratio of an LCA result to the median PVC LCA result for that impact category. Also note that the value axis is logarithmic, meaning that equal vertical distance represents a factor 10 difference.

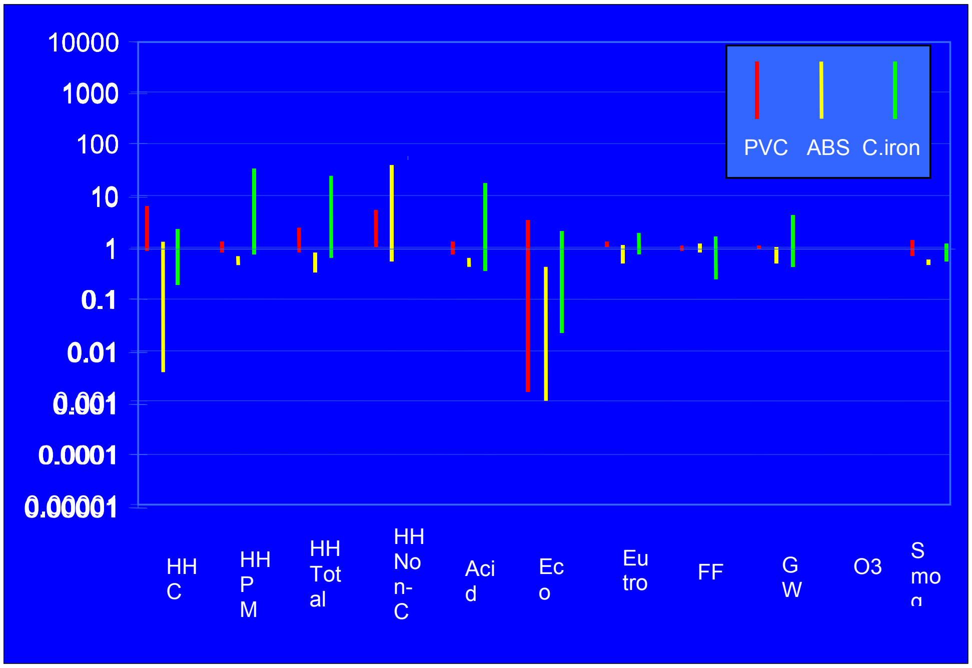

Figure 11. LCA results for pipe alternatives, internally normalized by PVC pipe (occupational risks included in human health impact categories)

Several separate conclusions can be drawn from the results in the graph. As with siding, it can be noted that different assumptions and data sources for the LCA modeling can lead to variations in the results for a given alternative, for a given impact category, that range over more than a factor of ten (an order of magnitude).

In terms of the relative performance of the different material options, the following general conclusions can be drawn. First, ABS pipe tends to perform well; its maximum values are below the maxima for all other material options, for many impact categories, and the midrange of its values is often the lowest of the three materials' midrange values. Contrary to the situation with siding, we do not observe one material option performing consistently poorest among the options across the impact categories. PVC has the highest values in some impact categories while cast iron in the others.

Note that ozone depletion has not been quantified for PVC and ABS pipe due to lack of data, and therefore the internal normalization has been impossible.

The results in the previous plot were normalized with respect to the median result for PVC in each impact category. A different way to normalize is with respect to the total annual impacts for some other "reference system." A common reference system chosen for such purposes is the total annual emissions per capita for the nation of interest. In our case we use data for the U.S. as the basis for normalization. The results of such an "external" normalization method shed light on the relative influence of the product choice upon the total annual impacts for the reference system on the different impact categories. For example, in the plot below, we note that the selection of pipe alternatives has relatively similar levels of impacts across impact categories except acidification and ozone depletion. Note that ozone depletion has not been quantified for PVC and ABS piping due to lack of data.

Figure 12. LCA results for pipe alternatives, externally normalized by BEES normalization factors for U.S. flows per capita (occupational risks included in human health impact categories)

Population *vs.* Occupational Risks

The figure below compares the results of the upstream (LCA) mortality health risks with the occupational risks per functional unit.

The main conclusions drawn from examination of these results among the siding alternatives appear valid for the pipe alternatives as well. First, by comparing the median LCA cancer impacts (labeled "C:") and the median LCA particulate matter impacts (labeled "PM"), it can be noted that for the pollutant releases to the environment over the life cycle, the population particulate risk to human health exceeds the estimated population cancer risk to human health. (compare the first three blue diamonds with the next three blue diamonds.)

Next, it is possible to note that the high-end occupational cancer risk estimates from the production of all three materials are within the same range of orders of magnitude as the population particulate health risks for all pipe alternatives. Moreover, the median occupational risks are lower than the population PM-related risks but are within an order of magnitude.

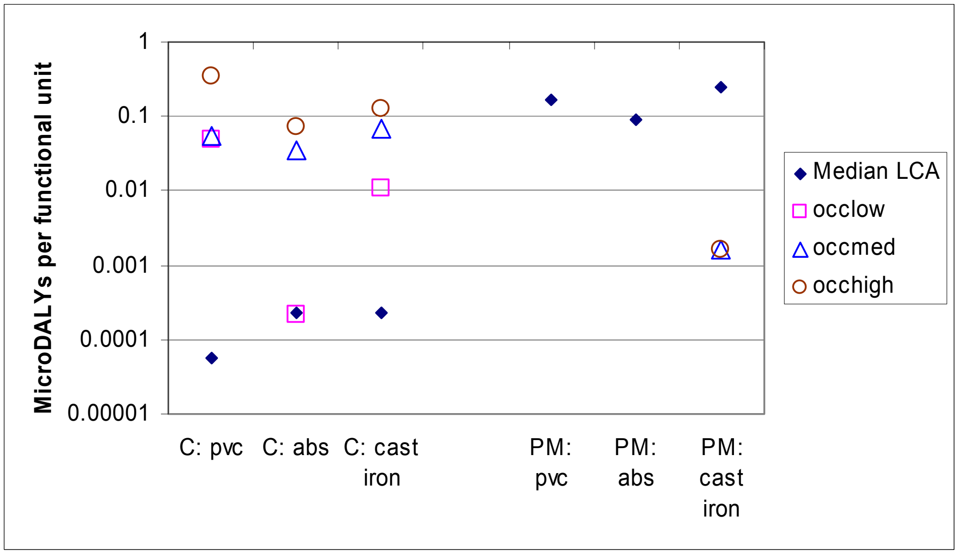

Figure 13. Population and occupational cancer and PM-related DALYs per functional unit – piping

The results for non-carcinogenic human health impacts are plotted below. For the non-lethal health effects, it is quite clear from the figure that the occupational exposures lead to expected total hazard indices which dwarf by several orders of magnitude the expected sum of hazard indices due to human exposures via environmental pathways.

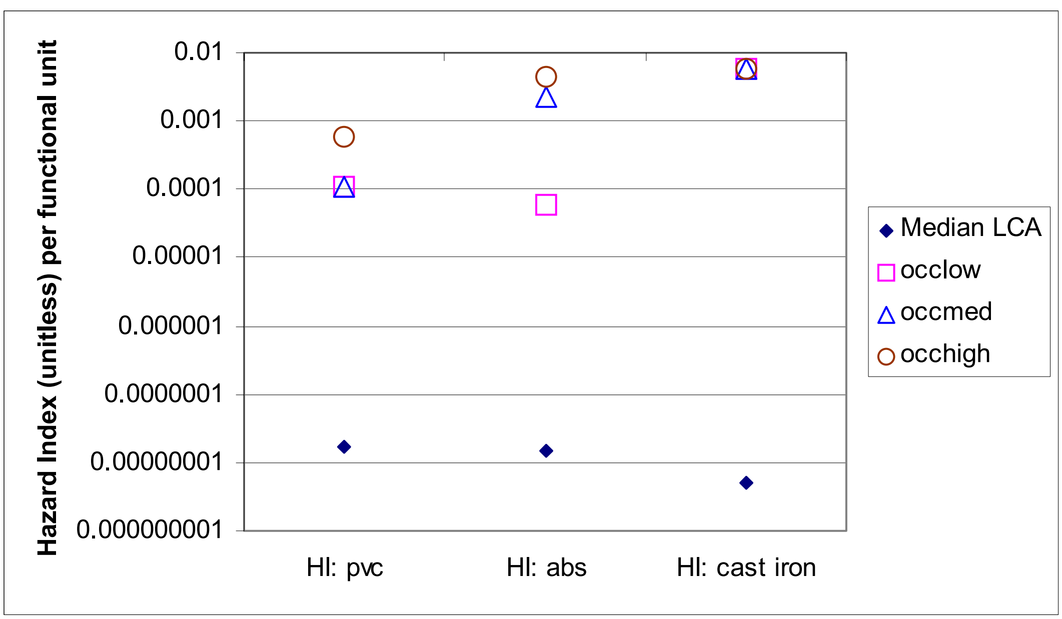

Figure 14. Population and occupational non-cancer hazard index per functional unit – piping

Lastly, it is instructive to compare the alternatives on the basis of total mortality risk, summing population and occupational exposures, and summing the impacts of carcinogens as well as particulate emissions. The results are displayed in the figure below. Because "stacked bars" are being plotted, the axis in this figure is not logarithmic, but linear.

The figure below summarizes the aggregate human mortality risks for the pipe alternatives. The conclusion can be made that cast iron pipe leads to the highest total mortality risks, while ABS pipe leads to the lowest among the alternatives. For all pipe types occupational risks contribute the total mortality risks by 22% for cast iron to 29% for ABS.

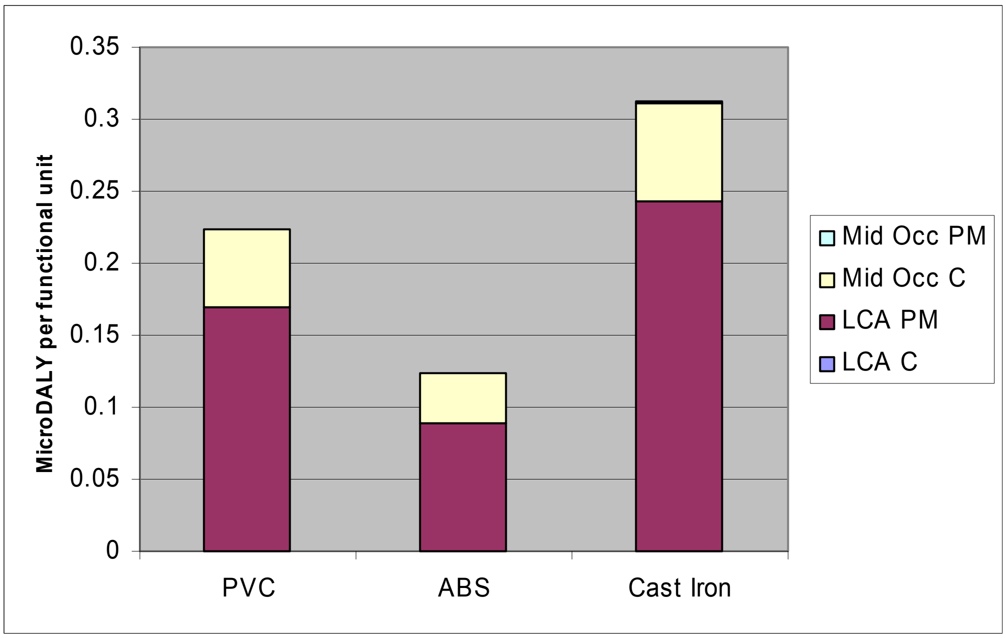

Figure 15. Aggregated population and occupational cancer and PM-related health risks (DALYs per f.u.) – piping

3.3.4 Flooring

Integrated LCA and Risk Assessment

A graphical summary is presented below, showing the cradle-to-site LCA plus cradle-to-site, installation and use-phase occupational risk results for all impact categories, for each of the flooring alternatives. In order to display the results for all impact categories on one graph, the results have been normalized separately within each impact category, with respect to the median LCA results for vinyl composition tile (VCT) for that impact category. Note that the value axis presents a unit-less result, which is the ratio of an LCA result to the median VCT result. Also note that the value axis is logarithmic, meaning that equal vertical distance represents a factor 10 difference.

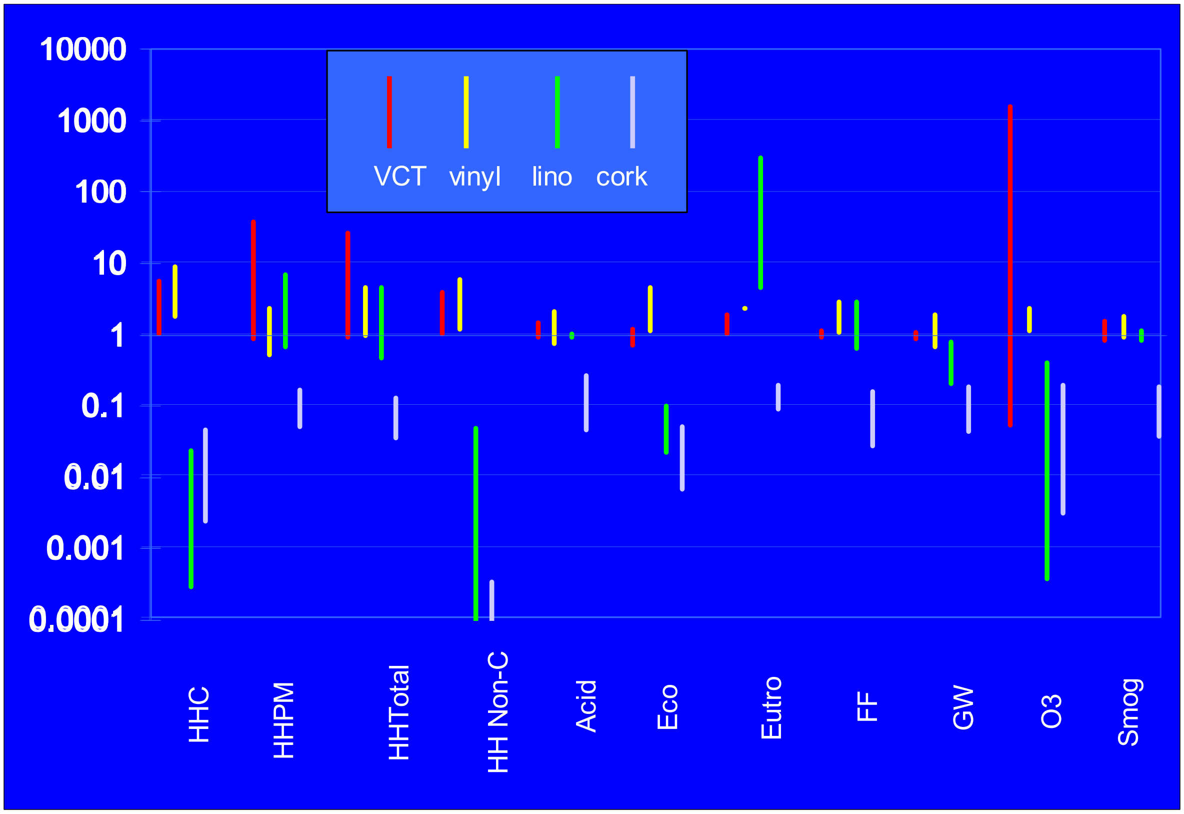

Figure 16. LCA results for flooring alternatives, internally normalized by VCT (occupational risks included in human health impact categories)

In terms of the relative performance of the different material options, the following general conclusions can be drawn. First, cork flooring performs very well; its maximum values are typically below the maxima for all other material options, for most impact categories, and the midrange of its values is often the lowest of the four materials' midrange values. This result may be attributed in part to the relatively long service life assumed for cork (50 years, compared with 15 to 18 years for the others). But even without that assumption cork would score better than the others in most categories.

Like piping alternatives, we do not observe one material option performing consistently poorest among the options across the impact categories. Vinyl has the highest values in some impact categories, VCT in others, and linoleum in others. In general, however, linoleum tends to perform better than the two vinyl options on a majority of impact categories.

The results in the previous plot were normalized with respect to the median result for VCT in each impact category. A different way to normalize is with respect to the total annual impacts for some other "reference system." A common reference system chosen for such purposes is the total annual emissions per capita for the nation of interest. In our case we use data for the U.S. as the basis for normalization. The results of such an "external" normalization method shed light on the relative influence of the product choice upon the total annual impacts for the reference system on the different impact categories. For example, in the plot below, we note that the selection of flooring alternatives has slightly higher influence on eutrophication and ecological effects, a relatively lower degree of influence on total acidification impacts and ozone depletion impacts, and otherwise similar level of influence on the remaining categories.

Figure 17. LCA results for flooring alternatives, externally normalized by BEES normalization factors for U.S. flows per capita (occupational risks included in human health impact categories)

Population *vs.* Occupational and Use-phase Risks

The figure below compares the results of the upstream (LCA) mortality health risks with the occupational risks as well as health risks during the usage-phase per functional unit. In contrast to the results for pipe, siding and windows, the flooring results also incorporate estimates of risk from the installation of the flooring and from exposure during the use phase. A range of alternatives are plotted for each material: the median of the LCA results; high, low, and median values for cradle to site (C2S) occupational exposures; high, low, and median values for risks during construction (Con); and high, low, and median values for risks to the end users (EU).

The main conclusions drawn from examination of these results among both the siding and pipe alternatives appear valid for the flooring alternatives as well. First, by comparing the median LCA cancer impacts (labeled "C:") and the median LCA particulate matter impacts (labeled "PM"), it can be noted that for the pollutant releases to the environment over the life cycle, the population particulate risk to human health exceeds the estimated population cancer risk to human health.

Next, it is possible to note that the high-end occupational cancer risk estimates from the production of the two vinyl flooring alternatives are within an order of magnitude of the population particulate health risks for the two vinyl flooring alternatives and linoleum, but not for the cork flooring, whose particulate emissions estimate is lower than the other options. This is true not only for the high-end occupational cancer risk estimate, but for the median and low-end estimates for vinyl flooring options as well.

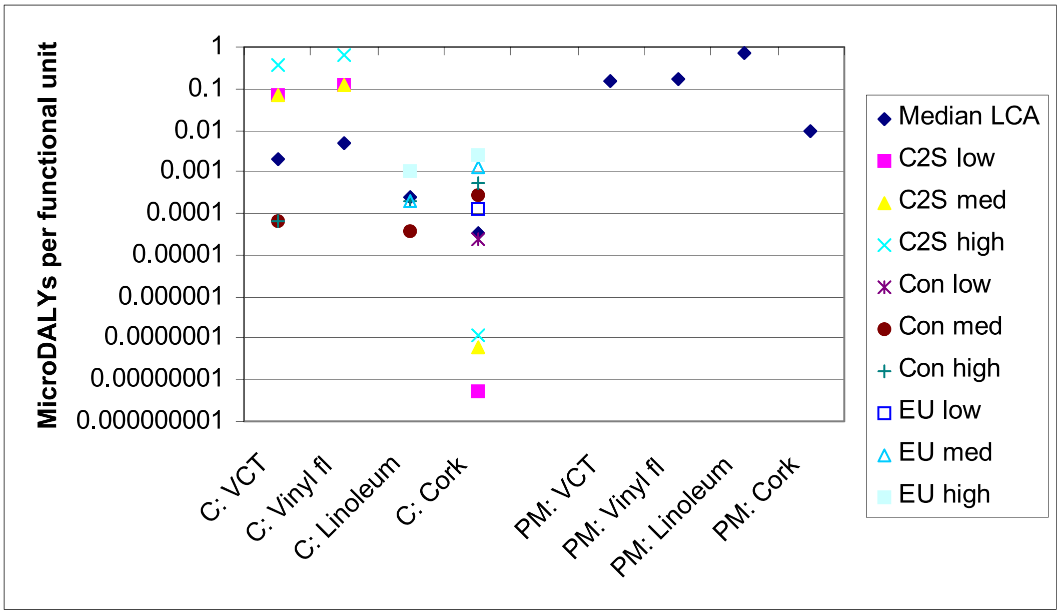

Figure 18. Population and occupational cancer and PM-related DALYs per functional unit – flooring

The results for toxic human health impacts other than cancer are plotted below. For these non-cancer health effects, the median occupational risks during the manufacturing stage (C2S) are higher than the population risks for all flooring alternatives (compare the yellow triangles with the blue diamonds.)

This means that the total non-lethal health effects from occupational exposures for all stages (i.e., manufacturing and installation) are even higher than the population risks associated with the outdoor environmental exposure. For vinyl flooring, the median health risks associated with indoor exposures during the use phase are higher than the population risks associated with outdoor exposures.

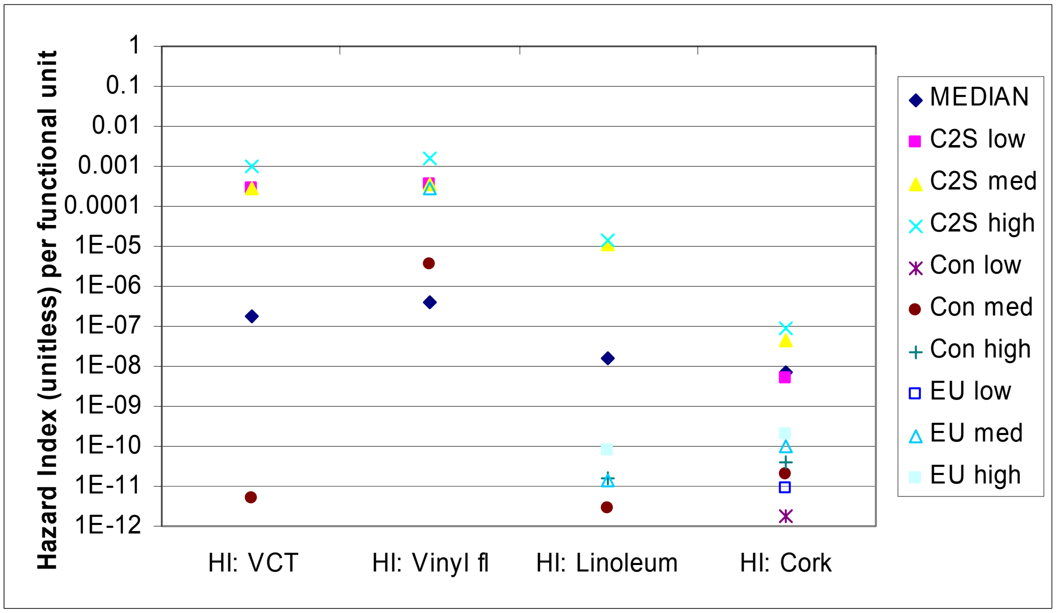

Figure 19. Population and occupational non-cancer hazard index per functional unit – flooring

The figures below summarize the aggregate human mortality risks for the flooring alternatives. Our sample of results contains seven low values and three values which are 50 times greater; thus, the comparative conclusion concerning which material has the highest aggregated health impacts depends on whether we choose the mean or median value for VCT as a basis of comparison. The median provides a measure of central tendency that is more robust, less susceptible to outliers, than the mean, and for this reason we have tended to use median results as our basis of comparison in this report. Figure 19, based on the median values, shows that the total mortality risks per functional unit for linoleum is the highest among the alternatives.. On the other hand, based on the mean values, as shown in Figure 20, the total mortality risks per functional unit for VCT is the highest among the alternatives.

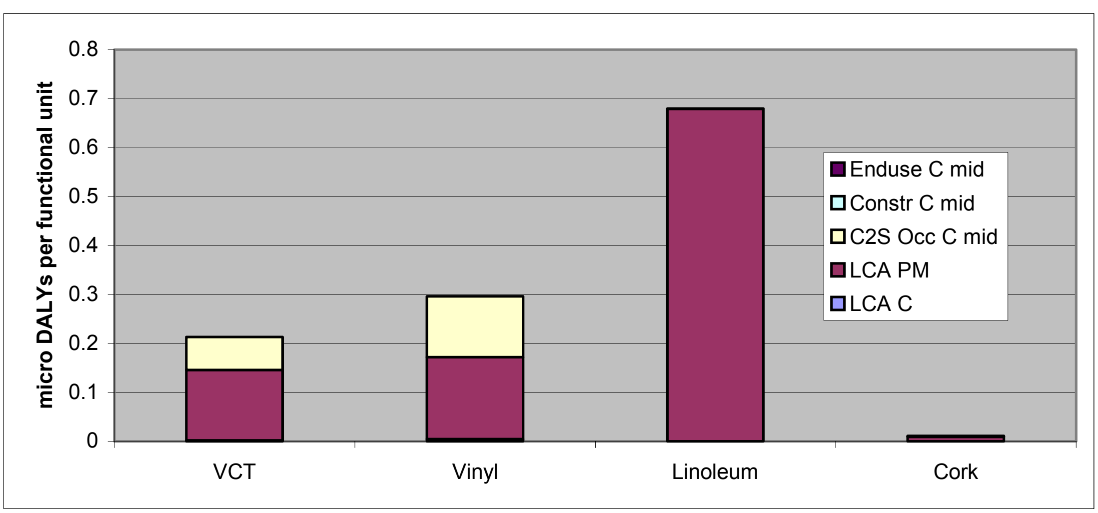

Figure 20. Aggregated population and occupational cancer and PM-related health risks (DALYs per f.u.) – flooring

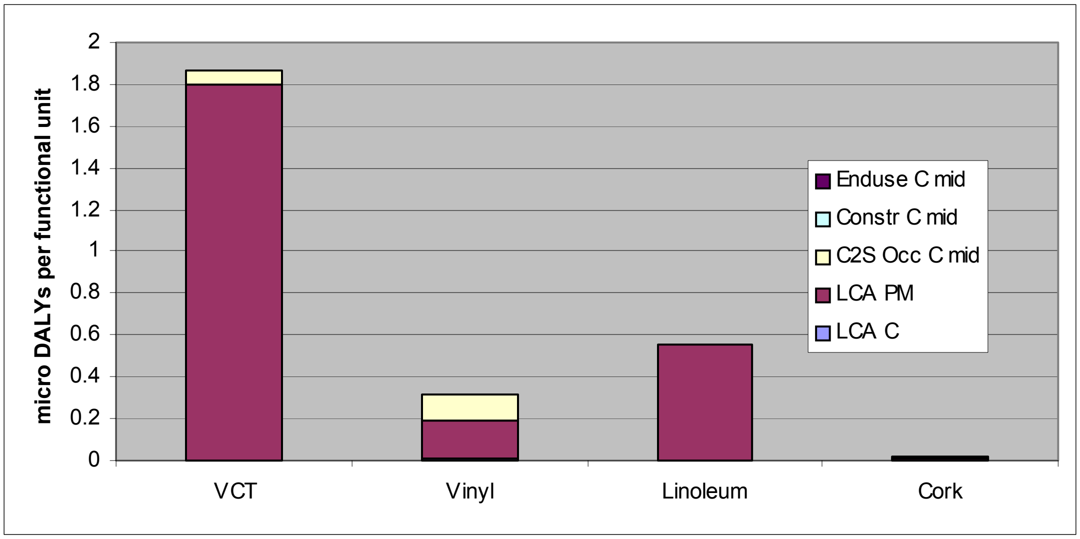

Figure 21. Aggregated population and occupational cancer and PM-related health risks (DALYs per f.u.) based on the mean values– flooring

Note that the selection of median versus mean only changes the aggregated human health rankings of materials for flooring, not for pipe, siding, or windows. In both cases, cork is the lowest for the aggregate human mortality risks. In this aggregated scale the only significant contributors are occupational cancer risk (C2S Occ C mid) and PM-related population mortality risks (LCA PM).

3.3.5 Windows

Integrated LCA and Risk Assessment

The figure presented below summarizes the cradle-to-site and use-phase LCA plus the cradle-to-site occupational risk results for all impact categories, for each of the window alternatives. In order to display the results for all impact categories on one graph, the results have been normalized separately within each impact category, with respect to the median results for the vinyl alternative for that impact category. Note that the value axis presents a unit-less result, which is the ratio of an LCA result to the median vinyl LCA result for that impact category. Also note that the value axis is logarithmic, meaning that equal vertical distance represents a factor 10 difference.

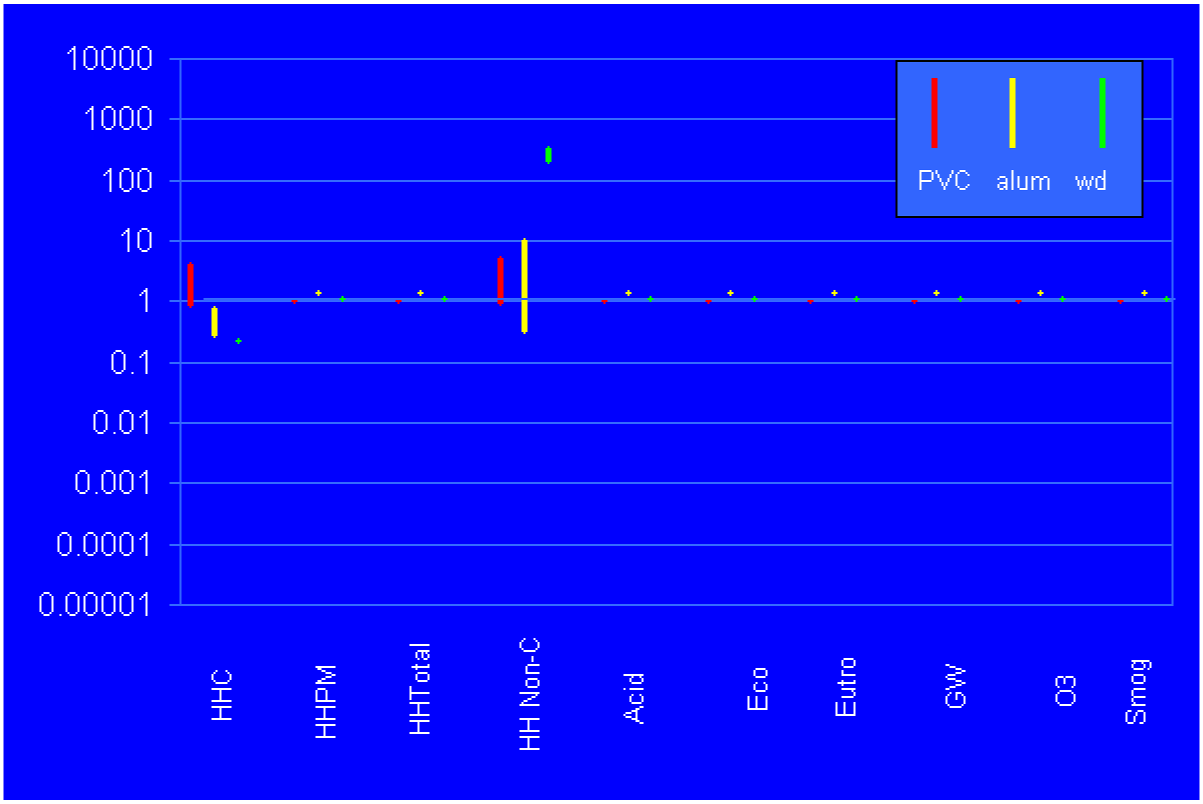

Figure 22. LCA results for window alternatives, internally normalized by vinyl window (occupational risks included in human health impact categories)

In contrast with the other product groups analyzed in this report, the results for windows are remarkably similar across the product alternatives. This is because the total life cycle results are not dominated by the frame material, but rather by the glazing and the usage phase energy use.

The results in the previous plot were normalized with respect to the median result for PVC in each impact category. A different way to normalize is with respect to the total annual impacts for some other "reference system." A common reference system chosen for such purposes is the total annual emissions per capita for the nation of interest. In our case we use data for the U.S. as the basis for normalization. The results of such an "external" normalization method shed light on the relative influence of the product choice upon the total annual impacts for the reference system on the different impact categories. For example, in the plot below, we note that the selection of window alternatives has slightly higher influence on ecological effects, global warming, and smog, a relatively much lower degree of influence on total acidification impacts and ozone depletion impacts, and otherwise similar level of influence on the remaining categories.

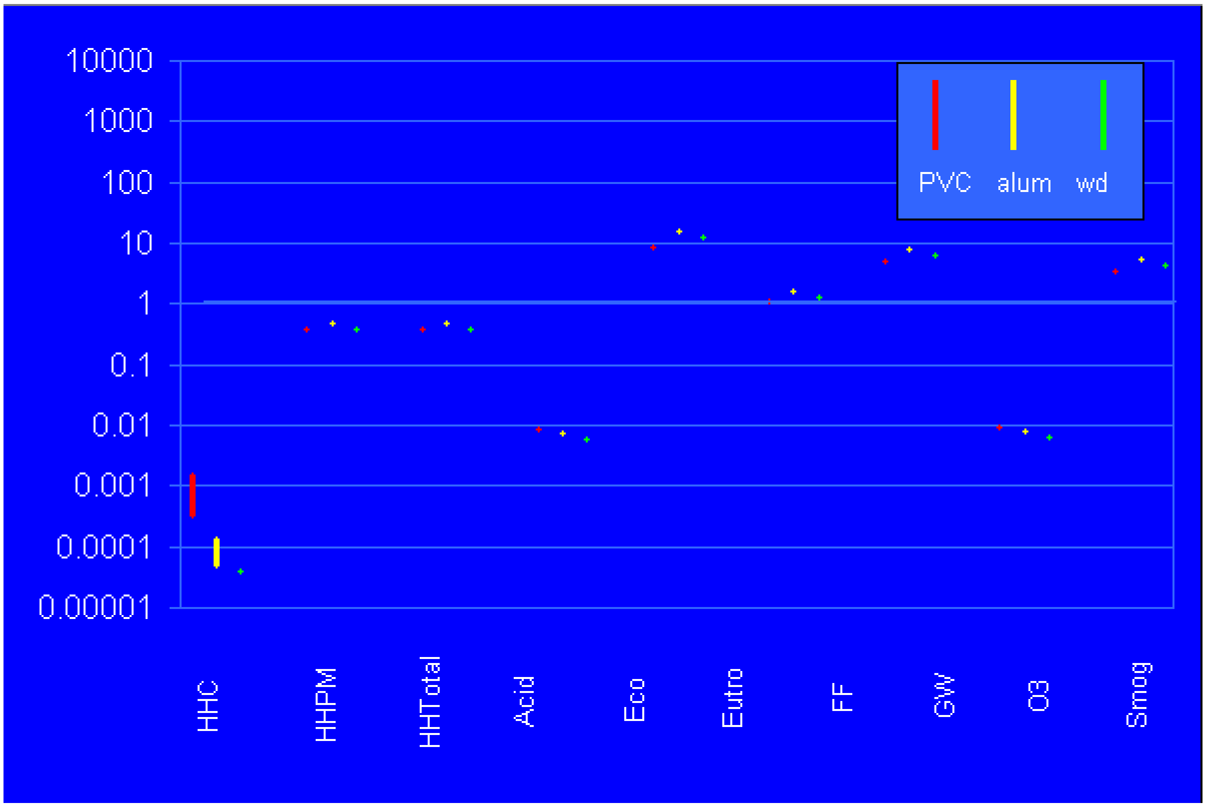

Figure 23. LCA results for window alternatives, externally normalized by BEES normalization factors for U.S. flows per capita (occupational risks included in human health impact categories)

Population *vs.* Occupational Risks

The figure below compares the results of the upstream (LCA) mortality health risks with the occupational risks per functional unit.

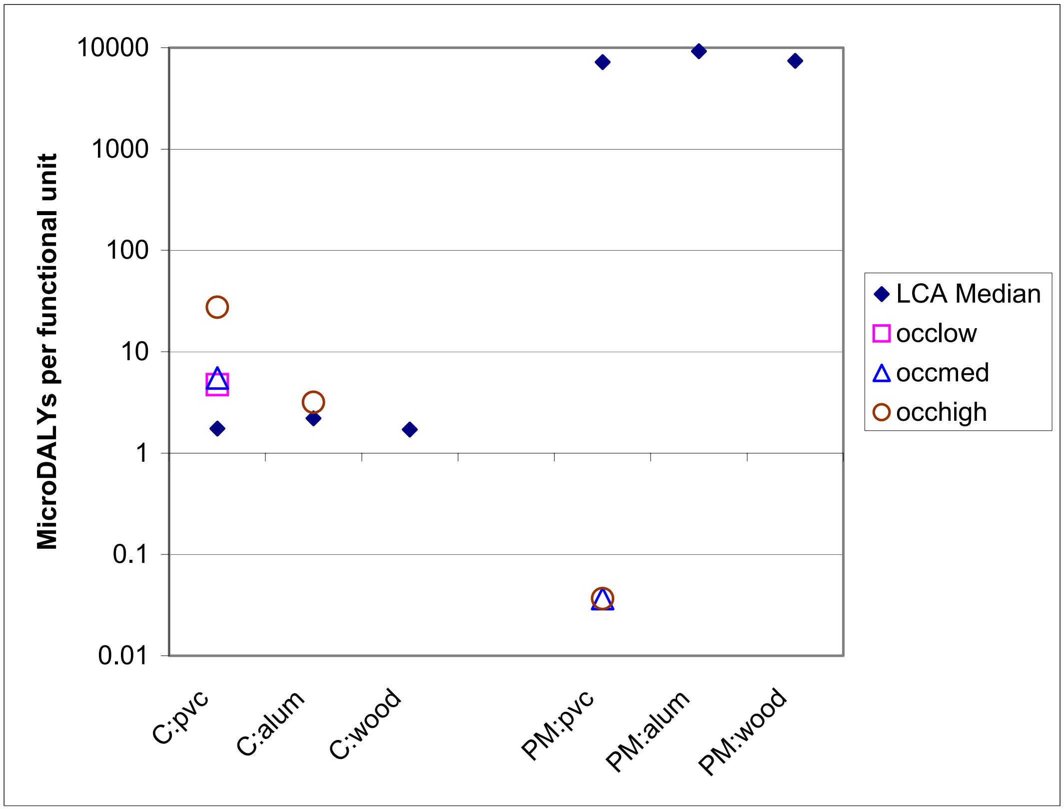

Figure 24. Population and occupational cancer and PM-related DALYs per functional unit – windows

Particulates dominate human mortality impacts for windows, a finding that differs from the pattern found with the other three materials. This is mainly due to the importance of the usage phase energy use.

The results for non-carcinogenic human health impacts are plotted below. For the non-lethal health effects, it is quite clear from the figure that for all window alternatives the occupational exposures lead to higher hazard indices than the population risks associated with outdoor emissions to the environment.

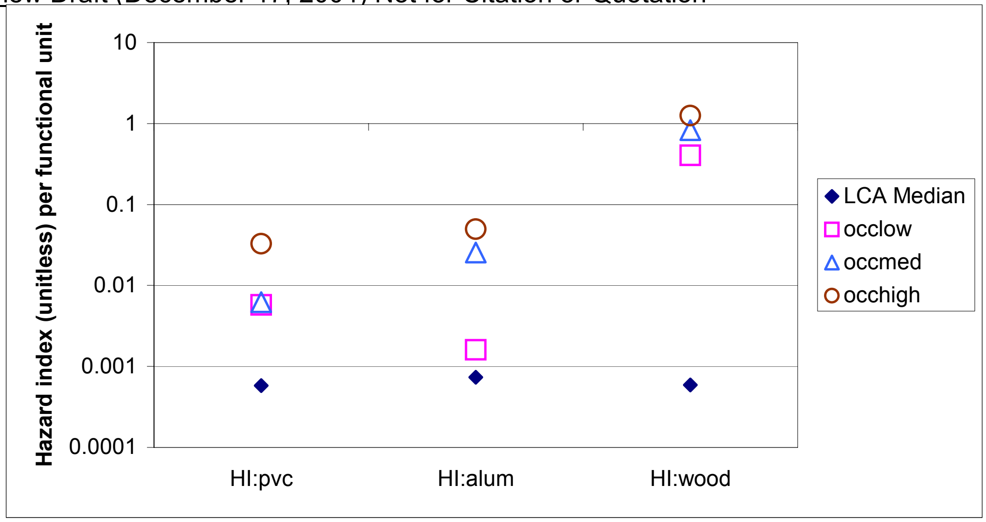

Figure 25. Population and occupational non-cancer hazard index per functional unit – windows

The figure below summarizes the aggregate human mortality risks for the window alternatives. The comparative performance of the window alternatives is entirely driven by differences in the energy efficiency of the studied alternatives. Thus, among our three frame types studied, the aluminum framed window had the worst energy efficiency, while the wood and PVC framed windows were virtually identical in efficiency. Rather than conclude that aluminum windows are always worse than wood and PVC, or that PVC and wood windows are always equal, the proper conclusion to be drawn from this analysis is that from a life cycle human health perspective, the key issue is window energy efficiency rather than frame material.

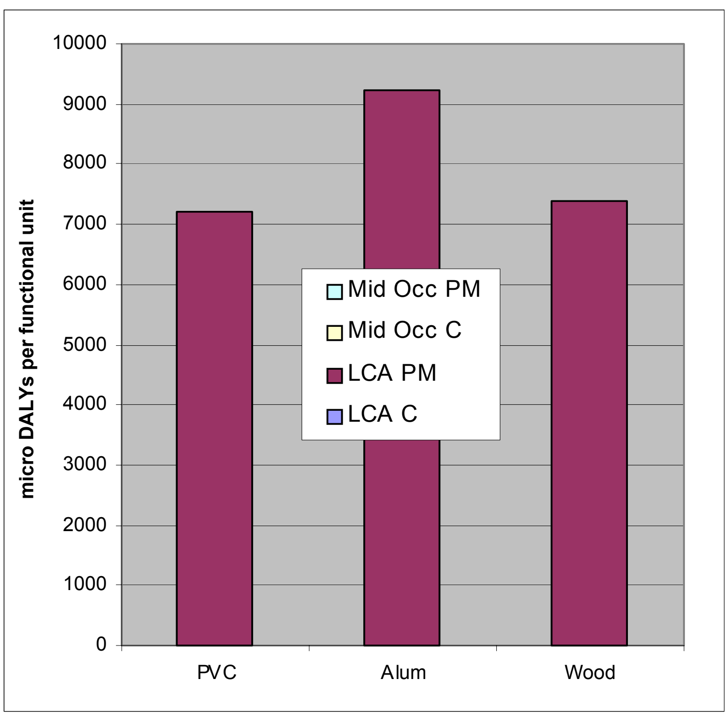

Figure 26. Aggregated population and occupational cancer and PM-related health risks (DALYs per f.u.)– windows

3.3.6 Focus on Human Health

This section discusses the results of the quantitative risk assessments for occupational exposures per functional unit, and their integration with the life cycle population health risk modeling. Integration on a common functional unit basis enables us to directly compare the relative contributions to total mortality risk from three different categories of pollutant exposure:

- Population risk from exposure to particulates released to air by processes throughout the product supply chains;
- Population risk from exposure to carcinogens released to air and water by processes throughout the product supply chains; and
- Occupational risk from exposure to carcinogens to which production workers may be exposed.

There are three major findings from our integrated evaluation, which are consistent across the four product groups evaluated. The first is that the total life cycle mortality results are dominated by particulate exposures, rather than cancer. As shown in the figure and table below, in fact, of the total median mortality risk (which is the sum of the median occupational risks and the median population risks from exposures to particulates and carcinogens combined), the cancer risks account for (across all product alternatives, and using all available LCA data alternatives), 0.04-43% for flooring alternatives, 0.1-29% for siding alternatives, 22-29% for pipe alternatives, and well under a tenth of 1% for all window alternatives.

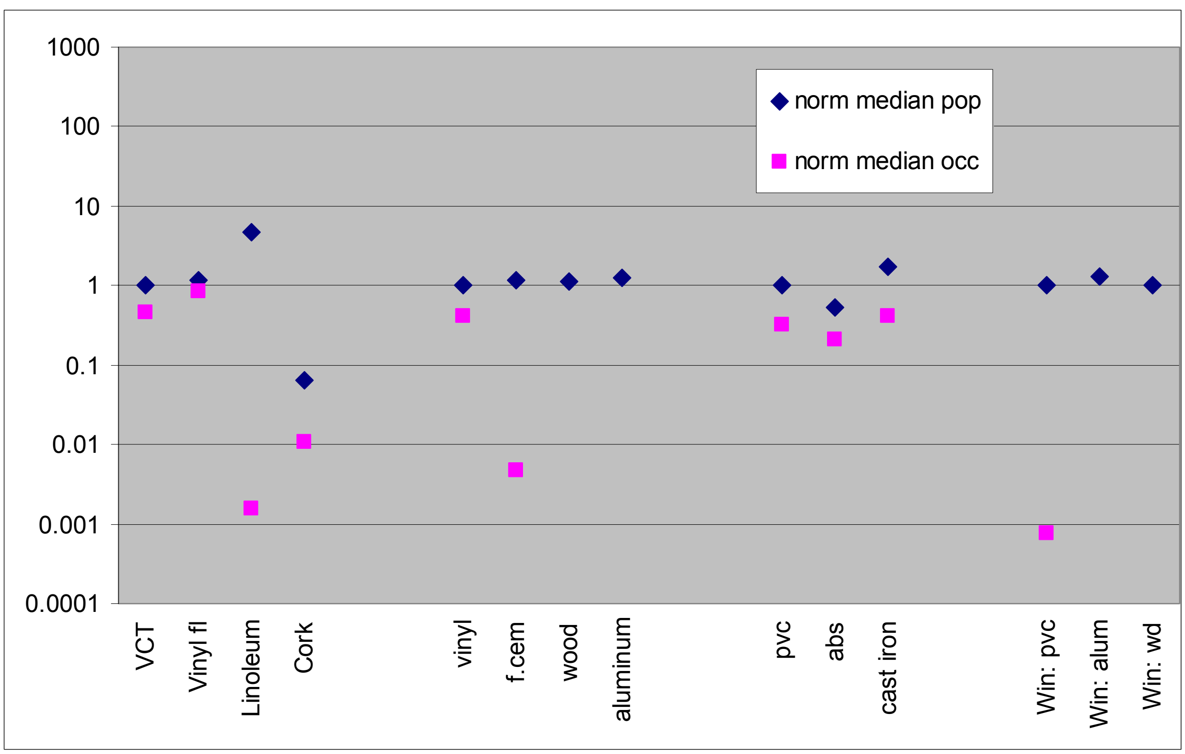

Figure 27. Population and occupational cancer and PM DALYs per f.u. by material type, normalized by population DALYs for PVC materials

Table 6. Cancer and PM-related risks for population and workers per functional unit

Material type		Median pop cancer (micro-DALYs/f.u.)	Median pop PM health impacts (micro-DALYs/f.u.)	Median occ..cancer (micro-DALYs/f.u.)	Median occ..PM health effects (micro-DALYs/f.u.)	% contribution by cancer
Flooring	VCT	2.00E-03	1.40E-01	6.74E-02	0	33%
	Vinyl fl	4.80E-03	1.70E-01	1.24E-01	0	43%
	Linoleum	2.40E-04	6.80E-01	3.88E-05	0	0.04%
	Cork	3.50E-05	9.40E-03	2.65E-04	0	3%
Siding	Vinyl	1.50E-05	1.10E-01	4.50E-02	0	29%
	F.cem	1.50E-04	1.20E-01	0	4.99E-04	0.1%
	Wood	5.40E-04	1.20E-01	0	0	0.4%
	Aluminum	4.40E-04	1.30E-01	0	0	0.3%
Pipe	PVC	5.70E-05	1.69E-01	5.40E-02	0	24%
	ABS	2.26E-04	8.81E-02	3.53E-02	0	29%
	Cast iron	2.32E-04	2.43E-01	6.74E-02	1.58E-03	22%
Window	PVC	1.70E+00	7.20E+03	5.50E+00	3.67E-02	0.1%
	Aluminum	2.20E+00	9.20E+03	0	0	0.02%
	Wood	1.70E+00	7.40E+03	0	0	0.02%

Integrated LCA and Risk Assessment

A second finding, also consistent across all the product alternatives studied, is that the occupational cancer risks *dominate* the population cancer risks posed by the life cycle releases of carcinogens to the environment. While these cancer risks are not found to be very important relative to the particulate emissions, this is nevertheless an important finding for two reasons. First, LCA tends to focus only on population risks and to neglect occupational risks. The approach to estimating occupational risks, and to normalizing those risks to the functional units of product produced, has been based on the best data the Task Group could obtain; but these data are *far* less precise than the data available in life cycle databases on the environmental releases of carcinogens. Thus, it is recommended, outside the scope of the Task Group's charge, that the data coverage and basis for addressing occupational exposures be improved as soon as possible, and that the results of this improved data basis be used to update the analyses and findings in this report. Second, the potentially dominant importance of occupational versus environmental cancer risks provides an important illustration of how some impact pathways can unexpectedly dominate others. This in turn shows the value of an integrated assessment, and the importance of using such an integrated evaluation to assess overall risks from product choices.

A third finding, also consistent across all the product alternatives studied, is that the occupational non-cancer risks *dominate* the population non-cancer risks posed by the life cycle releases of toxicants to the environment, as measured using a simple aggregate hazard index. While we are not of the judgment that these non-cancer risks are particularly important relative to the mortality risks posed by particulate emissions for a variety of reasons discussed elsewhere in this report, this is nevertheless an important finding for LCA in general. LCA tends to focus only on population risks and to neglect occupational risks. This approach to estimating occupational risks, and to normalizing those risks to the functional units of product produced, has been based on the best data the Task Group could obtain; but these data are *far* less precise than the data available in life cycle databases on the environmental releases of toxicants. Thus, it is recommended that LCA modeling and databases stop neglecting occupational exposures, and that the data basis for including these pathways be dramatically improved. These recommendations are not within the scope of the Task Group's charge but are mentioned as areas to review for a research agenda.

3.4 Quantitative evidence: 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. A specific focus has not been taken with regard to residents in neighborhoods adjoining the facilities who manufacture PVC resin or the final building materials. 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).

3.4.1 Ambient Exposure Data

Air Monitoring Data - Kentucky

A review of the data available from the state of Kentucky (KPPC, 2004) was conducted. These data include ambient air concentrations from monitoring stations from 1999-2003 for several volatile organic compounds, including vinyl chloride monomer (VCM) and ethylene dichloride (EDC). Data were available from the following 13 sites: Louisville Police Firearms Training Center (LPFTC), Ralph Avenue/Camp Ground Road, Old Lake Dreamland Fire Station, University of Louisville Shelby Campus, Otter Creek Park, Southwick Community Center, Farnsley Middle School, Chickasaw Park, New Lake Dreamland, Martin Luther King Elementary School, Cane Run Elementary School, and Riverside Gardens. It was not possible to determine the proximity of these monitoring stations to vinyl chloride manufacturing or use facilities in Kentucky using the data available on the website. According to information provided by Wilma Subra (Subra, 2004), there are two vinyl chloride producing facilities 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).

Of the 13 monitoring stations, data on EDC concentrations were only available from 6 sites: LPFTC, Otter Creek, Ralph Avenue, Riverside, Southwick, and UL Shelby Campus. The data are shown in the table below.

Table 7. Ethylene Dichloride Concentrations at Selected Monitoring Stations in West Louisville, Kentucky

Location	Time of Monitoring	Concentration (ppb_v)
LPFTC	11/11/1999	0.08
	11/20/1999	0.02
	11/4/2002	0.02
Ralph Avenue	8/7/1999	0.15
	11/2/1999	0.02
	11/20/1999	0.02
	12/14/2000	0.07
	11/4/2002	0.02
UL Shelby Campus	11/4/2002	0.02
Otter Creek Park	11/18/1999	0.02
Southwick Comm. Center	8/15/1999	150.06
	9/24/1999	0.01
	11/20/1999	0.02
Riverside Gardens	8/18/1999	0.15
	8/28/1999	0.17

LPFTC: Louisville Police Firearms Training Center

The majority of these samples (0.08 and lower) were below the method quantitation limit (MQL). The data indicate that the Southwick Community Center experienced a high spike of 150.06 ppb_v during August 1999, which was quickly corrected, resulting in measured concentrations below the MQL in September and November of that same year. More recent data from this monitoring station were unavailable, so it is not possible to determine if the accidental release or upset conditions that caused the spike were repeated in subsequent years. Concentrations at Riverside Gardens, averaging 0.16 ppb_v for August 1999, are an order of magnitude higher than those found at the other monitoring sites. However, this station also did not have more recent sampling data to indicate more recent conditions. For all of these sites, the dearth of data make conclusive predictions about ambient air concentrations impossible. The data do indicate that EDC concentrations are predominantly low (below the quantitation limit) for this compound. Because the data are old, however, they cannot be used to determine if significant health risks exist from ambient exposures to ethylene dichloride in neighborhoods adjacent and downwind of vinyl plants in these locations.

Data on vinyl chloride concentrations were downloaded and reviewed from the following stations (with the years in which data were available in parentheses): Cane Run Elementary School (June, 2000-September, 2003); Chickasaw Park (June, 2002-November, 2003); Farnsley Middle School (August, 2000-September, 2003); LPFTC (November, 1999-December, 2003); and Ralph Avenue (August, 1999-September, 2003). Mean concentrations of vinyl chloride in air over the entire published monitoring period were calculated. Data were normalized so multiple values from the same date did not skew the overall result. For example, in the event that two samples were reported from the same date, the average of those two samples was used as one sampling event for the date. Similarly, months with multiple sampling dates were normalized so that their values were not overrepresented compared to months with only one sampling date (in all cases, this normalization increased the mean concentration calculated for each monitoring station). These data are limited however in that for some stations, there are only 1-3 samples for certain years. The mean vinyl chloride concentrations are shown below in Table 8.

Table 8. Vinyl Chloride Air Concentrations in Select Monitoring Stations In West Louisville, Kentucky

Station	Proximity to Plant	Years Sampled	Mean Concentration, ppb_v
Cane Run	0.75	2000-2003	0.29
Chickasaw Park	1.5	2002-2003	0.17
Farnsley Middle	1.5	2000-2003	0.18
LPFTC	0.5	1999-2003	0.28
Ralph Avenue	0.75	1999-2003	0.32

As mentioned previously, the air toxics data are generally taken from 1999-2003 for these monitoring stations; data available for individual stations may be very sparse or not comprehensive within that time period. For example, for Cane Run Elementary School, only one sample for vinyl chloride was available for the 2001 monitoring year; this sample was obtained 11/20/2001. Further, in other years, more than one sample may be taken in a particular month, while only one or no samples were taken in different months within that same year. For example, four samples taken in July (two each on the same two days), and one sample the following January. Information justifying the variation in the sampling schedule was not provided by the KPPC on the website. The spottiness of the data makes predictions of trends or knowledge of "concentration spikes" that exceed the applicable air quality standard impossible. As analyzed, the mean chronic concentrations of airborne vinyl chloride are below the air quality standard of 0.47 ppb_v. Because this standard was developed using a conservative limit of cancer risk, mean values that are below the air standard indicate a risk of cancer from vinyl chloride exposure that is below the acceptable standard. Therefore, one can tentatively conclude that the monitoring data do not indicate exposures at the listed stations are of concern.

The data are interesting in that they do not follow an expected trend of decreasing concentrations with increasing distance from the facilities. As can be seen in the table above, the stations that are 0.75 miles from the sites have slightly higher mean VCM concentrations in air than does the LPFTC site, which is 0.5 miles from the site. The sites that are 1.5 miles from the facilities do have the lowest mean concentration, which is expected. However, it is hard to make conclusions about this observation based on unknown factors that may be causing the differences in the data. The reason for the increase in concentration at the midpoint stations (0.75-mile distance) is unlikely to be the result of differences in the sampling of those stations and the ones closer to the vinyl facilities, as they had 29-39 samples each. It may be the result of wind direction, with some monitoring stations getting a more consistent wind flow from the facility to their location. Without data on wind direction at and around the monitoring stations, it is not possible to verify this hypothesis.

Within the time frame of measurement at each monitoring station in West Louisville, there were instances of exceedances of the air quality standard for vinyl chloride. For example the following instances of individual exceedances were noted and the maximum concentration measured: Chickasaw Park,1--0.51 ppb_v; Cane Run, 5 (2 of these values were measured on the same day)--0.88 ppb_v; Farnsley, 1--0.53 ppb_v ; LPFTC,4--1.11 ppb_v; and Ralph Avenue, 3 (2 of these values were measured on the same day)--2.04 ppb_v. Some of the measured exceedances were observed at more than one station on a particular day. For example, the Chickasaw Park and Cane Run Elementary School stations both had concentrations higher than 0.47 ppb_v (values were 0.51-0.78 ppb_v) on 9/12/2003. The concentration measured at LPFTC (0.35 ppb_v) was also elevated. These data indicate an increased release from the vinyl facilities, potentially related to upset conditions. Other coincidental exceedances recorded at 4 out of 5 stations on June 25, 2002 also suggest historical upset releases at the facilities. Nevertheless, because the measurements are not taken from all stations on the same days, firm conclusions are not possible. When 2004 sampling data are made available, it will be possible to assess whether the ambient air concentrations (for 2004) at these locations are below the air standard for vinyl chloride. If this is the case, it might be possible to conclude there is no significant health risk to adjacent communities. If future sampling events were to include more localized, breathing-zone data (via TAGA truck or other sampling methodology, see below) that showed concentrations of VCM or EDC that exceeded the AAS consistently, this conclusion might not be accurate.

Air Monitoring Data - 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); Westlake (Calcasieu Parish).

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 (Sage, 2004).

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). Mean values for each monitoring year were compared to this AAS. Data for those monitoring stations with at least one annual average that approached or exceeded the AAS for vinyl chloride are presented in the table below. None of the remaining stations had an annual average VCM concentration that approached or exceeded the AAS.

Table 9. Vinyl Chloride Air Data for Select Monitoring Stations in Louisiana[a]

Station	Monitoring Year	Mean Concentration, ppb_v	Maximum Concentration, ppb_v	Number of Times Exceedances were Noted in the Year
Lighthouse	2000	0.06	0.43	0
	2001	0.72	30.84	2
	2002	0.08	0.38	0
	2003	0.09	0.41	0
Overall Mean	2000-2003	0.24		
South Scotlandville	1999	0.30	2.63	8
	2000	0.21	1.87	7
	2001	0.33	5.66	10
	2002	0.51	3.77	15
	2003	0.42	8.00	11
Overall Mean	1999-2003	0.35		

[a]All samples were taken on a 24-hour basis.

The data above indicate that although the AAS was exceeded on an annual average basis at both monitoring stations for one year, the overall mean VCM concentrations over all monitoring years were below the AAS. Further, the 2001 data for the Lighthouse station were affected by the presence of two values (including the 30.84 ppb_v maximum) that caused the overall mean for the year to exceed the AAS. If the underlying cause of this sporadic increase in concentration had not occurred, the average would have been much lower, and more consistent with data taken from the other sampling years.

When determining human health risks for chronic effects, it is appropriate to review data over all applicable and appropriate sampling periods, particularly when upward or downward trends in the data can be seen. The Lighthouse data indicate a consistent annual average VCM concentration (with the exception of the 2001 sampling year), with an annual average of 0.24 ppb_v, which is approximately one-half the AAS. Further, the 2001 sampling year was the only one in which daily exceedances of the AAS were detected. The data for South Scotlandville indicate an annual average VCM concentration of 0.35 ppb_v, which is also lower than the AAS. However, this monitoring station showed much more frequent daily exceedances of the AAS (ranging from 7-15), and in general showed an increasing trend in annual average VCM concentrations. Data for the 2004 sampling year will be useful to determine if this upward trend is continuing. The total data accumulated for the last 4-5 years for both sites indicate that ambient air VCM concentrations do not present an unacceptable health risk as defined by the AAS .

Trace Atmosphere Gas Analyzer (TAGA) Data for Louisiana

During June 14-22, 1999, the US EPA evaluated local air concentrations of VOCs at various locations around vinyl manufacturing facilities in Lake Charles, LA. The air concentrations were analyzed using a truck with a Trace Atmospheric Gas Analyzer (Sciex) onboard. The TAGA system records "instantaneous concentration maxima"; in other words, the system does not measure average concentrations over a specified period of time. The following compounds were monitored: vinyl chloride (VCM), benzene, toluene, 1,2-dichloroethylene (DCE), styrene, and xylenes. The TAGA system cannot distinguish between DCE and VCM; therefore, any "hit" attributed to VCM might be caused by EDC, VCM, or a combination of both (US EPA, 1999). The results from the TAGA analysis are shown in the table below.

Table 10. TAGA Vinyl Chloride Analysis, June 14-22, 1999

Date	Location	Concentration, ppb_v	Wind Speed/Direction
June 15	Pete Manena Rd.	15	5 mph; from NE
June 16	I-10	58	Calm
June 16	LA90 between Walcott & Bayou D'Inde	16	Calm
June 17	LA397 at Michigan	16	10 mph; from NE
June 17	Pete Manena Rd.	8	10 mph; from NE
June 21	Pete Manena Rd.	18	4 mph; from NE
June 22	Pete Manena Rd.	48	5 mph; from NE
June 22	Pete Manena Rd.	48	0.3 mph; from E
June 22	Pete Manena Rd.	21	0.3 mph; from E

Other sampling events, 18 in all, taken during the same period, did not show any detectable VCM in air samples. The data above indicate that the measurable concentrations of VCM are generally lower with increasing windspeed and higher when winds are calm. The highest frequency of detects was on Pete Manena Rd. The proximity of this road to neighborhoods is not stated in the TAGA report. These data, however, indicate strong potential for exposure of individuals in these locations. Unfortunately, these data represent the only ambient air concentration data available; more recent data were not found either at the US EPA website or at the Louisiana Department of Environmental Quality website at the time of this writing. Consistent exposure of individuals to these concentrations may indeed pose a significant health risk. Ambient air concentrations obtained during several sampling times within one or more years from neighborhoods located near these manufacturing facilities are needed in order to better assess the potential risk.

3.4.2 Relevance of Ambient Exposure Data for Estimation of Risk to Human Health

When trying to determine the potential adverse effects of a particular industry on the surrounding population, it is always preferable to have accurate, timely exposure data from nearby locations. Monitoring station data provide some indication of the concentrations in air at those locations. Ideal data would be breathing zone concentrations taken as 24-hour average values in the neighborhoods and school yards directly adjacent to these manufacturing facilities in Louisiana, Kentucky, Texas, and other states. The TAGA data provide a glimpse of environmental conditions in 1999, but they cannot be used to accurately predict trends in exposure and any risk estimates developed using them would be associated with a great deal of uncertainty.

As discussed above, more accurate data are necessary to predict the likelihood of a long-term resident adjacent to these manufacturing facilities to develop cancer or non-cancer health effects as a result of chronic exposure to emitted compounds. One charge that has been made with regards to emissions from these facilities is that the compounds emitted, VCM and EDC, are cancer-causing compounds and are associated with cancers that have been observed in occupational workers at vinyl manufacturing facilities (Subra, 2004). In light of this charge, it is useful to evaluate Louisiana's cancer incidence and cancer deaths as compared to national averages.

Cancer in Louisiana

According to the Louisiana Tumor Registry data, incidence rates for all cancers combined in Louisiana (and the Industrial Corridor, which encompasses these vinyl manufacturing facilities), are the same as, or even lower than, US rates, with the exception of white males. In fact, Louisiana cancer deaths peaked at approximately 242 per 100,000 residents in 1991, and have declined steadily since that year (NCI, 2004). Because tobacco use in white males in Louisiana exceeds that of white males nationwide, it is believed that this contributes to the increased cancer incidence in this subset of the population. Lung cancer accounts for 20% of all cancers in Louisiana males (Stevens, 2004). These data are shown more dramatically in the table below.

Table 11. Comparison of Louisiana to National Incidence and Mortality Rates, All Cancers Combined and Lung Cancer, 1988-1992

	Caucasian				Black			
Type	Male		Female		Male		Female	
	I	M	I	M	I	M	I	M
Louisiana								
All Cancer	469.8	240.0	309.7	140.6	515.5	339.4	307.0	175.6
Lung	108.6	91.3	44.4	35.8	131.4	124.9	40.1	33.8
U.S.								
All Cancer	484.2	212.8	351.4	140.0	585.5	318.0	337.3	168.0
Lung	80.2	72.5	42.5	31.8	124.3	105.3	46.8	31.8

I=Incidence Rate
M=Mortality Rate
All rates are age-adjusted; from NCI National Database
Data obtained from Louisiana Cancer and Lung Trust Fund Board

Although these data are quite dated, they do provide some important information about cancer incidence in Louisiana residents. First, if general population exposure to carcinogens was significant, one would expect the cancer incidence in the Louisiana population to be higher than that of the rest of the nation, for those cancers associated with VCM and EDC exposure. This does not appear to be the case. On the contrary, the data above show that Louisiana residents have a slightly lower incidence rate for all cancers, but a higher mortality rate than the national average, with the exception of mortality from all cancers in Caucasian women. The data are particularly striking when one looks at incidence rates in blacks in Louisiana; the incidence rates for all cancers is lower than the national average, but the incidence of lung cancer in males, particularly white males, exceeds that of the national average.

Evaluation of selected parishes indicates that lung cancer accounts for the majority of cases reported, followed by breast, colon, and prostate cancers. In none of the parishes reporting (Acadia, Evangeline, Iberia, Lafayette, St. Landry, St. Martin, St. Mary, Vermilion) was liver cancer listed as a cancer site (LTR, 2004). However, it is acknowledged that these parishes are not the ones in which the vinyl and other chemical plants are predominantly located.

A more complete data set was obtained from the National Cancer Institute's SEER (Surveillance, Epidemiology, and End Results) Program (NCI, 2004). The annual incidence rates of liver and bile duct cancers in Louisiana parishes over 1997-2001, compared to Louisiana as a whole, and the rest of the U.S., are shown below.

Table 12. Incidence Rate for Liver and Bile Duct Cancers in Louisiana Parishes

Location	Annual Incidence Rate (per 100,000)	Average Annual Count	Rate Period
Louisiana	6.1 (4.7, 7.8)*	67	2001
Louisiana	5.2 (4.6, 5.8)	55	1997-2001
U.S.	5.9 (5.6, 6.3)	**	2000
Jefferson Parish	8.9 (5.5, 14.1)	5	1997-2001
Orleans Parish	7.8	20	1997-2001
E. Baton Rouge Parish	5.3 (3.5, 7.8)	6	1997-2001
All Other Parishes	Suppressed	3 or fewer	1997-2001

Data were for both sexes, and all races
* (95% confidence interval)
**Data not provided due to quality control issues.

As seen in the above table, three parishes had, on average, 5 or more annual cases of liver/bile duct cancer from 1997-2001. The annual incidence rate in East Baton Rouge Parish was comparable to the rate throughout Louisiana and over the U.S. The rates in Jefferson and Orleans Parishes were increased compared to both the state and national incidence rates. The rates for all the other parishes were suppressed as they were too low to be reliable. These data indicate that the parishes with the highest density of vinyl plants do not exhibit the highest incidence rates of liver cancer.

An analysis of cancer death rates and trends (through 2001) in Calcasieu Parish for all races and both sexes of residents, as compared to the rest of the state, reveals the following: colon and rectum cancer deaths in females are higher than those in the rest of the state but stable (not rising or falling), breast cancer in females is stable and similar to that in the rest of the state, liver and bile duct cancer in males is stable and similar to that in the rest of the state, and leukemia in both sexes and non-Hodgkin's lymphoma in males are stable and similar to the rates in the rest of the state (NCI, 2004). Stomach cancer deaths in males in Calcasieu Parish are similar to the rest of the state, but falling in frequency. In Iberville Parish, death rates from lung and bronchus cancers in females and males are similar to the rest of the state but rising in females and stable in males. Death rates from other cancers, including breast cancer in females, leukemia in both sexes, and liver cancer in both sexes, and childhood cancers, are so low in Iberville Parish that the data have been suppressed to preserve confidentiality and stability of the rate estimates.

The populations in the parishes of the Industrial Corridor often have a predominantly high black population and are often poorer than those in the rest of the state and the country. The high cancer mortality rate in the state has been attributed to late diagnosis, which results in less effective treatments, lack of access to care, preventive measures and early detection. Under-utilization of cancer screening programs by Louisiana residents has been cited as a contributing factor to the high cancer mortality rates in the state (Stevens, 2004).

Childhood cancer mortality rates in Louisiana are lower than those of the US. For example, 1996 mortality rate data indicated that cancer is the third leading cause of death in children aged 1-14 years in Louisiana, accounting for 7% of total deaths in children. Cancer is the second leading cause of death nationwide, accounting for 10% of all deaths in this age group (LTR, 2004). Further, cancers in children represent only 0.7% of all cancers in Louisiana residents (LTR, 2004). When childhood (age 0-14) cancer death rates were analyzed through 2001, it was found that the death rate from cancer in Louisiana counties were similar throughout the state, the death rates were similar to the national average and were falling compared to those in the rest of the United States (NCI, 2004). If you break those data out by sex, the cancer death rate in male children is below the U.S. rate and falling, while that of female children is similar to the national rate and falling (NCI, 2004).

One could argue that past exposures of the general population to emitted VCM or EDC were too recent for any cancers to appear and be counted in the state and national cancer registries. 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. Residents of Mossville were found to have high blood levels of dioxin, compared to those in the rest of the country. 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 in either children or adults (ATSDR, 2002). Nevertheless, future monitoring of the exposed populations with regard to cancer incidence might yield valuable cancer dose-response information.

3.4.3 Summary

Available airborne sampling data from Kentucky and Louisiana indicate the presence of VCM and EDC in air samples nearby and downwind of vinyl manufacturing plants. Annual average concentrations of VCM measured at selected monitoring stations have generally been below the AAS in both states, although annual average VCM concentrations have exceeded this health standard for at least one year at two different monitoring stations in Louisiana. It is not clear that data from all monitoring stations in Louisiana were provided in the submitted data (Sage, 2004). Data taken closer to the breathing zone of residents adjacent to the facilities in the Lake Charles area show very high concentrations of VCM (8-58 ppb_v) which greatly exceed the AAS of 0.46 ppb_v. Additional data are needed to determine the potential health risk of individuals living in neighborhoods near vinyl manufacturing facilities. Current air monitoring data do not indicate a health risk, as mean air concentrations over all sampling years are below the AAS. Louisiana cancer incidence rates are generally below the national average, but mortality rates exceed the national average. Late detection of cancers, and reduced access to screening and treatment are believed to be the biggest contributors to the higher death rates in Louisiana. Childhood cancer rates in Louisiana are lower than the US national rates. Parish-specific cancer site data do not currently indicate a higher rate of cancers, compared to national averages, in sites associated with exposure to compounds used in the manufacture of vinyl building materials. Future data on cancer incidence might address the issue of latency (lag time for the development of cancer following exposure) for populations more recently exposed to environmental VCM, EDC and dioxin.

3.5 Discussion of morbidity

3.5.1 Morbidity—cancer

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).

Phthalates

Phthalates, particularly di(2-ethylhexyl)phthalate, are used to make vinyl products flexible. DEHP is the most commonly used phthalate in vinyl flooring, while shorter side chain phthalate moieties, such as butyl benzyl phthalate (BBP), dibutyl phthalate (DBP) and diisononyl phthalate (DINP), are found in many consumer products including toiletries, cosmetics, and flexible children's toys. DEHP was used as a model plasticizer in this analysis, because it is most widely used in the flooring materials examined in this report 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). 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. Humans and non-human primates have a low level of PPARα expression (Palmer et al., 1998) 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).

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). 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.

3.5.2 Morbidity—non-cancer

Mercury

Currently nine chloralkali plants producing chlorine for the PVC industry use outdated mercury cell technology (U.S. Senate, 2004). The market share of the plants using this technology in 1999 was 12% (EIA, 2000). 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). 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 (TRI; NRDC, 2004). According to the U.S. EPA's Final Rule for NESHAPS on Mercury (U.S. EPA, 2003), 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, 2003), 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, 2003).

Currently, the ACGIH TLV for mercury is 0.025 mg/m^3, a level which is documented to result in measurable, but sub-clinical, signs of neurotoxicity in exposed workers (hand tremor, increases in memory disturbance, slight evidence of autonomic dysfunction; U.S. EPA IRIS, Hg; 1995). Mercury exposures have varied over several orders of magnitude at chloralkali plants (Williams et al., 2001; Ellingsen et al., 2000) and current exposure data are not available. Occupational exposures to mercury in foreign chloralkali plants are not quantifiable at this stage, but are likely to be high, particularly in developing countries or countries with less strict industrial hygiene and environmental regulations. Mercury contamination of the environment results in bioconcentration of methylmercury in animals, particularly fish and birds. Consumption of fish contaminated with methylmercury is of concern to the U.S. government, as ingestion of contaminated fish by women of childbearing age and by young children could contribute to the incidence of children with delays in neurological development (NIEHS, 1997).

Given that it is known that the U.S. chloralkali plants have recently "lost track of" 65 tons of mercury into the environment, and the potential for further significant releases exists, the production of chlorine using mercury cell technology presents an environmental problem that is unique to the manufacture of PVC. The impact of metals on the general public is not well addressed using current TRACI methodology; therefore, the LCA results presented in this report do not assess the potential risk of neurotoxicity and potential resultant mortality from exposure to this chemical.

It has been estimated that approximately 10% of chlorine is manufactured using mercury cell plants (EIA, 2000). Based on the assumption that 10% of PVC resin is made using this chlorine, the non-cancer risk was estimated for all PVC resin manufacturing workers to be approximately 8 (using the ACGIH TLV of 0.025 mg/m^3 as the starting point). Although this risk value is small compared to the potential risk from exposure to VCM and EDC, it has already been stated that occupational exposure to the TLV is associated with measurable neurological deficits. Therefore, the continued use of mercury cell technology for chlorine production represents an unacceptable risk to occupational and human health, especially because chlorine is available from plants in the U.S. that do not use this technology.

Phthalates and Endocrine Disruption

Of particular concern to interested stakeholders with regard to exposures to phthalates is the potential for endocrine disruption and reproductive effects. DEHP and dibutyl phthalate are reproductive toxicants in both sexes of rodents (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 "helper" 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 building 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. As discussed above, the available data indicate that occupational exposures to DEHP do not result in decreased reproductive success.

The available data do not indicate that DEHP is estrogenic. DEHP was shown to not 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).

The European Commission on Health and Consumer Protection Directorate-General recently released a revised risk assessment of DEHP (CSTEE, 2004) in which it cited a draft version of a 2003 three-generation study in Sprague-Dawley rats (Wolfe and Layton, 2003). A NOAEL for testicular toxicity and developmental toxicity for this study was identified as 4.8 mg/kg-day; the LOAEL was 14 mg/kg-day for decreased testicular weight, small or aplastic testes, seminiferous tubular atrophy, and infertility at high doses. This LOAEL is two orders of magnitude lower than that resulting in hormonal differences and anovulation in female rats. It is noted that a copy of this study was not available for review by the Task Group.

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 (NTP-CERHR, 2000) is correct. This daily dose is based on a 1mg/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 of 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, 1993, as cited in CSTEE, 2004). Interestingly, the concentration of di-n-butylphthalate was significantly higher, with a mean value of 630 ng/m^3 in the daytime. The DEHP values were comparable in the Southern California study to the median DEHP value (77 ng/m^3) reported in 120 Cape Cod homes by Rudel et al. (2003). If a female worker was exposed to DEHP both at work and at home, her daily dose would only be 0.286 mg/kg-day (assumed from occupational exposure); the dose received at home is negligible at 18.75 ng/kg-day (or 1.875E-05 mg/kg-day) assuming an inhalation rate of 0.8 m^3/hr (intermediate between sedentary and light activity), 15 hours spent inside the home, and a body weight of 64 kg (EFH, 1996). This exposure assessment does not take into account ingestion of DEHP or exposure via any other source. The daily dose of 0.286 mg/kg-day has a margin of safety of approximately 17; however, an individual not exposed occupationally would have a margin of safety (assuming inhalation exposure only) of 2.56E5. These data indicate that occupational exposures to DEHP should still be regulated and controlled, although it must be noted that the greatest exposure to phthalates, including DEHP, is believed to be through the diet, followed by indoor air (NTP-CERHR, 2000).

Infants, particularly those who receive large doses of phthalates as the result of medical treatment, are considered a susceptible subpopulation to the reproductive toxicity of phthalate exposure (CERHR, 2000; Lovekamp-Swan and Davis, 2003). Rodent studies have shown that DEHP and DBP have anti-androgenic properties *in utero* and affect reproductive development of the male reproductive system, particularly when exposure occurs early in development (Moore et al., 2001). Studies have not shown an effect on female reproductive development (Moore et al., 2001). Further, *in vitro* studies show these compounds or their primary metabolites have little or no estrogenic activity (Harris et al., 1997; Picard et al., 2001). As discussed earlier, only one study was identified that assessed pubertal development in children who were most likely exposed to high doses of DEHP (as high as 42-140 mg/kg) for a short period early in infancy (Rais-Bahrami et al., 2004). The authors showed that exposure did not affect thyroid, liver, and kidney function or reproductive development in either male or female adolescents.

Exposure of pregnant women to DEHP and other phthalates, either occupationally, or via food and other sources, is of concern, as *in utero* exposure of rodents is shown to affect reproductive development of male offspring (Moore et al., 2001). Therefore, blood burden of DEHP and MEHP of pregnant and non-pregnant animals provide some information as to the potential for exposure of the fetus to these compounds. Kessler et al. (2004) dosed pregnant and nonpregnant rats and marmosets repeatedly throughout gestation (gestation days, gd, 14-19 for rats, and gd 96-124 for marmosets) with 30 or 500 mg/kg-day DEHP. It was shown that the blood areas under the concentration-time curves (AUCs) of DEHP (essentially a measure of the systemic body burden of DEHP) were two orders of magnitude lower than those of MEHP in rats and one order of magnitude lower than those of MEHP for marmosets. Further, maximum bloodstream concentrations of MEHP in marmosets were 7.5 times lower, while the AUCs were up to 16 times lower, than in rats receiving the same daily oral DEHP dose (per kg body weight). If it is reasonable to assume that maternal doses can serve as a surrogate to fetal doses, these data suggest strongly that extrapolation of rodent toxicity data to humans may be overly conservative. These data suggest that primates have lower MEHP body burdens than do rodents, when the parent compound is administered at the same body weight basis. Preliminary data from Latini et al. (2003) indicate that concentrations of MEHP in human fetal cord blood do not correlate with those of MEHP or DEHP in the maternal bloodstream. Therefore, additional studies are necessary to determine the relevance of rodent reproductive and developmental toxicity studies to humans.

Of great interest is the higher concentration of airborne dibutylphthalate (630 ng/m^3) in indoor air of California homes cited above compared to that of DEHP. This may be due to the use of DBP in other consumer products such as toiletries and cosmetics. The relative concentrations are consistent with the respective excreted amounts of their monoesters in human urine (Blount et al., 2000). In that study, part of the NHANES III survey (Third National Health and Nutrition Examination Survey), geometric mean MBP levels in urine were 36.9 µg/g creatinine, with a maximum value of 2,760 µg/g creatinine. The same values for MEHP were 3.0 and 192 µg/g creatinine, respectively. These data indicate that although DEHP is more widely used in building materials, humans may actually be exposed to and absorb more DBP from their environment.

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 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. There were 175 children with an allergic condition, and 177 controls in this study. The study authors report that butyl benzyl phthalate 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 of children with these symptoms (diagnosed by a doctor) was 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 effecting 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.

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.

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. By contrast, Larsen et al. (2002) found that phthalates with 8 or 9 carbon atoms as side chains were effective at inducing an allergic response in mice that had been injected with the plasticizers. Phthalates with shorter or longer alkyl side chains were not as effective. The relevance of these data to inhalation exposures in man is unknown.

Additional studies regarding asthma and other respiratory diseases in children and adults are needed to help inform the relationship between phthalates and asthma. Current data in children suggest a link between phthalate exposure and allergic reactions, but they are not supported by current animal studies.

4 Data Gaps

4.1 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 a completely unique set of values. 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. Lack of more accurate data does not negate the findings or lessen their significance.

The most significant data gap in the estimation of human health non-cancer and cancer risk estimates is the lack of accurate exposure data. This lack of information is consistent across all the building materials, although 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 prior to June 2004. 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 OSHA PELs as a means to estimate upper bound risks for adverse health effects. Although these values can be substituted for more accurate exposure data in the equations used to estimate risk, they can result in very high estimates of risk. The reason for this is that 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, and therefore do not have to be met by industry. The exception to this is when no PEL value was available. Many industries do try to minimize exposures to hazardous chemicals in the workplace such that they meet or even exceed ACGIH TLVs. It was impossible, however, to determine if all the manufacturing facilities producing the building materials investigated here were successful in achieving those goals. 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 judgement 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. and it was not presumed that the same industrial hygiene practices that are common here would be in practice in Europe.

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

Additional data gaps are present in the lack of knowledge of all compounds to which an occupational worker may be exposed within an industry and whether or not personal protective equipment is used. For example, it was not known if respirators are commonly used to prevent human inhalation exposure to coke oven emissions and heavy metals in iron and steel foundries.

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.

4.2 Data gaps in process data

For the life cycle analysis component of this report, process-based life cycle inventories for all pollutant species (or as many as possible) were required to avoid underestimating impacts. Emissions data and process information were sought through various search engines and government websites.

A few of the sources that were found in published articles and government sources proved not to be useful for this study. They were usually not process specific or were incomplete in listing all the important pollutant species or parameters needed for analysis. For example, the annual emissions data from a plant cannot be converted to an emission factor (kg pollutant per kg of product) without the annual production information for each material produced in the plant. For the same reasons, the published emission data sources such as the Toxic Release Inventory (TRI) and AirData, which are plant or industry-specific, are not usually adequate for process-based analysis (U.S. EPA, 2000a; U.S. EPA, 2000b). Therefore, LCA tools that contain LCI databases, namely BEES and SimaPro, were used as the basis for this analysis.

BEES has data collected from the U.S. manufacturers of a various building materials (NIST, 2002). On the other hand, SimaPro mainly contains data from the European sources (APME, 1997/1998; APME, 1997a; APME, 1997b; APME, 1997c; APME, 1998; APME 1999; BUWAL, 1998; ETH Zurich, 1996; Kemna, 1981; IDEMAT, 2001; PRé Consultants, 2001a; PRé Consultants 2001b; PRé Consultants, 2004; RIVM 1992; RIVM 1993). Although European data were modified whenever possible to reflect the U.S. energy-related emissions based on the Franklin data (also available in SimaPro), any processes that are different from the corresponding U.S. processes and/or have different emission intensities would lead to underestimation or overestimation of impacts (Franklin Associates, et al., 1998).

Also, the confidence intervals could not be derived from our results for lack of enough data points to represent the entire scale of alternative processes in the US. Some products have more alternatives than others, depending on the number of available sources. Fewer sources often lead to a relatively narrow range of impact estimates, but they should not be understood as representing a higher degree of precision in the results.

4.3 Data gaps in emissions data

As described in section 2.2.1, the characterization factors for chemical groups such as unspecified polycyclic aromatic hydrocarbons (TPAHs), phthalates (Tphthalates), aldehydes (Taldehydes), and dioxins (Tdioxins) were derived with some simplifying assumptions (these are separate from any of the specific compounds modeled).

The weighting factors of TPAHs for cancer and non-cancer impacts were estimated using the toxicity values (e.g., Cancer Slope or Potency Factor, CSF [CPF], and Reference Dose (RfD) and intake fractions of the species emitted from four major sources as identified by the U.S. EPA (1996; 1998). Thus, the characterization factor of TPAHs represents the average weighting factors for those four sources, as (i.e., wood burning, utility, aluminum, wildfire). This may be a reasonable assumption for cancer impacts because the weighting factors vary relatively little between the sources. However, for non-carcinogens (especially water-borne emissions), the weighting factor varies by orders of magnitude between sources. Therefore, for the products whose non-cancer impacts are contributed largely by water-borne TPAHs, it would be important to reevaluate the impacts using the process-specific weighting factors rather than the average.

Another important source of uncertainty is the number and types of species included in TPAHs, Taldehydes, Tphthalates and Tdioxins. As mentioned in Section 2.3.3, with the exception of human health cancer and non-cancer characterization factors for PAHs, the derived characterization factors for all those groups of chemicals are the simple averages of the TRACI factors. By taking the average of those characterization factors, it is assumed that the constituents in each chemical group are in the same proportions and that no other species are contained in the group. For the characterization factors TPAH's, both high (40 species including 7 carcinogens, 9 non-carcinogenic toxic pollutants and 24 non-toxic pollutants) and low (7 carcinogens only or 9 non-carcinogenic toxins only) values for the number of species have been tested. Consequently, these alternative assumptions have contributed to the variability of the impact levels. For example, if it is assumed that the group of TPAHs consists of 40 species instead of only 7 carcinogens, the cancer characterization factor of TPAHs decrease by a factor of 7, leading to a lower cancer risk estimate.

In the current analysis, both scenarios were tested for cancer and non-cancer effects, but for better estimates more detailed information about the "PAHs" in terms of types and species will be useful.

For the same reason, the size fraction of "dust" as a pollutant group may be important. Epidemiological studies show that fine particles (PM2.5) are more harmful than coarse particles for particulate matter- related health impacts. Therefore, dust needs to be defined better in terms of size fractions so that a better characterization factor may be developed.

Due to the lack of detailed emissions data for specific phthalates, aldehydes and dioxins, these pollutants were weighted by toxicity and intake fractions but assuming an equal amount of each of the constituents. The table below shows the relative weighting factors for phthalates, aldehydes and dioxins in air and waterborne emissions.

Table 13. Relative weighting factors in air and waterborne emissions

	C-Air	C-Water	NC-Air	NC-Water
Aldehydes:formaldehyde eq	1924%	49080%	415%	2022%
Phthalates: DEHP eq	17%	17%	85%	52%
Dioxins: 2,4,7,8-TCDD eq	19%	27%	10%	11%

To better weight those unspecified chemical groups, it is essential to know the number and types of the constituents. It may be worthwhile to revisit this issue once such data have been collected and become available.

4.4 Data gaps for exposure analysis

For the exposure estimates, intake fractions for air toxics and particulate matter (as incorporated in TRACI) were used (Bare et al., 2003; 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. but does not account for local impacts. Using the average intake fractions may lead to underestimation of risks if, for example, sources are located in or near a densely populated area, or if the climate and landscapes allow pollutants to stay in the populated area for a long time. On the other hand, it may lead to overestimation of risks if most of the exposed population is located upwind, or if the source is located near the ocean where most of the emissions can be washed away from the continental U.S. by wind or tide. Likewise, although the particulate matter intake fractions take into account the population density around sources, they are not useful for estimating the risks of the local population for the same reasons.

Metals

Intake fractions for metals have not been evaluated. Therefore this report's analysis could not take into account the impacts from metals. Alternatively, human toxicity potentials for metals can be used to estimate metal-related impacts once the travel distance information is obtained from the author of these toxicity potentials, Edgar Hertwich (Hertwich EG, et al, 2001). However, the model uncertainty for metals would lead to uncertain results that are hard to be verified.

IAQ

The exposure levels to pollutants in homes emitted from indoor materials are hard to estimate. Although Lai and others (2000) have shown that intake fractions for the indoor air particles are about 1000 times higher than those for particles emitted outdoors, the indoor intake fractions for individual air toxics have yet to be developed. Also the material emissions information for each species is required to estimate the associated risks.

4.5 Additional 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 is discussed further in Section 4.1.

5 Conclusions and Recommendations

Conclusions & Recommendations

The Task Group draws the following conclusions from the research described in this report:

1. Using current data for LCA and risk assessment, our analysis of the chosen building material alternatives shows that PVC does not emerge as a clear winner or loser. <u>In other words, the available evidence does not support a conclusion that PVC is consistently worse than alternative materials on a life cycle environmental and health basis.</u> With the possible exception of cork flooring, neither is any material dominantly best across all impact categories. Instead, there are some impact categories in which each material performs poorly and others in which it performs well. This result holds up even though we have subjected both the LCA and the risk assessment portions of the analysis to sensitivity analyses. <u>Therefore, the *current body of knowledge* as analyzed in this report in Section 3 as it relates to the Task Group's charge from TSAC does not support a credit in the LEED rating system for eliminating PVC or any particular material.</u> Further, with respect to a PVC-related credit, the available evidence indicates that for some product categories, such a simple credit could steer designers to use materials which performed worse over their life cycles with respect to the bulk of the impact categories.

2. Section 4 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.

3. The Task Group's charge includes "…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." Based on its analysis of current quantifiable data, the Task Group concludes that material-based credits that discourage the use of specific materials are unnecessarily "blunt instruments." Rather, human and environmental health would be better-served by developing credits based on issues or impacts such as those on human health and ecosystems.

4. While credits that promote or discourage the use of specific materials across a range of applications may be suspect, the analysis in the report has shown that within a given product category predicted health and environmental impacts of material alternatives can differ widely. Thus, within product categories more and less desirable materials can be identified.

The foregoing conclusions represent the Task Group's advice to the LEED Steering Committee in response to its charge. Additionally, based on its work on this issue the Task Group has developed certain policy recommendations that would build upon the analysis techniques developed in this report and would help to fill the gaps in knowledge identified herein. These recommendations are separate from the conclusions and should be weighed with other factors as they are considered by the Steering Committee. Specific recommendations include the following:

- The literature is overwhelmingly PVC focused, and additional research needs to be done on the risks associated with alternative materials to move towards comprehensive comparative analysis. 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, lead the Task Group 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.

- In the long-term, the Steering Committee is encouraged to consider developing issue-based credits in conjunction with other developments under way for LEED version 3.0. For such issue-based credits to be technically sound, they should be informed by both LCA and risk assessment. The Task Group is willing to share and develop examples of this approach.

- In the shorter term, the Steering Committee should consider allowing Innovation and Design credits in current versions of LEED for projects that fill data gaps and provide credible evidence that is in keeping with the findings of this report and that furthers the body of knowledge toward lowering health impacts of building materials. The Task Group recommends beginning this effort by working toward eliminating a class of pollutants and/or particulate emissions associated with building materials. This information can be used to support the development of scientifically sound credits for future versions of LEED and will provide incentive for project teams and others to research the issues that could make such credits scientifically robust. These ID credits and related CIRs should be reviewed against the integrated methodology outlined herein in order to maintain consistency with the findings of this report.

Appendix A: Acronyms

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
CDC	Centers for Disease Control
CDD/CDF (PCDD/PCDF)	Polychlorinated dibenzodioxin/dibenzofuran
CERHR	Center for the Evaluation of Risks to Human Reproduction
CPF/CSF	Cancer Slope Factor
DEHP	Di(2-ethylhexyl)phthalate
EPC	Exposure Point Concentration
HHPM	Human Health Particulate Matter
HHC	Human Health Cancer
HHO	Human Health Other
HI/HQ	Hazard Index/Hazard Quotient
ILCR	Integrated (Excess) Lifetime Cancer Risk
IRIS	Integrated Risk Information System
LCA	Life Cycle Assessment
LOAEL	Lowest Observed Adverse Effect Level
MEHP	Mono(ethylhexyl)phthalate
MOE	Margin of Exposure
NIOSH	National Institute of Occupational Safety and Health
NOAEL	No Observed Adverse Effect Level
NTP	National Toxicology Program
OSHA	Occupational Health and Safety Administration
PAH	Polycyclic Aromatic Hydrocarbon
PEL	Permissible Exposure Limit
PVC	Polyvinyl chloride
RfD/RfC	Reference Dose/Reference Concentration
TLV	Threshold Limit Value
TRACI	Tool for the Reduction and Assessment of Chemical and Other Environmental Impacts
TWA	Time-Weighted Average
U.S. EPA	U.S. Environmental Protection Agency
VCM	Vinyl chloride monomer
WHO	World Health Organization

Appendix B: Life Cycle Analysis Modeling Assumptions

Assumptions for SimaPro: DWV Pipe

For the SimaPro modeling, the following alternative assumptions were made for each pipe type.

ABS alt1

- ABS processes reported by APME (1997a)
- Extrusion process for PVC is assumed

ABS alt2

- ABS process data based on LCA for the production of 1 kg of ABS in Europe (APME, 1999). Average data for 1995.

ABS alt 3

- ABS process data based on information from 5 plants in Germany, Italy and the Netherlands, producing 360 000 tonnes in 1995 (APME, 1999).

ABS alt 4 – alt 7

- ABS process data based on LCA for the production in the Netherlands. Average data 1992. General purpose mix 25/20/55 with 30% glass fiber (plus 4 different process data for polybutadiene assessed) (APME, 1997b).

PVC alt1

- PVC process data based on FAL (Data for the material and energy requirements and process emissions for the production of 1000 pounds of Polyvinyl Chloride (PVC). Average USA technology, late 1990's.)
- PVC extrusion process assumed (APME, 1997c).

PVC alt2

- Process data based on AMPE (1997c) -- Production of PVC extruded pipe including packaging. Data from three factories in the Netherlands producing PVC and PP pipes. For PVC pipes the stabilizers are treated as homopolymers.

PVC alt3

- Process data published by APME (1999) --Production of PVC pipes, including production of PVC resin, transport of the resin to the converter, the conversion process itself and packaging of the finished product for onward despatch. In pipe extrusion the molten polymer is extruded through an annular die and cooled by passing through a water trough. The effects of stabilisers have been ignored so that in the calculations all of the weight of the pipe is assumed to be PVC homopolymer. Data from 3 plants in the Netherlands (1995), producing 60.000 tonnes.

Cast iron alt1

- Based on "Cast iron ETH", which represents "Western Europe's average technology, " the process was modified to replace the crude iron contents with scrap iron in Simapro.

Cast iron alt2 - alt5

- The processes in SimaPro for the other types of cast iron with various contents of manganese, silicon, Cr, Ni and Co were modified to replace the crude iron contents with scrap iron (RIVM, 1992).

Assumptions for SimaPro: Siding

Simapro's life cycle inventory for building materials is mainly European. On the other hand, the Franklin database provides U.S. LCI data for energy-related processes as well as products such as PVCs and SBR. To simulate the U.S. life cycle of most of the materials, we replaced the European data whenever possible with U.S. data (FAL, 1998).

In the list below, LCI specifications are on the left side, and the corresponding modeling assumptions are described on the right.

Aluminum siding based on BEES

- Aluminum – aluminum with 0%~100% of recycled contents (ETH, 1996; BUWAL, 1998)
- PVC thermoset topcoat -- no proxy for thermoset top coat is available. 100% PVC is assumed (FAL, 1998)
- Manufacturing energy not described in BEES – "aluminum extrusion process (electricity)" is assumed (Duin, 1998)
- Energy for cutting not described in BEES -- 1.186 MJ/m2 electricity, or 0.11018345 MJ/ft2 (FAL, 1998, PRé Consultants, 2001a)
- Aluminum nail – 0-100% recycled aluminum (ETH, 1996; BUWAL, 1998)

Vinyl siding based on BEES

- Process energy not specified -- extrusion process of PVC (APME, 1997c) is assumed
- Galvanized nails – a 6d nail has approximately 4.3 cm2 of surface area. 0.036kg zinc galvanization is assumed required per 1m2 of steel surface. 50% scrap steel for nail and the electricity requirement for galvanization is assumed to be 1.54 MJ/kg nail. (BUWAL, 1998; ETH, 1996)

Wood siding based on BEES

- Stain -- alkyd varnish ETH (ETH, 1996)
- latex primer -- paint
- cedar wood siding -- wood board ETH (ETH, 1996)
- Galvanized nail -- 6d (modeled for vinyl siding) (BUWAL, 1998; ETH, 1996)

Fibercement roofing (proxy for siding) based on BEES: alt1

- Portland cement – cement ETH
- Wood chips as organic fiber -- cellulose cardboard less chlorine(BUWAL, 1990)
- NG Production energy -- "Heat from nat. gas FAL, 1998" See above for the SP description.
- Galvanized nail -- 6d (modeled for vinyl siding) (BUWAL, 1998; ETH, 1996)

Fibercement based on Certainteed's MSDS: alt2

- Calcium silicate (hydrate) 30-55% -- clinker 36.6% (IDEMAT, 2001)
- Crystalline silica (quartz) 45-55% -- silicate (waterglass) ETH 49.4%
- Unbleached cellulose fiber 5-15% -- cardboard cellulose less chlorine 9.4% (BUWAL, 1990)
- Hydrous aluminum silicate (kaolin or ball clay) 2-8% -- kaolin 4.6% (BUWAL, 1996)
- NG Production energy (based on BEES fiber-cement roofing manufacturing energy)-- "Heat from nat. gas FAL," which is defined as "data for the cradle-to-gate resource requirements and emissions for the combustion of 1000 cubic feet of natural gas (1.03 Million Btu) in industrial boilers. Average USA technology, late 1990's.

- Galvanized nail -- 6d (modeled for vinyl siding) (BUWAL, 1998; ETH, 1996)

Fibercement based on MSDS for Hardie Building Products including Singleside, Hardiboard, Hardisoffit, etc.): alt3

- Calcium silicate (hydrate) 50-60% -- clinker 55% (IDEMAT, 2001)
- Crystalline silica (quartz) 35-45% -- silicate (waterglass) ETH 40%
- cellulose: <10% -- cardboard cellulose less chlorine 2.5%(BUWAL, 1990)
- other fillers (non hazardous ingredients): <10% -- kaolin (based on Certainteed's MSDS)2.5% (BUWAL, 1996)
- NG Production energy (based on BEES fiber-cement roofing manufacturing energy)-- "Heat from nat. gas FAL, 1998 " See above for the SP description.

Assumptions for SimaPro: Flooring

Some products are not available in SimaPro (e.g., vinyl acetate, DEHP, etc). We tried to substitute with proxy material whenever possible (PVC suspension, dimethyl p phthalte, etc). When detail processes are not available (e.g., jute), process energy is included whenever possible to account for the "fossil fuel depletion", "climate changes" and energy-related emissions.

Vinyl sheet alt 1

- Vinyl sheet not included in the BEES database -- calendered PVC sheet (European data) and the same amount of SB rubber (substitute for SB latex adhesive) as VCT are assumed. The description for the calendered PVC sheet says "Producion of 300 mm calendered rigid PVC sheet, including production of the PVC resin, and all operations trough mixing, preplastifying, calendering, pulling, cooling rolls, wind-up and packaging. All internal factory transport is also included. In practice the polymer contains 2.5% to 5% of additives (stabilisers, polymeric modifiers, slip agents and pigments). However, in the calculations these have all been treated as if they were PVC homopolymer. Data from 4 plants in Germany (1995), production amount unknown."

In the list below, LCI specifications are on the left side, and the corresponding modeling assumptions are described on the right.

Vinyl sheet alt 2 **(Based on AIA, 1996)**

- Based on the composition of sheet vinyl described in AIA's Environmental Resource Guide, the corresponding proxy materials were chosen in Simapro:
- 47% vinyl resin – 47% PVC suspension (APME, 1998)
- 33% fillers and pigments -- limestone (ETH, 1996) 29%, 4% TiO2 (PRé Consultants, 2001a) (as for BEES linoleum)
- 16% plasticizers – 16% dimethyl phthalate (ETH, 1996)
- 4% expoxidized oils and stabilizers – 4% linseed oil (selected "linseed" by IDEMAT, 2001) no info on stabilizers
- process electricity and NG assumed same as VCT -- average elec and NG heat (FAL, 1998)
- The SB latex requirement is assumed to be the same as BEE's VCT (FAL, 1998)

***Vinyl sheet alt 3*(Based on AIA, 1996)**

- the process electricity and NG in *alt2* are replaced with "calendering PVC foil process (Kemna, 1981)"

Vinyl sheet alt 4(Based on Potting and Blok, 1995)

- Based on the composition of sheet vinyl described in Potting, the proxy materials were chosen in Simapro:
- 50% PVC – PVC suspension (APME, 1998)
- 30% plasticizer – dimethyl p-phthalate (ETH, 1996)
- 15% limestone – limestone (ETH, 1996)
- 3% stabilizers + some other additives – 4% linseed (IDEMAT, 2001)
- 0.3% pigments – 3% TiO2 (PRé Consultants, 2001a)

***VCT alt 1* (Based on BEES)**

- 10% vinyl acetate and 90% VC for the vinyl component (12% by mass)—suspension PVC is assumed (APME, 1998)
- 4% DEHP -- 4% dimethyl p-phthalate (ETH, 1996)
- NG (0.85MJ/kg) of process energy -- Heat from nat. gas (FAL, 1998)
- SB adhesive -- SB rubber (FAL, 1998)

***VCT alt2* (Based on BEES)**

- NG and electricity for process energy as in *alt1*are replaced with "calendering PVC foil process (Kemna, 1981)"

***VCT alt 3*(Based on AIA, 1996)**

- Based on AIA (1996), the following compositions were chosen in Simapro:
- 80% fillers and pigments -- 4% TiO2 (PRé Consultants, 2001a), 76% limestone (ETH, 1996)
- 13% vinyl resins -- PVC suspension (APME, 1998)
- 5% plasticizers -- dimethyl phthalate (ETH, 1996)
- 2% processing aids and stabilizers -- 2% linseed (for linseed oil) assumed (IDEMAT, 2001)

***Linoleum alt1*(Based on BEES)**

- Linseed oil production (23.3%) -- Linseed production (IDEMAT, 2001)
- Pine rosin (7.8%) – no data available. The description for Forbo linoleum in BEES assumes there is no environmental impacts from pine rosin because "Pine rosin it's manufactured manually".
- Jute backing (10.9%) -- 18.6 MJ/kg is used (data from AIA, 1996)
- Pigment (4.4%) -- TiO2 assumed (PRé Consultants, 2001a)
- Acrylic lacquer (0.35%) – assume 50% methyl methacrylate and 50% acetone (APME, 1999)
- cork flour (5%) -- 1.68 MJ/kg assumed for barking and grounding (BEES) (FAL, 1998)
- wood flour (30.5%) -- 1.68 MJ/kg for barking and grounding, process energy assumed same as cork flour (FAL, 1998)
- diesel truck for pigment and acrylic lacquer -- Truck (single) diesel FAL
- SB latex adhesive -- SB rubber(FAL, 1998)

Linoleum alt2

- The "linoleum" contents in Alt1 were replaced by "linoleum" in SimaPro. The data source for the SP linoleum is RIVM, 1993. They assumed that linoleum is made from linseed oil, natural resin and Tall oil, a waste product of the paper industry and that 0.36kg oil is produced from 1kg seed.

Cork alt 1(Based on BEES)

- Since the cork constituent is a waste product, the environmental burdens from virgin production of the cork are not included. The energy used to grind the cork, however, is included as manufacturing energy. Electricity and an on-site boiler are used to blend and cure
- Manufacturing parquet flooring requires about 0.8 MJ of both thermal and electrical energy per unit produced (0.09 m2 or 1 ft2).
- Producing each unit of product generates about 1 kg of waste, 94 % of which is used to produce energy and 3 % of which is recycled. The recycled material is accounted for in the BEES life cycle inventory. However, in Simapro, the production heat from cork waste was ignored (energy content of cork not readily available)
- The finished cork products are shipped first from the manufacturing facility in Portugal to the Natural Cork warehouse in Georgia–a distance of about 6437 km (4000 mi) (FAL, 1998).
- 0.9 MJ/ft2 manufacturing fuel energy and 0.854 MJ/kg primary energy for electricity is assumed (FAL, 1998)
- Phenol-formaldehyde resin is assumed for the binder (IDEMAT, 2001)

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Appendix D: Toxicants Assessed in the Human Health Risk Assessment

The following is a list of the primary toxicants believed to be used in the manufacture or emitted during installation and use of the building materials assessed in this report. As stated in the text of the report, many more compounds are used in the manufacture of these building materials, emitted by the materials after production, and or are produced during combustion or disposal. However, the risk assessment was limited to those compounds believed to be most significant in either exposure or potential for adverse effects; only the manufacturing, construction, and use phases were considered. (End of life was considered in the life cycle assessment.)

The first table lists the compounds for each building material, their Chemical Abstract Service number, and the inhalation toxicity Reference Dose and Cancer Slope Factor. The second table lists the exposure concentrations for the receptors. These exposure concentrations were considered to be representative of low, likely, and high exposures, based on published data, data provided by industry, or using OSHA PELs as an upper exposure limit (see report for additional information on exposure modeling).

Risk estimates were obtained using the equations provided in Section 3 of the report, the exposure concentrations provided in Table B below and the toxicity values provided in Table A below.

Table A. Compounds Associated with Vinyl and Alternative Building Materials

Material	Compound	CAS Number	RfD, inhalation mg/kg-day	CSF, inhalation $(mg/kg\text{-}day)^{-1}$
Vinyl	VCM*	75-01-4	2.86E-02	1.54E-02
	vinyl acetate	108-05-4	5.71E-02	--
	EDC*	107-06-2	1.14E-01	9.1E-02
	DEHP*	117-81-7	2E-02	NC#
	cadmium	7440-43-9	1E-03	6.3
	barium	7440-39-3	7E-02	--
	dialkyl tins	--	NA	--
Vinyl & linoleum	toluene	108-88-3	1.14E-01	--
	benzyl alcohol	100-51-6	3E-01	--
	2-butoxyethanol	111-76-2	3.71	--
	ethylene glycol	107-21-1	2	--
	naphthalene	91-20-3	8.57E-04	--
	propionaldehyde	123-38-6	2.57E-03	--
	trimethylsilanol		NA	NA
	acetaldehyde	75-07-0	2.57E-03	7.7E-03
	acetone	67-64-1	9E-02	--
Cork	phenol	108-95-2	5.71E-02	--
	furfural	98-01-1	3E-03	--
	formaldehyde	50-00-0	2.86E-03	4.55E-02
Cast Iron	metal fumes		NA	NA
	coke oven emissions	8007-45-2	--	2.17
	manganese	7439-96-5	1.42E-05	--
	pyrene	129-00-0	3E-02	--

Material	Compound	CAS Number	RfD, inhalation mg/kg-day	CSF, inhalation $(mg/kg\text{-}day)^{-1}$
	benzo(a)pyrene	50-32-8	--	3.1
	fluoride	7782-41-4	6E-02	--
	limestone	1317-65-3	NA	--
	calcium silicate	1344-95-2	NA	NA
ABS	acrylonitrile	107-13-1	5.71E-04	2.38E-01
	1,3-butadiene	106-99-0	5.71E-04	1.05E-01
	styrene	100-42-5	2.86E-01	--
	ethylbenzene	100-41-4	2.86E-01	--
	isopropylbenzene	98-82-8	NA	--
	n-propylbenzene	103-65-1	NA	--
	methylstyrene (mix)	2501-31-54	1E-02	--
	acetophenone	98-86-2	1E-01	--
Fiber cement	Calcium silicate	1344-95-2	NA	NA
	Crystalline silica	14808-60-7	NA	NA
	Cellulose	9004-34-6	NA	--
Aluminum	Aluminum	7429-90-5	1E-03	--
	PAHs (see cast iron)			
Wood	Wood dust	NA	NA	NA

*VCM, vinyl chloride monomer; EDC, ethylene dichloride; DEHP, di(2-ethylhexyl)phthalate
Data obtained from U.S. EPA's IRIS (Integrated Risk Information System) Database (www.epa.gov/iris); U.S. EPA's Health Effects Assessment Summary Tables, U.S. EPA Region III Risk Based Concentration Table; Cal/EPA Chronic REL Values (www.oehha.ca.gov/risk/ChemicalDB) and ATSDR MRL Tables (www.atsdr.cdc.gov).
DEHP not considered carcinogenic to humans for the purpose of this assessment.
Inhalation RfD values were calculated from Reference Concentrations (from IRIS) by multiplying by 20 m3/day inhalation rate and dividing by a 70 kg weight (RfC mg/m3 x 20 m3/day/70 kg = mg/kg-day)
Inhalation CSF values were calculated from Inhalation Unit Risk values (from IRIS) by multiplying by the body weight (70 kg) and dividing by a 20 m3/day inhalation rate (UR mg/m3 x 70 kg/20 m3/day = $mg/kg\text{-}day^{-1}$)
NA= not available

Table B. Exposure Concentrations for Manufacturing, Construction and Use Phases—Building Materials

Material	Compound	Low (mg/m3)	Likely (mg/m3)	High (mg/m3)
PVC Pipe and Vinyl Siding				
Vinyl Resin	VCM	0.13	--	2.6
	EDC	40!	--	200
PVC Pipe and Siding	VCM	0&	0.13	2.6
	Barium	0.25	0.25	0.5
	Cadmium	0#	0.0025	0.005
	Dialkyl tins	0.0097*	0.0097	0.0097
ABS Pipe				
ABS Resin	Acrylonitrile	0.078+	--	4.34
	1,3-butadiene	0.0015+	--	2.21
	Styrene	230	--	230
ABS Pipe	acrylonitrile	**		4.74
	1,3-butadiene	**	--	0.97
	styrene	**	--	196
	ethylbenzene	**	--	33.7

	isopropylbenzene	**	--	10.8
	n-propylbenzene	**	--	5.15
	methylstyrene	**	--	30.4
	acetophenone	**	--	35.1
Cast Iron Pipe				
Cast Iron	iron oxide fume	10	--	10
	coke oven emissions	0#	--	0.15
	limestone	5	--	5
	silica	0.05	--	0.05
	metal alloys	NA	--	NA
	cadmium	0.005	--	0.005
	manganese	0.033	--	0.033
	pyrene	0#	--	0.0037
	benzo(a)pyrene	0#	--	4E-04
Windows				
Vinyl	VCM	O&	0.13	2.6
	cadmium	0&	0.005	0.005
	barium	0.25	0.5	0.5
	dialkyl tin	0.0097*	0.0097	0.0097
	metal alloys	NA	NA	NA
	coke oven emissions	0#	0.15	0.15
	silica	0.05	0.05	0.05
	pyrene	0.0037	0.0037	0.0037
	benzo(a)pyrene	4E-04	4E-04	4E-04
Aluminum	aluminum	0.029	--	0.9
	benzo(a)pyrene	0#	--	8.4E-04
	pyrene	0#	--	3.7E-04
	fluoride	0.0022	--	0.69
	coke oven emissions	0#	--	0.15
Flooring				
Vinyl sheet manufacture	VCM	0&	0.13	2.6
	DEHP@	--	--	5
Vinyl sheet installation/use$	toluene	--	1	--
	phenol	--	0.0048	--
Vinyl composition resin manufacture	VCM	0.13	--	2.6
	vinyl acetate##	30	--	30
	EDC	40	--	200
Vinyl composition tile installation/use$	acetaldehyde	--	0.0073	--
	benzyl alcohol	--	0.0061	--
	toluene	--	0.0502	--
	trimethylsilanol	--	0.0029	--
Cork manufacture$$	formaldehyde	6.08E-07	6.68E-06	1.28E-05
	furfural	9.11E-06	1.11E-04	2.13E-04
	phenol	8.51E-05	2.40E-04	3.95E-04
Cork installation/use$$	formaldehyde	4.75E-04	5.23E-03	9.98E-03
	furfural	7.13E-03	8.70E-02	1.66E-01

	phenol	6.66E-02	1.88E-01	3.09E-01
Linoleum installation/use$	acetaldehyde^		0.0045	0.023
	acetone	--	2.2E-02	2.2E-02
	2-butoxyethanol^	--	0.005	0.012
	ethylene glycol	--	0.1	0.1
	naphthalene	--	0.0032	0.0032
	propionaldehyde	--	0.043	0.043
	toluene	--	0.4027	0.4027

High concentrations for most compounds represent OSHA PELs.

Values for VCM, 0.13 mg/m3 based on exposure data provided by Vinyl Institute (Vinyl Institute USGBC PVC TG Data Submission, 2004)

!Low value for EDC represents the ACGIH TLV based on the achievability of this exposure limit according to OSHA.

+Low levels of acrylonitrile and 1,3-butadiene taken from Perbellini et al. (1998) and Fustinoni et al. (2004), respectively.

*Data from Boraiko et al. "Evaluation of Employee Exposure to Butyltin Compounds used as Tin Stabilizers at Polyvinyl Chloride Processing Facilities", downloaded from http://www.cpia.ca/files/files/files_VCC_Study_on_Emp._Exposure_to_Butyltin_Compounds.pdf

**Exposure to low concentrations of ABS emitted compounds not modeled based on data from Forrest et al. (1995) indicating concentrations of emissions below detection limits. Data for high levels of emitted compounds are from Contos et al. (1995).

#Exposure to these compounds assumed below detection limits based on controls in-house.

@ Exposure was assumed to be 0.286 mg/kg-day based on data in NTP-CERHR Expert Panel Report on DEHP.

Exposure point concentration for vinyl acetate from ATSDR Toxicity Profile for Vinyl Acetate (1992).

Data on steel component of vinyl windows are from Omland et al. 1994—total PAH of 10 µg /m3, with 4.4% comprised of benzo(a)pyrene and 36.9% comprised of pyrene.

Data on cast iron and aluminum from Healy et al. (2001) with a total PAH of 19 µg /m3, with PAH and B(a)P composition as described above.

$Data for vinyl flooring, vinyl composition tile, and linoleum installation/use are from the Building Material Emissions Study, Nov. 2003; California Integrated Waste Management Board. Toluene concentrations represent measured toluene as well as total VOCs that were not individually speciated. Concentrations for toluene were estimated using the equations provided in the emissions study and the following emissivity factors: VCT, 100 µg/m2-hr; vinyl, 2100 µg /m2-hr; linoleum, 840 µg /m2-hr proved in the study.

$$Data for cork manufacture, installation and use estimated using compounds and emission factors reported in Horn et al. (1998) and using exposure concentrations provided in the emissions study cited above.

^Differences in exposure concentrations reflect data from a letter from Berkeley Analytical Associates to Mr. Tim Cole, Forbo Linoleum stating that the flooring evaluated met Section 01350 Material Specifications as stated in the Collaborative for High Performance Schools (www.chps.net); therefore, it was assumed the concentrations of these compounds were half their CA chronic RELs (www.oehha.ca.gov).

Appendix E: Task Group Biographies

Kara Altshuler

Dr. Altshuler has ten 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 recently managed a 5-year contract to provide human health and ecological risk assessment support to EPA's Office of Pesticide Programs. In this role, Dr. Altshuler supervised a staff of over 20 scientists and support personnel in the development of toxicological analyses and health effects documents regarding human exposure to antimicrobial pesticides.

In continuing support to the Significant New Alternatives Policy Program (SNAP), Dr. Altshuler helps develop risk-based exposure levels for occupational workers, end-use consumers, and the general public for compounds developed as alternatives to Halon 1301 and other ODCs. She provides EPA with technical guidance and expertise regarding exposure and risk assessment issues. She identifies data needs with regards to toxicological assays necessary for determining potential risks from human exposure to ODC-replacements in a variety of end uses: solvents, fire suppressants, sterilants, foam-blowing applications, and refrigerants. Dr. Altshuler was instrumental in the development of the Acceptable Exposure Limit (AEL) for n-propylbromide (nPB; a solvent used in adhesive applications) using benchmark dose analysis and state of the art risk assessment techniques. She was also a key member of the team responding to peer review and public comments regarding the AEL and helped prepare and deliver a briefing before US EPA Assistant Administrator Jeffrey Holmstead regarding the derivation of the AEL.

Scot Horst

Scot Horst began Horst, Inc., a sustainable materials consulting firm, in 1994. Horst 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 US, 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 US 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.