ABSTRACT

ARNOLD, ALEXIS BROOKE. Disparities in Burden from Chlorinated Air Pollutants in the United States: A Temporal Analysis Using National Emissions Inventory Data. (Under the direction of Drs. Jennifer Richmond-Bryant, Madhusudan Katti, and Stacy Nelson)

This study investigates temporal trends in disparities in emissions burden from chlorinated air pollutants across both racial/ethnic and poverty subgroups in the United States from 2008 to 2020. Using data from the U.S. Environmental Protection Agency's National Emissions Inventory (NEI), we calculate both absolute and proportional burdens of 50 chlorinated compounds at the Census block group level. A distance-based centroid-containment method was applied to spatially assign emissions to nearby block groups. Findings reveal that while national emissions of many chlorinated compounds have declined over time, the distribution of these reductions has not been equitable. The non-Hispanic Black, Hispanic, and below-poverty subgroups consistently experienced the highest absolute and proportional burdens across multiple compounds and years. Notably, some compounds demonstrated increasing burdens or persistent disparities despite overall emission reductions, suggesting that structural inequities in facility siting and environmental governance continue to drive unequal exposure. These results underscore health disparities and the need to address not only emissions but also the demographic distribution of chlorinated compound burdens over time.

Disparities in Burden from Chlorinated Air Pollutants in the United States: A Temporal Analysis Using National Emissions Inventory Data

by
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TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

CHAPTER 1. INTRODUCTION

The widespread use of chlorinated compounds has been increasingly challenged and regulated due to the well-documented environmental and health effects of these pollutants. Consequently, many individual chlorinated compounds have been banned. In April 2024, the Environmental Protection Agency (EPA) finalized a rule aimed at reducing toxic air pollution from the Synthetic Organic Chemical Manufacturing Industry and the Polymers and Resins Industries, specifically targeting emissions of chlorinated compounds such as chlorine, chloroprene, ethylene dichloride (1,2-dichloroethane), and vinyl chloride (U.S. Environmental Protection Agency, 2024b). The primary objective of this regulation is to mitigate pollution exposure for communities residing near these industrial facilities, where elevated cancer risks have been observed (Arias-Ortiz et al., 2018; Domingo et al., 2020). As part of the rulemaking process, the EPA conducted both a risk-based demographic assessment and a community risk assessment, concluding that the policy is likely to reduce the number of individuals disproportionately affected by emissions of the listed chemicals (U.S. Environmental Protection Agency, 2024b). These assessments further revealed that among the 9.3 million people living within six miles of the facilities covered by the regulation, 95,000 face heightened cancer risks due to prolonged exposure to airborne toxic pollutants. However, even with the implementation of this rule, elevated risks persist, disproportionately affecting Black Americans and individuals living in poverty (Madrigal et al., 2024). While these findings specifically address air toxics exposure in general, it is hypothesized that similar disparities extend to chlorinated compounds. Here, we present estimated burden for many chlorinated compounds on a national scale in an effort to document temporal changes in burdens from proximity to emissions of these compounds.

1.1 Health Effects of Chlorinated Compounds

Chlorinated compounds are associated with a wide range of acute and chronic health effects, many of which pose serious risks to human health. Acute exposure may result in irritation of the eyes, skin, mucous membranes, and respiratory system, which documented cases linked to allyl chloride (U.S. Environmental Protection Agency, 1990b), benzyl chloride (U.S. Environmental Protection Agency, 1989a), bis(chloromethyl) ether (U.S. Environmental Protection Agency, 1988a), chlorine (U.S. Environmental Protection Agency, 1994), chloromethyl methyl ether (U.S. Environmental Protection Agency, 1987b), hydrochloric acid (U.S. Environmental Protection Agency, 1995), and pentachlorophenol (U.S. Environmental Protection Agency, 2010d). Additionally, many chlorinated compounds exhibit neurotoxic effects, which can cause headaches, dizziness, lethargy, memory loss, and confusion. Prolonged or repeated exposure may further impair cognitive function and motor coordination. Compounds associated with temporary central nervous system depression include allyl chloride (U.S. Environmental Protection Agency, 1990b), benzyl chloride (U.S. Environmental Protection Agency, 1989a), chlordane (U.S. Environmental Protection Agency, 1998a), chloroacetic acid (U.S. Environmental Protection Agency, 2004b), chlorobenzene (U.S. Environmental Protection Agency, 1989b), chloroform (U.S. Environmental Protection Agency, 2001), ethylidene dichloride (U.S. Environmental Protection Agency, 1988b), hexachloroethane (U.S. Environmental Protection Agency, 2011a), propylene dichloride (U.S. Environmental Protection Agency, 1991), toxaphene (U.S. Environmental Protection Agency, 1998b), trichloroethylene (U.S. Environmental Protection Agency, 2011b), vinyl chloride (U.S. Environmental Protection Agency, 2000), and vinylidene chloride (U.S. Environmental Protection Agency, 2002).

In addition to these acute effects, long-term exposure to chlorinated compounds has been widely associated with severe chronic health conditions, including cardiovascular, hepatic, and pulmonary risks. Disruption of the cardiovascular system can lead to arrhythmia, low blood pressure, and tachycardia, particularly from chloroprene (U.S. Environmental Protection Agency, 2010c) and ethylidene dichloride (U.S. Environmental Protection Agency, 1988b). Several chlorinated compounds are also hepatotoxic, impairing liver function and metabolic processes. Documented cases of liver toxicity have been linked to 1,1,2,2-tetrachloroethane (U.S. Environmental Protection Agency, 2010a), carbon tetrachloride (U.S. Environmental Protection Agency, 2010b), chloroform (U.S. Environmental Protection Agency, 2001), ethylene dichloride (U.S. Environmental Protection Agency, 1987a), hexachlorobenzene (U.S. Environmental Protection Agency, 1988c), polychlorinated biphenyls (PCBs) (U.S. Environmental Protection Agency, 1996), vinyl chloride (U.S. Environmental Protection Agency, 2000), and vinylidene chloride (U.S. Environmental Protection Agency, 2002). Pulmonary complications are another notable concern, with lung inflammation and chronic bronchitis associated with 2,4,6-trichlorophenol (U.S. Environmental Protection Agency, 1990a), bis(chloromethyl) ether (U.S. Environmental Protection Agency, 1988a), hydrochloric acid (U.S. Environmental Protection Agency, 1995), and titanium tetrachloride (Agency for Toxic Substances and Disease Registry (ATSDR), 1997). Additionally, pulmonary edema has been linked to 2,4,5-trichlorophenol (U.S. Environmental Protection Agency, 1987c), allyl chloride (U.S. Environmental Protection Agency, 1990b), benzyl chloride (U.S. Environmental Protection Agency, 1989a), chloromethyl methyl ether (U.S. Environmental Protection Agency, 1987), hydrochloric acid (U.S. Environmental Protection Agency, 1995), and titanium tetrachloride (Agency for Toxic Substances and Disease Registry (ATSDR), 1997).

Several chlorinated compounds have also been widely classified as carcinogenic by regulatory agencies, with long-term exposure associated with elevated cancer risks. Specific associations include benzotrichloride and respiratory cancers (U.S. Environmental Protection Agency, 1990c), tetrachloroethylene and bladder cancer, non-Hodgkin's lymphoma, and multiple myeloma (U.S. Environmental Protection Agency, 2012), and trichloroethylene and kidney, liver, cervical cancer (Chiu et al., 2013; U.S. Environmental Protection Agency, 2011b), and non-Hodgkin's lymphoma (Chiu et al., 2013; Purdue et al., 2011). Additionally, chloroprene exposure has been linked to increased risk of liver cancer (U.S. Environmental Protection Agency, 2010c), PCBs have been associated with breast cancer (Negri et al., 2003; Rodgers et al., 2018; Woolcott et al., 2001), and vinyl chloride has been correlated with angiosarcoma (U.S. Environmental Protection Agency, 2000).

Beyond carcinogenic effects, chlorinated compounds have been linked to adverse reproductive outcomes, developmental risks, and potential birth defects. Exposure to PCBs (Feeley & Brouwer, 2000; Colborn et al., 1993), trichloroethylene and tetrachloroethylene (Choi et al., 2006; Chiu et al., 2013; Forand et al., 2012) has been associated with hormonal disruption, fetal development risks, and potential birth defects. This compilation underscores the major health concerns associated with chlorinated compounds, highlighting the need for continued monitoring and regulation to mitigate their risks to human health.

1.2 Exposure to Chlorinated Compounds

Chlorinated compounds have historically played a critical role in industrial processes, with applications spanning multiple industrial processes and applications (Huang et al., 2014; Hathaway & U.S. Environmental Protection Agency, 1980; Wolf & Chesnutt, 1987; Doherty, 2000a; Doherty, 2000b; U.S. Environmental Protection Agency, 2004a). Many chlorinated compounds have been used as solvents, intermediates, and components in manufacturing. Their industrial applications include use as metal degreasing and cleaning agents (Huang et al., 2014; Wolf & Chesnutt, 1987; Doherty, 2000a; Doherty, 2000b), dry cleaning solvents (Huang et al., 2014; Wolf & Chesnutt, 1987; Hathaway & U.S. Environmental Protection Agency, 1980; Doherty, 2000a; Doherty, 2000b), pesticides, agricultural chemicals, and grain fumigants (Huang et al., 2014; Hathaway & U.S. Environmental Protection Agency, 1980; Doherty, 2000a; Doherty, 2000b), refrigerants and aerosols, including the now banned CFC-113 (Freon) (Wolf & Chesnutt, 1987; Doherty, 2000a; Doherty, 2000b), intermediates in pharmaceutical production (Huang et al., 2014), adhesives and paint removers (Huang et al., 2014; Wolf & Chesnutt, 1987; Hathaway & U.S. Environmental Protection Agency, 1980; Doherty, 2000a; Doherty, 2000b), and in chlorination processes for water treatment, and chemical manufacturing (Huang et al., 2014; Wolf & Chesnutt, 1987; Hathaway & U.S. Environmental Protection Agency, 1980). The production and use of chlorinated compounds are highly interconnected, with one solvent often serving as a precursor or byproduct of another (Wolf & Chesnutt, 1987; Doherty, 2000a; Doherty, 2000b). As a result, restricting or banning a single compound can reduce the use of related chemicals while simultaneously leading to substitutions with alternative compounds, further complicating exposure pathways (Wolf & Chesnutt, 1987; Doherty, 2000a; Doherty, 2000b).

Beyond industrial applications, chlorinated compounds have been incorporated into a variety of everyday consumer products. Freon, a chlorinated compound, was widely used in air conditioning systems before its ban, due to concerns about ozone layer depletion (Wolf & Chesnutt, 1987). Chlorinated compounds have been detected in personal care products, including methylene chloride, tetrachloroethylene, and trichloroethylene in shoe polish (Hathaway & U.S.

Environmental Protection Agency, 1980; Doherty, 2000a; Doherty, 2000b), 2,4,6-trichlorophenol in acne creams (Hathaway & U.S. Environmental Protection Agency, 1980), chloroform in toothpaste (Huang et al., 2014) and cough medicine (Hathaway & U.S. Environmental Protection Agency, 1980), and tetrachloroethylene and trichloroethylene in typewriter correction fluid (Doherty, 2000a; Doherty, 2000b). Although these are historical examples, the impacts of these pollutants are still felt today. Take methylene chloride for example; classified as a probable human carcinogen by the International Agency for Research on Cancer (IARC), methylene chloride continues to be used as a solvent in decaffeinating coffee (Wolf & Chesnutt, 1987; Doherty, 2000b). Whether in factories or households, the widespread use of chlorinated compounds has contributed to multiple avenues of human exposure.

Widespread use of chlorinated compounds has resulted in environmental accumulation, raising concerns about human exposure and associated health risks (Jayaraj et al., 2016; Huang et al., 2014; Doherty, 2000a). Human exposure to chlorinated compounds occurs through multiple pathways, including some occupational settings, household products, and environmental contamination. Factories and chemical plants are major sources of chlorinated compound emissions, releasing these substances into the air and water (Huang et al., 2014; Wolf & Chesnutt, 1987; Hathaway & U.S. Environmental Protection Agency, 1980; Doherty, 2000a; Doherty, 2000b). Workers in industries such as metal degreasing, dry cleaning, and chemical manufacturing may face heightened exposure risks due to prolonged contact with these substances. In addition to workplace exposure, chlorinated compounds are commonly used in household products, leading to inhalation, skin absorption, or ingestion. Common household sources of chlorinated compound exposure include paint strippers, cleaning products, and disinfectants (Huang et al., 2014; Wolf & Chesnutt, 1987; Doherty, 2000a; Doherty, 2000b).

These diverse uses have led to extensive environmental disruption, increasing the likelihood of human exposure via the persistent nature of these pollutants, and leading to contamination of natural resources.

Environmental contamination of, and from, various chlorinated compounds is downstream of the use of these materials in industrial and household products. Environmental contamination presents a substantial concern, as many chlorinated compounds are highly persistent, particularly PCBs, dioxins, and organochlorine pesticides (Jayaraj et al., 2016). The persistent nature of chlorinated compounds has led to a concern about bioaccumulation in living organisms. Studies have shown that even low-level environmental exposures can result in serious bodily disruptions (Jayaraj et al., 2016; Colborn et al., 1993; Miranda et al., 2008; Norström et al., 2010). Further, chlorinated compounds have been detected in air, water, and soil, highlighting their ubiquitous nature and multimedia environmental distribution (Huang et al., 2014; U.S. Environmental Protection Agency, 2004a; Doherty, 2000a; Doherty, 2000b). Hazardous waste sites and landfills further contribute to exposure risks, with trichloroethylene, tetrachloroethylene, methyl chloroform, vinyl chloride, and methylene chloride among the most frequently detected chlorinated compounds and Superfund sites (Huang et al., 2014; Moran et al., 2006; Doherty, 2000b). Additionally, several chlorinated compounds are listed on both the EPA’s Toxic Pollutant List and the Priority Pollutant List under the Clean Water Act and the Comprehensive Environmental Response, Compensation, and Liability Act (also known as the Superfund Act), underscoring the potential for widespread human exposure beyond airborne emissions (Huang et al., 2014; Doherty, 2000a). Although regulatory agencies, such as the EPA, account for chlorinated compound emissions through tracking systems like the National Emissions Inventory

(NEI), there remains limited data on the extent of exposure across different population subgroups to our knowledge.

1.3 Disparities in Exposure to Air Pollution

While chlorinated compounds are widespread in the environment due to their industrial, commercial, and household uses, airborne exposure to these chemicals is likely not experienced equally across all populations. Disparities in exposure to air pollution refers to the unequal distribution of environmental pollutants across different populations and communities, shaped by longstanding structural and systemic factors, including poverty status, race and ethnicity, geographic location, and sometimes occupation (Geldsetzer et al., 2024; Hajat et al., 2015; Lane et al., 2022; Mohai & Saha, 2006; Jiawen Liu et al., 2021; Morello-Frosch et al., 2002). Populations residing in low-income neighborhoods, communities of color, and industrial zones are disproportionately likely to be situated near major sources of pollution (Mohai et al., 2009; Brender et al., 2011), and further geography is a key determinant of air pollution exposure (Yoo et al., 2023). This geographic proximity results in elevated risks of exposure to hazardous air pollutants, including but not limited to chlorinated compounds.

Poverty is a critical factor in understanding disparities in air pollution exposure. Individuals and families living below the poverty line often reside in neighborhoods characterized by lower property values, limited infrastructure investment, and closer proximity to pollution sources such as industrial facilities, landfills, and highways (Mohai et al., 2009; Brender et al., 2011). These spatial patterns of disadvantage are not random; they reflect long-standing economic and environmental policies that have concentrated environmental burdens in communities with fewer financial and political resources (Lane et al., 2022). Research indicates that poverty is consistently associated with elevated exposure to various pollutants (Clark et al., 2014). Poverty, as typically defined, incorporates household size and composition within the United States (U.S. Census Bureau, 2025b). While not encompassed within the more formal definitions of poverty, factors like housing instability and reduced access to environmental protections can be associated with poverty status and may serve as pathways through which material deprivation contributes to environmental exposure (Anyanwu & Beyer, 2024). These structural conditions make low-income populations more likely to live in environments with cumulative pollution burdens (Morello-Frosch et al., 2002).

In addition to socioeconomic factors, race and ethnicity are integral to understanding patterns of air pollution exposure, with numerous studies documenting that Black and Hispanic communities are disproportionately exposed to higher levels of air pollution (Bullard et al., 2008; Tessum et al., 2021; Mikati et al., 2018; Morello-Frosch et al., 2011; Jiawen Liu et al., 2021). The distribution of specific air pollutants, not solely the locations of emitting facilities, reflects persistent racial disparities in exposure. Research consistently demonstrates that Black and Hispanic residents are exposed to substantially higher levels of air pollution compared to their White counterparts (Woo et al., 2019). A study by Tessum et al. (2019) found that $PM_{2.5}$ emissions are disproportionately produced by non-Hispanic White U.S. residents and disproportionately produced in close proximity to non-Hispanic Black and Hispanic U.S. residents.

A central finding in environmental exposure research is that race may serve as a stronger predictor of pollution burden than poverty or income alone (Ringquist, 2005; Jbaily et al., 2022; Mikati et al., 2018; Woo et al., 2019; Jiawen Liu et al., 2021; Tessum et al., 2021). While poverty status is undeniably important, relying solely on income fails to capture the full scope of disparities in air pollution exposure. Ringquist (2005) emphasizes that although the effect sizes may be modest, race remains a statistically significant determinant of exposure. Tessum et al. (2021) further show that racial disparities persist across all income levels, suggesting that these inequities are not simply proxies for economic disadvantage. Liu et al. (2021) and Mikati et al. (2018) also found that income disparities alone could not account for racial disparities in emissions burdens, reinforcing the need to examine race and ethnicity as independent axes of environmental injustice.

Racial and ethnic disparities in health outcomes linked to air pollution are rooted in broader systemic inequities. Studies consistently show that Black, Asian, and Hispanic populations in the U.S. are exposed to higher levels of $PM_{2.5}$ than White populations (Jbaily et al., 2022). These exposure disparities stem from unequal emissions burdens, with particularly severe consequences for Black Americans, who experience disproportionately high rates of premature mortality attributable to air pollution (Geldsetzer et al., 2024). Additional research has corroborated these findings demonstrating that even at similar levels of exposure, Black individuals often face greater health risks due to the compounding effects (Clark et al., 2014; Mikati et al., 2018). The disproportionate health impacts observed in marginalized communities are frequently tied to a legacy of discriminatory practices, including redlining, exclusionary zoning practices, and the siting of polluting industries in non-White neighborhoods (Lane et al., 2022). Redlined neighborhoods, in particular, remain disproportionately burdened by elevated levels of air pollution, and residents in these areas continue to experience higher rates of respiratory and cardiovascular illness (Lee et al., 2022).

Evidence shows that industrial facilities are more likely to be sited in communities that are disproportionately composed of non-Hispanic Black and Hispanic residents (Pastor et al., 2001; Mohai and Saha, 2007; Mohai and Saha, 2015). This challenges the interpretation that demographic patterns around environmental hazards merely reflect economic circumstances or individual housing preferences. One such interpretation, often referred to as the “minority move-in myth,” posits that hazardous facilities are not intentionally placed in marginalized communities, but rather that groups tend to move into these areas after facilities have been sited there (Pastor et al., 2001). Specifically, the potential decline in property values following facility siting could increase housing affordability, thereby attracting lower-income populations. Been and Gupta (1997), found evidence for this feedback loop, in which disproportionate siting and later demographic shifts occur in response to environmental disamenities. Pastor et al. (2001) noted that land use and housing market responses can reinforce the concentration of marginalized groups near pollution sources over time.

However, multiple studies have provided evidence contradicting the “minority move-in myth.” For example, Mohai and Saha (2015) found that racial and socioeconomic characteristics substantially predict the initial placement of polluting facilities, indicating that these communities were disproportionately targeted from the outset. Their earlier national-level analysis (Mohai and Saha, 2007) similarly demonstrated that race remained an important predictor of facility location, even after accounting for other variables. Complementing these findings, the regional study by Pastor et al. (2001) in Los Angeles refuted the “minority move-in myth” by showing that neighborhoods proposed to host hazardous sites already had high concentrations of low-income, majority Black and Hispanic residents, and had a high proportion of renters. This provides further evidence that siting precedes major demographic change.

Other research has examined how demographic transitions, particularly the emigration of White residents, commonly referred to as “White flight,” may precede facility siting in some cases. According to Mohai and Saha (2015), facilities are sometimes placed in neighborhoods that were already undergoing racial and socioeconomic change prior to siting. This pattern became more prominent after the 1970s, coinciding with the rise of "not in my backyard," (NIMBY) resistance to undesirable land uses in wealthier and predominantly White communities (Saha and Mohai, 2005). These findings suggest that siting decisions may be influenced not only by existing demographic characteristics but also by broader patterns that emerge during periods of demographic transition.

Although urban populations are more likely to experience high concentrations of harmful pollutants due to dense traffic, industrial operations, and infrastructure, rural communities face their own distinct risks, often from agricultural chemicals, waste burning, and fewer pollution controls (American Lung Association, 2001). Additionally, regional disparities in exposure are not evenly distributed nationwide. According to Mohai et al. (2009), exposure burdens are especially pronounced in Midwestern and Westen metropolitan areas and suburban areas of the South, suggesting that geography intersects with race and class in complex and regionally specific ways. Conversely, Mikati et al. (2018), found that differences by race/ethnicity and poverty level diminished for rural areas.

1.4 Disparities in the Health Effects of Air Pollution

Residential proximity to pollution sources has been linked to a range of adverse health outcomes, independent of exposure levels. Numerous studies have documented how living near industrial sites, hazardous waste facilities, and solvent-emitting operations increases the risk of chronic and developmental health conditions. For instance, residing near hazardous waste sites has been associated with an elevated risk of developing diabetes, likely due to long-term exposure to environmental contaminants that disrupt metabolic function (Kouznetsova et al., 2007). Similarly, proximity to sites releasing solvents has been connected with central nervous system malformations during fetal development, highlighting heightened vulnerability during critical stages of development (Marshall et al., 1997). In addition, research shows that mothers living near Toxic Release Inventory (TRI) facilities or plants emitting carcinogenic pollutants face a greater likelihood of bearing children diagnosed with childhood brain cancer (Choi et al., 2006). These findings underscore how geographic location acts as a determinant of health by shaping cumulative environmental burdens that often reflect broader structural patterns in health disparities.

There are well-documented disparities in the health effects of air pollution, which are not evenly distributed across populations. Instead, the burden of pollution-related health risks disproportionately affects certain demographic groups, with disparities often arising based on poverty status, race, and residential proximity to pollution sources (Mikati et al., 2018; Ringquist, 2005; American Lung Association, 2001). Moreover, multiple environmental hazards, not just air pollution, disproportionately impact low-income and minority communities (Morello-Frosch et al., 2012). While race and income are often analyzed independently, research suggests that racial disparities persist even among individuals with comparable income levels (Bell and Ebisu, 2012), reflecting the intersectional and compounded nature of environmental health inequities. These disparities in exposure highlight where and to what extent different groups are subjected to harmful environmental conditions, while disparities in health effects focus on how these exposures manifest into differential health outcomes, depending on a myriad of biological, social, and economic factors.

Disparities in health effects due to air pollution focus on the different health outcomes experienced by individuals who are exposed to similar levels of pollutants. While exposure is an imperative factor, not everyone exposed to the same level of air pollution will experience the same health effects. These differences arise due to a variety of factors, including pre-existing health conditions, genetics, access to healthcare, and other social determinants of health. Race and ethnicity may also influence how individuals physiologically respond to pollutants (Woo et al., 2018), contributing to divergent health outcomes.

Socioeconomic status (SES), encompassing income, poverty level, education, health insurance coverage, and Medicaid enrollment, is a critical indicator for understanding disparities in health outcomes related to air pollution exposure. Hajat et al. (2015, 2021) demonstrate that individuals with lower SES are disproportionately affected by the health effects of air pollutants, including chlorinated compounds, due to both increased exposure and heightened vulnerability. While these SES factors often intersect, each contributes uniquely to health inequities. Lower levels of education and limited access to health insurance or Medicaid coverage can impede awareness of associated health risks and reduce opportunities for preventative healthcare and intervention, thereby compounding adverse health outcomes (Bell & Ebisu, 2012; Morello-Frosch et al., 2011). These interrelated socioeconomic factors help illustrate why health impacts from pollutants like chlorinated compounds are not equally distributed across populations.

1.5 Study Objective

The objective of this research is to quantify and analyze temporal patterns of chlorinated compound emissions burdens across the United States and evaluate whether there were disproportionate emissions burdens among racial, ethnic, and poverty groups in the United States between 2013-2022, and if those burdens were reduced or augmented over time. This work builds upon the methodology used by Mikati et al. (2018), expanding it by calculating temporal changes in chlorinated compound emissions burdens on a national scale. A key contribution of this research is the quantification of both absolute and proportional burdens at multiple time points, enabling the analysis of temporal trends in chlorinated compound burdens. By tracking shifts in burdens across demographic subgroups, this study provides new insights into the evolution of inequities associated with chlorinated compound emissions.

CHAPTER 2. METHODS

2.1 Data Sources

We accessed population data via the United States Census Bureau's American Community Survey (ACS) estimates for the years 2009 to 2013, 2012 to 2016, 2015 to 2019, and 2018 to 2022 to obtain estimates for both racial/ethnic identification and poverty status at the Census block group level for all 50 states and Washington, DC (U.S. Census Bureau, 2025a). The ACS is a nationwide, continuous survey used to collect demographic, economic, social, and housing data (U.S. Census Bureau, 2025a). The ACS takes a representative sample of households across the U.S. and aggregates the responses to produce population estimates that are representative of the broader U.S. population (U.S. Census Bureau, 2025a). Categorical measures of racial/ethnic subgroups identified within the ACS for this project included non-Hispanic White (NHW), non-Hispanic Black (NSB), and Hispanic (Hisp.). Using block-group level data, income subgroups were divided into at or below the poverty line (below poverty) and above the poverty line (above poverty). Poverty status is determined by a set of income thresholds that vary by household size and age distribution (U.S. Census Bureau, 2025b). Rural-urban status for all block groups was noted using determinations from the U.S. Department of Agriculture's Rural-Urban Commuting Area (RUCA) codes of the most recent release in 2010 (U.S. Department of Agriculture, 2025).

Emissions data were accessed via the EPA's National Emissions Inventory (NEI) "Facility-level by Pollutant" data for stationary, anthropogenic point-source data for the years 2008, 2011, 2014, 2017, and 2020 (U.S. Environmental Protection Agency, 2024a). The NEI publishes emissions data on a triennial basis, providing detailed information on facility-level releases across the U.S. Each facility's location is given by latitude and longitude. Stationary sources represented in the NEI span a wide range of industrial and institutional sectors, including, but not limited to, pulp and paper plants, wastewater treatment facilities, petroleum refineries, landfills, compressor stations, petrochemical manufacturing, steel mills, gas processing plants, electric power generation stations, oil and gas manufacturing, steel mills, gas processing plants, electric power generating stations, oil and gas infrastructure, universities, hospitals, and chemical production facilities. For all chlorinated compounds assessed in this research, emissions are reported in units of pounds per year. For a comprehensive list of all compounds studied, please refer to Appendix A.

2.2 Methods for Evaluating Environmental Disparities

Previous literature has explored different methodological approaches for assessing environmental disparities, primarily unit-hazard coincidence and distance-based methods. The unit-hazard coincidence method assumes that all residents living within a defined geographic unit (e.g. Census block group) are equally exposed to the hazard (Mohai and Saha, 2006; Mohai et al., 2009; Bullard et al., 2008). However, this method does not account for the exact location of the hazard or its proximity to neighboring units (Mohai and Saha, 2006; Mohai et al., 2009; Bullard et al., 2008). With these drawbacks, using this method could potentially lead to a misclassification of exposure, particularly in areas with large, irregularly shaped Census block groups. For this research, we utilized the distance-based method used by Mikati et al. (2018), originally from Boyce and Pastor (2013), because the distance-based method overcomes the limitations of the unit-hazard coincidence method by considering spatial proximity to hazards. This method accounts for variability in the size and shape of geographic units. Additionally, this method provides a more precise estimate of exposure by including populations in neighboring units who may be closer to the hazard than some within the host unit (Mohai and Saha, 2006).

A distance-based “centroid-containment” method was used to assign a facility and its emissions to any block group if that facility falls within 2.5 miles of a block group’s centroid. One facility could be assigned to several block groups using this method, which uses distance-based measurements instead of unit-hazard coincidence to produce similar scaled geographic areas to deal with irregularities of boundary changes (Mohai and Saha, 2006; Mohai et al., 2009; Boyce and Pastor, 2013). Additionally, Mikati et al. (2018) conducted sensitivity testing to determine if there were appreciable differences in results for radii of 0.50, 1.0, 2.5, or 5.0 miles, validating the choice of a 2.5-mile radius as previously suggested by Boyce and Pastor (2013). No considerable differences were noted, so facilities and their emissions meeting the 2.5-mile centroid containment were assigned to the population within that block group.

2.3 Calculating Burden as an Indicator of Exposure

Burden calculations quantify the disparity of emissions by demographic subgroups. Absolute burden measures total emissions weighted by population within an “exposure” zone. Once absolute burdens are calculated, the values are normalized by the burden calculated for the reference population to obtain proportional burdens. Absolute burden, as seen in (1), was calculated as the average amount of specified pollutant per pound/year emitted within 2.5 miles from a centroid. It represents population-weighted average emissions per block group.

$$Absolute\ Burden = \frac{\sum(Population_{Block\ Group} \times Emissions_{Block\ Group})}{\sum(Population_{Block\ Group})} \qquad (1)$$

Proportional burden compares subgroup-specific emissions burden to the overall population burden, indicating whether certain groups bear disproportionate pollution exposure risks. Using burdens as an exposure surrogate metric allows for more nuanced comparisons of disparities beyond just proximity measures. Proportional burden, as seen in (2), was calculated as the ratio between a subgroup's absolute burden and the overall population. Concerning proportional burdens, scores above 1.00 indicate that the subgroup had a higher absolute burden than the total population in regard to the specific pollutant being considered.

$$Proportional\ Burden = \frac{Absolute\ Burden_{Subgroup}}{Absolute\ Burden_{Overall}} \qquad (2)$$

Burdens were calculated to measure subgroup differences in proximity to facilities and corresponding facility emissions and then aggregated to a national scale. Both absolute and proportional burdens were calculated for each pollutant considered following the procedure detailed by Mikati et al. (2018).

To assess changes in burden over time, burdens were calculated for each NEI year: 2008, 2011, 2014, 2017, and 2020. Regarding Census boundaries, we used two centroid years, 2013 and 2022, to assess differences in burden estimates over time. The years 2013 and 2022 were chosen because 2013 was the first year block group level data was available via data.census.gov and 2022 was the latest year available at the time these calculations were made.

To check the stability of burden estimated across the two centroid years, we computed differences in absolute and proportional burdens between centroid years for each NEI year. We then assessed the percentage magnitude of the change using (3):

$$Percent\ change\ in\ (absolute\ or\ proportional) burden\ = \frac{Difference\ between\ centroid\ years}{Average\ of\ two\ centroid\ years} \qquad (3)$$

In order to calculate the change in burden over time, we averaged the two centroid year's absolute burdens for each NEI year, subtracted the difference between 2008 and 2020, and then plotted with zero at the vertical axis. Additionally, we computed the average proportional burden for each chlorinated compound across all NEI years using the same process.

Further, to better understand the general temporal trends of compounds, we took the overall mean across each group for each year and fit a linear trendline to the plot. Using this trendline, we categorized compounds by the sign of the slope; further, we defined a threshold of -0.001 to +0.001 and any slope within this range was sorted into the variable, or unstable, category as needed. By doing so, we were able to visualize the temporal trends across these compounds.

CHAPTER 3. RESULTS

We are able to quantify patterns in emissions burdens disparities for each compound by quantifying and analyzing temporal patterns of chlorinated compound emissions burdens across the United States. These temporal patterns helped highlight which chlorinated compounds have produced consistently high burdens, population subgroups that are persistently burdened by these compounds, and changes in emission burdens patterns over time. As anticipated, chlorinated compound burdens affect different population subgroups to different extents, which we have outlined below.

3.1 Absolute Burdens

Absolute burdens of chlorinated compounds showed mixed results across most population subgroups. Of the 50 chlorinated compounds, approximately 56% showed overall decreasing trends, 22% showed overall increasing trends, and 22% showed some type of variability or instability over the time frame. Table 1 shows the list of chlorinated compounds and the change over the time period of the study. These estimates provide an updated assessment of emissions burdens for the selected chlorinated compounds. Despite overall declines. Disparities persist across racial/ethnic and poverty subgroups, with shifts in which groups experience the highest burdens. See Appendix B for a full list of compounds and absolute burdens for each subgroup across the study period.

Table 1. Chlorinated compounds for which emissions burdens were calculated, and change over the time period of the study.

Chlorinated Compound	Direction Over Time
1,1,2-trichloroethane	↑
1,1,2,2-Tetrachloroethane	↑
1,2-dibromo-3-chloropropane	—
1,2,4-Trichlorobenzene	↓
1,3-dichloropropene	↓
1,4-dichlorobenzene	↓
2-chloroacetophenone	↓
2-Chlorobiphenyl (PCB-1)	—
2,4,4-TrichlorobIphenyl (PCB-28)	—
2,4,5-Trichlorophenol	↑
2,4,6-Trichlorophenol	↑
Allyl Chloride	↓
Benzyl Chloride	↓
Bis(Chloromethyl)Ether	—
Carbon Tetrachloride	↑
Chlordane	↓
Chlorine	↓
Chloroacetic Acid	↑
Chlorobenzene	↑
Chloroform	↓
Chloromethyl Methyl Ether	—
Chloroprene	↓
Decachlorobiphenyl (PCB-209)	—
Dibenzofuran	↓
Dichloroethyl Ether	↓
Dichlorvos	↓
Dimethylcarbamoyl Chloride	—
Ethyl Chloride	↓
Ethylene Dichloride	↑
Ethylidene Dichloride	↑
Heptachlor	—
Hexachlorobenzene	↑
Hexachlorobutadiene	—
Hexachlorocyclopentadiene	↓
Hexachloroethane	↓
Hydrochloric Acid	↓

Table 1 (continued).

Methoxychlor	↓
Methyl Chloride	—
Methyl Chloroform	↓
Methylene Chloride	↓
Pentachloronitrobenzene	↓
Pentachlorophenol	↑
Polychlorinated Biphenyls	↓
Propylene Dichloride	↓
Tetrachloroethylene	↓
Titanium Tetrachloride	↓
Toxaphene	—
Trichloroethylene	↓
Vinyl Chloride	↓
Vinylidene Chloride	↓

Across all years analyzed, the NHB subgroup consistently exhibited the highest absolute burden values for each year tested. The NHW and Hisp. groups had roughly comparable numbers of chemicals for which these groups had the highest absolute burdens over the span of the study. A breakdown of the absolute burden rankings can be found in Table 2.

Table 2. Counts of highest absolute burden per compound, by population subgroup and year. A total of 50 compounds were examined.

Year	Hispanic	Non-Hispanic Black	Non-Hispanic White
2008	12	28	8
2011	14	28	8
2014	5	33	8
2017	6	35	9
2020	6	30	14

While the majority of compounds displayed overall decreasing trends for emissions burdens, some compounds showed consistent decreasing trends over the time frame studied. In particular, 2-chloroacetophenone, tetrachloroethylene, trichloroethylene, methyl chloroform, hexachlorocyclopentadiene, and hydrochloric acid exhibited consistent decreasing trends. Hydrochloric acid had both the highest absolute burden values at the beginning of data collection and the largest decrease from 2008 to 2020, with a more than 80% decrease across all population groups, making this compound the largest decrease of all compounds considered here. Other compounds with substantial reductions included tetrachloroethylene, trichloroethylene, and methylene chloride. Heptachlor, dimethylcarbamoyl chloride, 2-chlorobiphenyl (PCB-1), and 2,4,4-trichlorobiphenyl (PCB-28) exhibited the smallest changes across all population groups, with changes approaching zero. While these compounds maintained low variability, they were not classified as completely stable.

Some compounds exhibited increases in emissions burdens over the last three years of the time frame considered. In particular, 2,4,5-trichlorophenol, bis(chloromethyl)ether, and hexachlorobenzene. Hexachlorobenzene began with absolute burden values below zero in all population subgroups and showed variable percent increases, notably over 63% increase for the NHB population subgroup. While these trends illustrate overall emissions changes, they do not capture how different population groups may be affected by these changes.

When breaking down trends within and across population subgroups, these trends varied. NHW populations exhibited decreasing burdens in 35 compounds, NHB populations exhibited decreasing burdens in 31 compounds, and Hispanic populations exhibited decreasing burdens in 33 compounds. Considering the poverty distinction, both below-poverty and above-poverty subgroups exhibited decreasing burdens in 33 total compounds.

Table 3. Percent change in absolute burden from 2008 to 2020. Note. Ethylene dichloride, decachlorobiphenyl (PCB-209), and bis(chloromethyl)ether are not present due to missing 2008 NEI data.

Chlorinated Compound	Hisp.	NHB	NHW	Below Poverty	Above Poverty	Total Pop.
1,1,2-trichloroethane	-0.15	0.018	0.050	-0.11	-0.064	-0.077
1,1,2,2-Tetrachloroethane	1.3	1.6	1.7	1.7	1.5	1.5
1,2-dibromo-3-chloropropane	-0.020	-0.11	0.93	-0.084	0.069	0.031
1,2,4-Trichlorobenzene	-0.66	-0.65	-0.78	-0.61	-0.68	-0.67
1,3-dichloropropene	-0.47	-0.35	-0.45	-0.42	-0.42	-0.42
1,4-dichlorobenzene	-0.023	0.096	-0.642	-0.25	-0.24	-0.24
2-chloroacetophenone	-0.90	-0.91	-0.97	-0.93	-0.91	-0.91
2-Chlorobiphenyl (PCB-1)	-0.29	-0.30	-0.24	-0.32	-0.28	-0.29
2,4,4'-TrichlorobIphenyl (PCB-28)	-0.29	-0.30	-0.24	-0.32	-0.28	-0.29
2,4,5-Trichlorophenol	-0.24	6.4	6.5	2.4	1.9	2.0
2,4,6-Trichlorophenol	2.3	17	57	11	4.5	5.5
Allyl Chloride	-0.63	-0.71	-0.18	-0.41	-0.64	-0.60
Benzyl Chloride	-0.89	-0.94	-0.95	-0.92	-0.91	-0.91
Carbon Tetrachloride	0.54	-0.42	-0.53	-0.14	0.13	-0.0020
Chlordane	-0.97	-1.0	-0.41	-1.0	-0.96	-0.97
Chlorine	-0.43	-0.063	-0.66	-0.41	-0.39	-0.40
Chloroacetic Acid	6.3	1.7	10	8.7	4.5	5.1
Chlorobenzene	0.23	1.2	-0.33	0.39	0.25	0.30
Chloroform	-0.285	-0.408	-0.421	-0.382	-0.367	-0.366
Chloromethyl Methyl Ether	74	0.62	10	21	40	34
Chloroprene	-0.86	-0.88	-0.75	-0.87	-0.87	-0.87
Dibenzofuran	-0.58	-0.58	-0.75	-0.63	-0.67	-0.65
Dichloroethyl Ether	1.5	2.5	1.0	0.54	1.7	1.5
Dichlorvos	-0.94	-0.92	-1.0	-0.99	-0.98	-0.98
Dimethylcarbamoyl Chloride	-0.98	-0.98	-0.95	-0.98	-0.98	-0.98
Ethyl Chloride	-0.82	-0.26	-0.78	-0.69	-0.75	-0.75
Ethylidene Dichloride	1.0	1.1	2.0	1.3	1.1	1.2
Heptachlor	-0.19	7.5	-0.82	2.0	-0.28	-0.068
Hexachlorobenzene	24	63	3.9	29	22	23
Hexachlorobutadiene	1.091	0.151	2.420	0.451	0.872	0.731
Hexachlorocyclopentadiene	-0.74	-0.94	-0.90	-0.91	-0.79	-0.83
Hexachloroethane	-0.55	-0.93	-0.87	-0.83	-0.64	-0.70
Hydrochloric Acid	-0.88	-0.89	-0.84	-0.89	-0.88	-0.88

Table 3 (continued).

Methoxychlor	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0
Methyl Chloride	0.059	0.24	-0.35	0.032	-0.009	-0.55
Methyl Chloroform	-0.79	-0.73	-0.92	-0.90	-0.84	-0.85
Methylene Chloride	-0.40	-0.34	-0.53	-0.37	-0.44	-0.43
Pentachloronitrobenzene	-1.0	-1.0	-1.0	-1.0	-1.0	-1.0
Pentachlorophenol	-0.41	0.52	0.25	-0.37	-0.14	-0.19
Polychlorinated Biphenyls	-0.85	-0.65	-0.79	-0.60	-0.77	-0.74
Propylene Dichloride	-0.062	-0.26	-0.11	-0.013	-0.11	-0.085
Tetrachloroethylene	-0.82	-0.75	-0.82	-0.79	-0.83	-0.82
Titanium Tetrachloride	-0.50	-0.80	-0.50	-0.68	-0.49	-0.52
Toxaphene	16	4.2	34	1.2	7.7	9.6
Trichloroethylene	-0.71	-0.69	-0.74	-0.71	-0.73	-0.72
Vinyl Chloride	-0.101	-0.22	0.45	0.12	-0.065	-0.034
Vinylidene Chloride	-0.51	-0.35	-0.63	-0.50	-0.5	-0.51

Some chlorinated compounds exhibited particularly wide burden gaps between population groups. The NHW subgroup has substantially higher burdens than both the NHB and Hispanic groups for both decachlorobiphenyl (PCB-209) and chloromethyl methyl ether, although the difference was smaller for chloromethyl methyl ether. Both the NHB and NHW subgroups showed higher burdens than the Hispanic group for 2,4,4-trichlorobiphenyl (PCB-28) and 2-chlorobiphenyl (PCB-1). In contrast, the Hispanic and NHB subgroups showed higher burdens than the NHW subgroup for dibenzofuran. Similar trends were observed for 1,4-dichlorobenzene, with the NHB group having higher absolute burdens than the NHW subgroup. Conversely, the Hispanic subgroup showed higher burdens than the NHB and NHW subgroups for chloroform. For the following compounds, the NHB subgroup showed a higher absolute burden than both Hispanic and NHW populations: ethylene dichloride, ethylidene dichloride, chloroprene, vinyl chloride, 1,2-dibromo-3-chloropropane, 1,1,2-trichloroethane, chlorobenzene, hexachlorobutadiene, chlorine, 2,4,6-trichlorophenol, propylene dichloride, and methyl chloride.

The Hispanic subgroup also had higher absolute burdens than the NHW subgroups for methyl chloride.

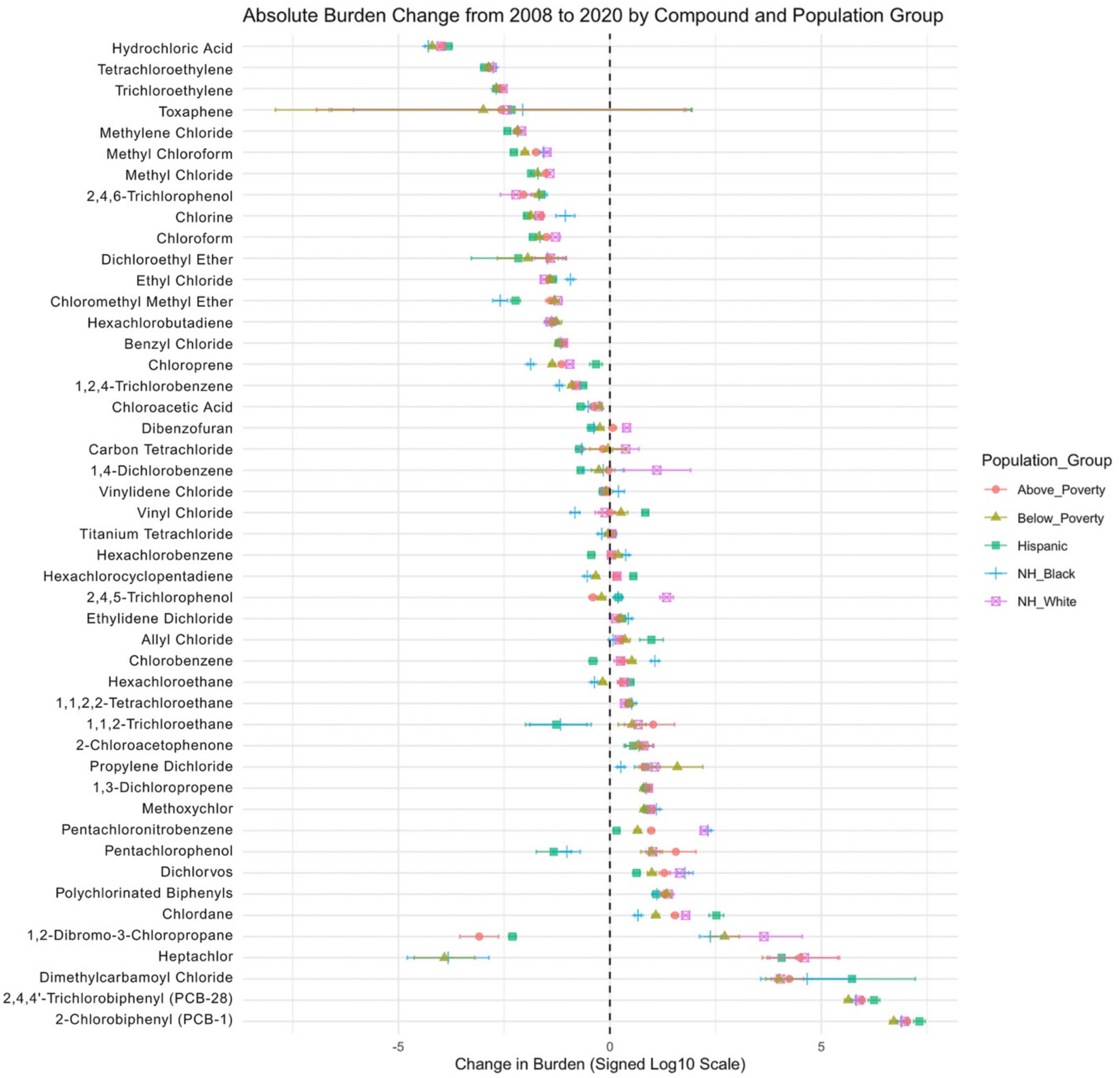

Figure 1. Absolute burden change from 2008 to 2020 by compound and population group. Note. Ethylene dichloride, decachlorobiphenyl (PCB-209), and bis(chloromethyl)ether are not present due to missing 2008 NEI data.

Some chlorinated compounds exhibited distinct subgroup disparities. Chlorobenzene, for example, increased in absolute burden for all population groups except the Hispanic group, which experienced a decrease. Among groups with increasing burdens, NHB individuals experience an increase approximately 6.5 times greater than the NHW group. Additionally, vinyl chloride showed variability across racial/ethnic subgroups, with absolute burden decreasing by 6.68 times in the NHB subgroup, while increasing 6.85 times in this Hispanic subgroup. Carbon tetrachloride decreased for both NHB and Hispanic individuals but increased for NHW individuals. Meanwhile, 1,2,4-trichlorophenol decreased across all population subgroups, though NHB individuals experienced a decline 2.5 times greater than the next highest population subgroup.

The NHB subgroup exhibited the highest absolute burden for 33 of the 50 compounds (66%), averaged across all NEI years. For chlorobenzene, for instance, the NHB subgroup consistently exhibited burdens twice as high as the NHW subgroup and three times as high as the Hispanic group (from 2011 to 2020). Although chloroprene absolute burdens have been declining, NHB populations continue to experience higher burdens as compared to the other racial/ethnic groups. Additionally, the Hispanic subgroup has overtaken the NHB subgroup in absolute burden for several compounds, including 2,4,6-trichlorophenol, 2,4,5-trichlorophenol, 1,1,2,2-tetrachloroethane, vinyl chloride, and titanium tetrachloride.

The below-poverty subgroup had the highest absolute burdens in 45 of the 50 compounds compared to the above-poverty subgroup. Of these 45 compounds, 1,1,2-trichloroethane, 2,4,4-trichlorobiphenyl (PCB-28), decachlorobiphenyl (PCB-209), 1,4-dichlorobenzene, 2-chlorobiphenyl (PCB-1), allyl chloride, propylene dichloride, ethylene dichloride, 1,2-dibromo-3-chloropropane, methyl chloride, chloroacetic acid, and hexachlorobutadiene had some of the highest average absolute values for the below-poverty subgroup. In particular, 2,4,4-trichlorobiphenyl (PCB-28), decachlorobiphenyl (PCB-209), and 2-chlorobiphenyl (PCB-1) had the highest average absolute burdens of all population groups for all years considered here. In contrast, the above-poverty group exhibited higher burdens for only five compounds: 2,4,6-trichlorophenol, bis(chloromethyl)ether, dichloroethyl ether, toxaphene, and titanium tetrachloride. Although absolute burdens provide insight into emissions levels, they do not account for distribution among the population. To further understand inequities, proportional burdens offer a population-normalized perspective.

3.2 Proportional Burdens

Among the 50 chlorinated compounds analyzed, proportional burden trends have shown both increases and decreases over time. Several compounds have exhibited extremely high proportional burdens; here, we define high proportional burden as any value exceeding 1.50, indicating that the group received at least 50% more emissions relative to their population share. Table 4 presents 23 compounds that exceeded the 1.50 proportional burden threshold, ranked from highest to lowest number of average burdens across the study period. Figure 2, Figure 3, Figure 4, and Figure 5 present compounds with relatively high proportional burdens over the time frame. See Appendix C for a full list of compounds and proportional burdens for each subgroup across the study period.

Table 4. Chlorinated compounds with high proportional burdens (≥1.50) ranked from highest to lowest number of average burdens. Note. Population groups marked with * indicate that the group referenced had the highest proportional burden across the time frame, not that the group had the highest proportional burden for all years.

Rank	Chlorinated Compound	Population Group with Highest Burden
1	Dibenzofuran	NHB
2	Methyl Chloride	NHB
3	1,1,2-Trichloroethane	NHB
4	Hexachlorobutadiene	NHB
5	Chloroprene	NHB
6	Hydrochloric Acid	HNB
7	Polychlorinated Biphenyls (PCBs)	NHB *
8	Ethylene Dichloride	NHB
9	1,2-Dibromo-3-Chloropropane	NHB
10	1,2,4-Trichlorobenzene	NHB
11	Pentachloronitrobenzene	Hispanic *
12	Methoxychlor	NHB *
13	Hexachlorocyclopentadiene	NHB
14	Dimethylcarbamoyl Chloride	NHB *
15	Toxaphene	NHB
16	Methyl Chloroform	Hispanic
17	Dichlorvos	Hispanic *
18	Chloroform	Hispanic
19	Chlordane	NHB*
20	Ethylidene Dichloride	NHB
21	Vinyl Chloride	NHB
22	1,3-Dichloropropene	NHB
23	1,4-Chlorobenzene	NHB *

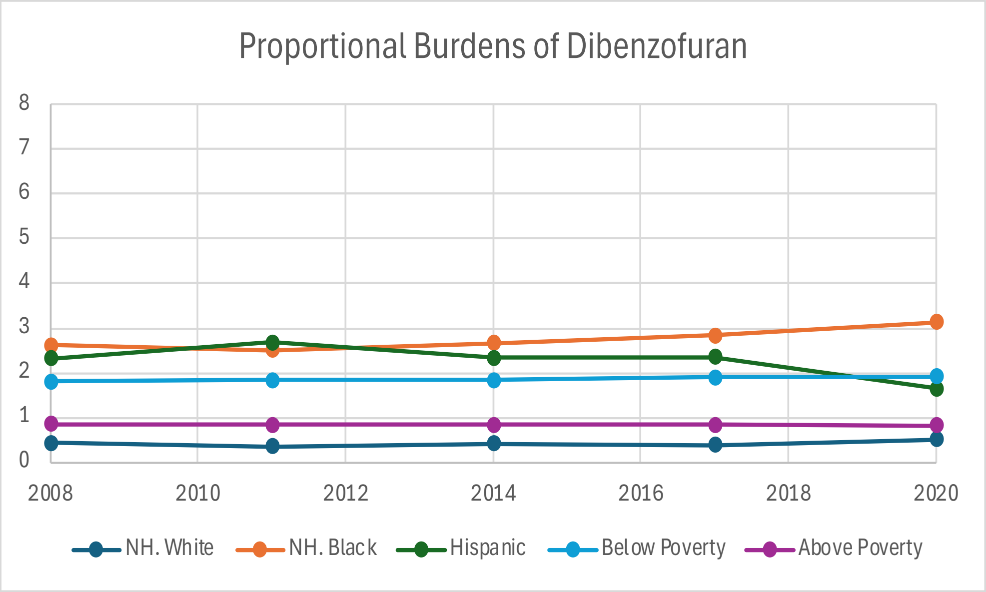

Figure 2. Proportional burdens of dibenzofuran from 2008 to 2020 by population subgroup.

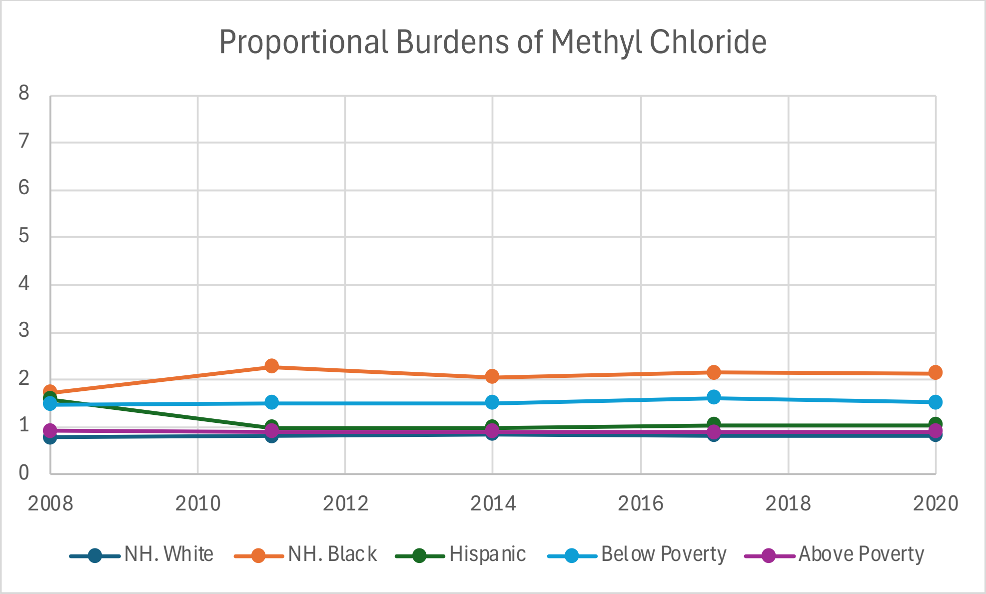

Figure 3. Proportional burdens of methyl chloride from 2008 to 2020 by population subgroup.

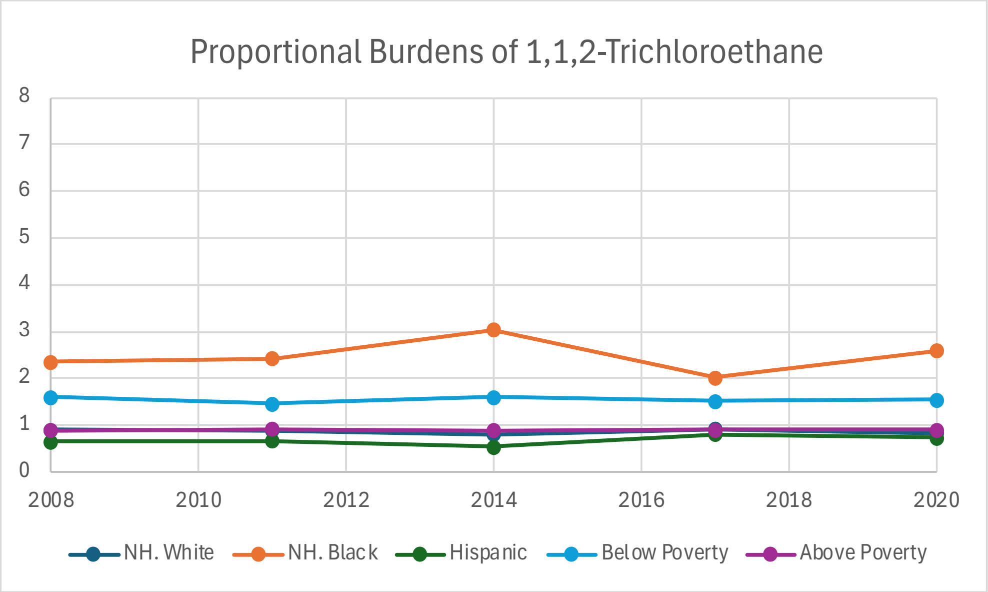

Figure 4. Proportional burdens of 1,1,2-trichloroethane from 2008 to 2020 by population subgroup.

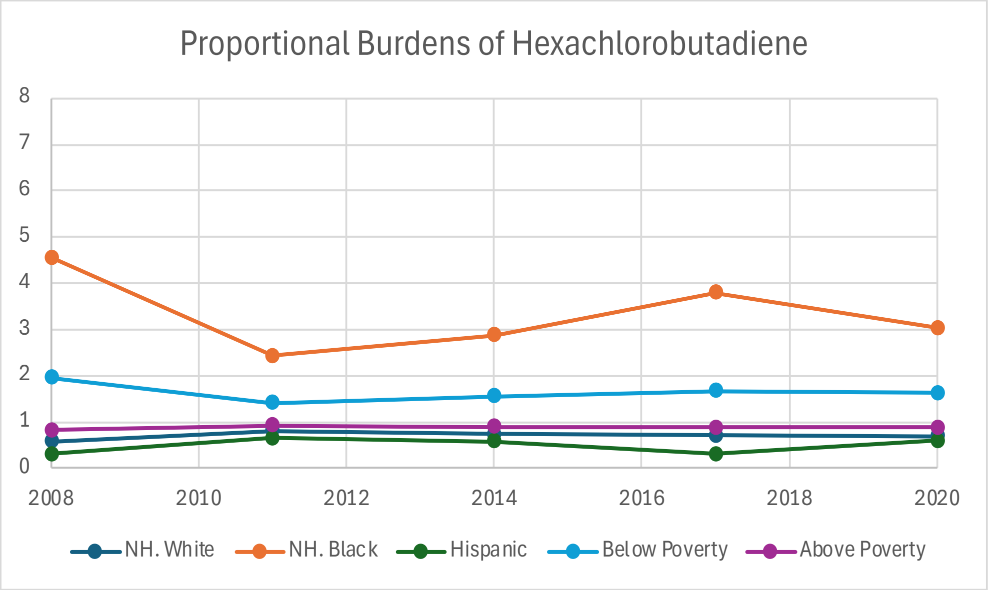

Figure 5. Proportional burdens of hexachlorobutadiene from 2008 to 2020 by population subgroup.

Although 92.4% of all data show a proportional burden below 2.00, disparities remain evident. A particularly striking finding is that 77.9% of all proportional burdens greater than or equal to 2.00, but below 3.00, occur in the NHB subgroup. Additionally, the NHB subgroup had the highest proportional burden for 33 of the 50 compounds analyzed, with the highest burden recorded at 7.15 for dimethylcarbamoyl chloride in the year 2014.

Further, some compounds exhibited a "switching places" phenomenon, in which one racial/ethnic subgroup overtook another as the group with the highest proportional burden. Table 5 presents the compounds in which racial/ethnic subgroups switched places as the group with the highest ranking observed over the time frame.

Table 5. Chlorinated compounds exhibiting crossover in proportional burden rankings by subgroups. Note. A crossover, or switching places, is defined as a change in the subgroup with the highest proportional burden over the study period.

Chlorinated Compound	Subgroups Involved	Crossover Direction
Benzyl Chloride	NHW & Hispanic	Hispanic overtook NHW
2,4,6-Trichlorophenol	NHW & Hispanic	Hispanic overtook NHW
Allyl Chloride	NHB & Hispanic	Hispanic overtook NHB
Methoxychlor	NHB & Hispanic	NHB overtook Hispanic
Pentachloronitrobenzene	NHB & Hispanic	NHB overtook Hispanic
Pentachlorophenol	NHB & NHW	NHB overtook Hispanic

In some cases, proportional burdens were higher in the NHW subgroup. In particular, we found the NHW group to have the highest proportional burden for all years for the compound decachlorobiphenyl. This was the only compound where we observed this finding for all years available for the compound. However, the NHW group held the highest proportional burden for three or more years for the compounds benzyl chloride, chloromethyl methyl ether, hexachloroethane, and 2-chloroacetopehnone. In addition, the above-poverty group was observed as having the highest proportional burden for three or more years (as compared to the below-poverty subgroup) for 2,4,6-trichlorophenol, dichloroethyl ether, heptachlor, toxaphene, and titanium tetrachloride. No compounds were found to have the above-poverty group with the highest ranking for all years considered.

For some compounds, such as benzyl chloride, carbon tetrachloride, chloroform, hydrochloric acid, trichloroethylene, tetrachloroethylene, and chlorine, high absolute burdens were observed alongside proportional burdens close to 1.00. This pattern suggests that no specific subgroup bears a disproportionate share of the emissions burden. These patterns do not necessarily reflect exposure inequities between subgroups.

CHAPTER 4. DISCUSSION

4.1 Main Findings

A key aspect of this analysis is the spatial assignment of emissions burdens to communities. In this study, a distance-based centroid-containment method was used to assign emissions from facilities to surrounding Census block groups. While unit-hazard coincidence remains widely accepted in environmental exposure research for its simplicity and compatibility with geographic unit-based datasets, it assumes uniform exposure within units, an assumption that can result in misclassification, particularly where block groups vary substantially in size and shape (Mohai and Saha, 2006; Mohai et al., 2009). In contrast, the distance-based approach used here, following Mikati et al. (2018) and Boyce and Pastor (2013), incorporates spatial variability by assigning burden based on proximity to pollution sources. The use of a 2.5-mile radius, validated through sensitivity testing by Mikati et al. (2018), improves both the precision of emissions burden estimates and comparability with other studies, lending strength to the generalizability of our findings.

When interpreted together, absolute and proportional burdens offer complementary insights. While national trends show overall declines in absolute burden for most compounds, the persistence of elevated burdens in certain populations indicates that disparities remain entrenched. Eleven compounds showed increasing absolute burdens, suggesting growing exposure for all groups. These upward trends suggest that reductions in national emissions have not translated into uniform environmental health gains. Systemic drivers, such as discriminatory zoning (Mohai and Saha, 2015), weak enforcement in communities of color (Bullard et al., 2008), and political marginalization in siting decisions (Pastor et al., 2001), may continue to shape the distribution of emissions burdens.

Racial and ethnic disparities were evident across both burden metrics. Black and Hispanic populations faced higher absolute burdens for many compounds, consistent with literature linking these disparities in residential proximity to industrial sources (Mikati et al., 2018; Bell and Ebisu, 2012). Clark et al. (2014) found that Black Americans are exposed to 1.5 times more $PM_{2.5}$ than White Americans, even after controlling for income. Similarly, Tessum et al. (2019) demonstrated that non-White groups disproportionately bear exposure burdens resulting from the consumption patterns of White populations.

These patterns are reinforced by research on industrial siting practices, which show that hazardous facilities are more frequently located in communities of color, regardless of socioeconomic status (Mohai and Saha, 2015; Pastor et al., 2001; Taylor et al., 2014). Such siting decisions reflect a history of exclusionary zoning, redlining, and environmental governance practices that continue to influence present-day exposure disparities (Brender et al., 2011; Lane et al., 2022).

Proportional burden analysis further clarifies how emissions are distributed relative to population share. These proportional burden results echo prior findings showing that race is a stronger predictor of pollution burden than income alone (Ringquist, 2005; Mohai et al., 2009; Mikati et al., 2018). Even after adjusting for socioeconomic factors, racial and ethnic minorities face disproportionately high burdens, underscoring the structural nature of these disparities.

In many cases, communities with the highest proportional burdens also experienced elevated average burdens, likely intensifying both the scale and inequity of exposure. For example, the NHB subgroup had the highest absolute burdens for several carcinogenic compounds, suggesting not only more exposure overall but exposure in excess of what would be expected based on population share alone. The temporal dimension of this analysis strengthens the evidence base for these conclusions by demonstrating that disparities are not isolated to individual years but are sustained, and in some cases, echoing the intensification over time. This temporal continuity underscores the structural persistence of environmental inequality, leading to support the conclusion that systemic disparities endure despite technological and regulatory improvements (Richmond-Bryant et al., 2020).

The persistence of these disparities reflects more than demographic coincidence. They are the outcome of spatial, political, and regulatory inequities that produce environments where marginalized communities face cumulative risks. The public health consequences of these findings are substantial. Many of the chlorinated compounds analyzed, like trichloroethylene, chloroform, vinyl chloride, and methoxychlor, are linked to serious health effects, including cancer, neurodevelopmental harm, and reproductive toxicity (Rodgers et al., 2018; Negri et al., 2003; Woolcott et al., 2001; Colborn et al., 1993; Choi et al., 2006; Chiu et al., 2013).

Notably, disparities appear to be widening rather than narrowing as it relates to chlorinated compound emissions burdens. While the NHW subgroup has seen decreases in proportional burden across more compounds, NHB, Hispanic, and below-poverty subgroups showed more increases than decreases. These diverging trends suggest that even as total emissions decline, their benefits are not being shared equally. Reductions in emissions burden have disproportionately favored already advantaged groups, reinforcing structural inequality.

Several compounds highlight how inequities persist and evolve. For instance, chlorobenzene emissions burdens increased among NHB populations, deepening the gap relative to NHW populations. In the case of vinyl chloride, absolute burden declined, but proportional burden shifted, decreasing for NHB populations while increasing for Hispanic populations. This reallocation of burden suggests not uniform progress but redistributed risk.

These findings are consistent with previous studies demonstrating that pollution reductions do not necessarily produce equity gains (Hajat et al., 2015; Clark et al., 2014; Tessum et al., 2019). Instead, disparities may persist or shift across subgroups, highlighting the need for environmental policy to consider not only total reductions but distributive outcomes.

Reducing emissions alone is insufficient to resolve environmental injustices, yet existing environmental regulations continue to treat aggregate emissions reductions as indicators of success, potentially obscuring persistent disparities in who remains exposed. Many chlorinated compounds originate from sectors with long histories of regulatory complexity: chemical manufacturing, dry cleaning, plastics production, pesticide formulation, and metal degreasing (Huang et al., 2014). Facilities in these industries are often sited in or near marginalized communities, compounding potential health risks due to cumulative exposures.

4.2 Study Limitations and Considerations

Limitations should be considered when interpreting the results of this study. First, our analysis used two different centroid years (2013 and 2022) to calculate absolute and, in turn, proportional burdens. While this approach allowed for comparisons over time, it also introduced a level of instability in the data as some geographic boundaries had changed. Variations between centroid years necessitated the inclusion of error bars to represent uncertainty. For some compounds, these fluctuations resulted in concerning magnitudes of change, complicating the interpretation of burden trends.

Second, data for three compounds were missing data for the 2008 NEI year. The absence of this early data point hinders our ability to present fully consistent temporal analyses for these compounds. Although this limitation affects only a small portion of the data set (0.06%) it may slightly impact the accuracy of trends in both absolute and proportional burdens for the compounds affected.

Third, the use of broad racial and ethnic categories such as NHB, Hispanic, and NHW presents an important conceptual limitation. While these categories are necessary for aligning with Census and ACS datasets and allow for broad comparisons across populations, they risk oversimplifying the complex and heterogeneous experiences of different racial and ethnic subgroups. Scholars have noted that aggregated racial categories can obscure within-group differences and reinforce reductive understandings of race and ethnicity (Morello-Frosch and Jesdale, 2006). Nonetheless, these categorizations can still be valuable in identifying persistent disparities that warrant policy attention, as race is often a critical factor in air pollution exposure research (Beard et al., 2024). By demonstrating systemic patterns of disproportionate burden, even within generalized categories, this research supports calls for stronger environmental protections for communities of color and low-income populations.

The interpretation of burden trends in this study must consider both the compounds in context and broader the implications for public health and environmental quality. While this analysis offers valuable insight into changing patterns of chlorinated compound emissions burdens, data quality and spatial assignment choices influence the strength and specificity of those insights. The use of population-weighted absolute burdens in conjunction with proportional burdens allows for a more nuanced understanding of both scale and disparity. However, limitations inherent in emissions reporting, spatial resolution of demographic data, and assumptions underlying proximity-based exposure necessitate cautious interpretation. These findings should therefore be understood as indicative of structural exposure patterns rather than precise individual-level exposure estimates, with implications for how burden trends inform both regulatory and community-level interventions.

CHAPTER 5. CONCLUSIONS

While overall emissions of chlorinated compounds appear to be declining at the national level, this trend does not reflect an equitable distribution of environmental progress. This study highlights that, despite aggregate improvements, structurally disadvantaged populations, particularly Black, Hispanic, and below-poverty communities, continue to experience disproportionately high emissions burdens for chlorinated compounds. These disparities are evident across both absolute and proportional burden measures and, in some cases, appear to be widening over time.

Chlorinated compounds, due to their environmental persistence, toxicity, and well-documented associations with adverse health outcomes, represent a particularly urgent focus for public health and regulatory intervention. Addressing these inequities, from an environmental justice perspective, requires not only sustained reductions in overall emissions but also the implementation of targeted strategies that explicitly account for geographic, racial, and socioeconomic disparities in exposure.

These findings underscore the need for policies and enforcement mechanisms that prioritize environmental protections for historically burdened communities and advance a more equitable and health-conscious outline for chlorinated compound regulation. While current federal permitting processes, like those under the CAA, may consider environmental justice impacts, there is no real enforcement mechanism, leading to challenges in integrating equity considerations into facility permitting, emissions monitoring, and enforcement practices that would represent an important step forward. This includes allocating resources for remediation in disproportionately affected areas and developing public health initiatives that specifically address the cumulative risks faced by marginalized populations. By embedding environmental justice into the regulatory landscape, future policies can move beyond uniform emission reductions toward meaningful reductions in exposure disparities.

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APPENDICES

Appendix A

Table A1. A full list of chlorinated compounds considered in this study. Note. All chlorinated compounds can be found in the National Emissions Inventory and are present on the Environmental Protection Agency's Initial List of Hazardous Air Pollutants with Modifications.

1,1,2-Trichloroethane
1,1,2,2-Tetrachloroethane
1,2-Dibromo-3-chloropropane
1,2,4-Trichlorobenzene
1,3-Dichloropropene
1,4-Dichlorobenzene
2-Chloroacetophenone
2-Chlorobiphenyl (PCB-1)
2,4,4-Trichlorobiphenyl (PCB-28)
2,4,5-Trichlorophenol
2,4,6-Trichlorophenol
Allyl chloride
Benzyl chloride
Bis(chloromethyl)ether
Carbon tetrachloride
Chlordane
Chlorine
Chloroacetic acid
Chlorobenzene
Chloroform
Chloromethyl methyl ether
Chloroprene
Decachlorobiphenyl (PCB-209)
Dibenzofuran
Dichloroethyl ether (Bis(2-chloroethyl)ether)
Dichlorvos
Dimethylcarbamoyl chloride
Ethyl chloride (Chloroethane)
Ethylene dichloride (1,2-Dichloroethane)
Ethylidene dichloride (1,1-Dichloroethane)
Heptachlor
Hexachlorobenzene
Hexachlorobutadiene
Hexachlorocyclopentadiene
Hexachloroethane

Table A1 (continued).

Hydrochloric acid
Methoxychlor
Methyl chloride (Chloromethane)
Methyl chloroform (1,1,1-Trichloroethane)
Methylene chloride (Dichloromethane)
Pentachloronitrobenzene (Quintobenzene)
Pentachlorophenol
Polychlorinated biphenyls
Propylene dichloride (1,2-Dichloropropane)
Tetrachloroethylene (Perchloroethylene)
Titanium tetrachloride
Toxaphene (chlorinated camphene)
Trichloroethylene
Vinyl chloride
Vinylidene chloride (1,1-Dichloroethylene)

Appendix B

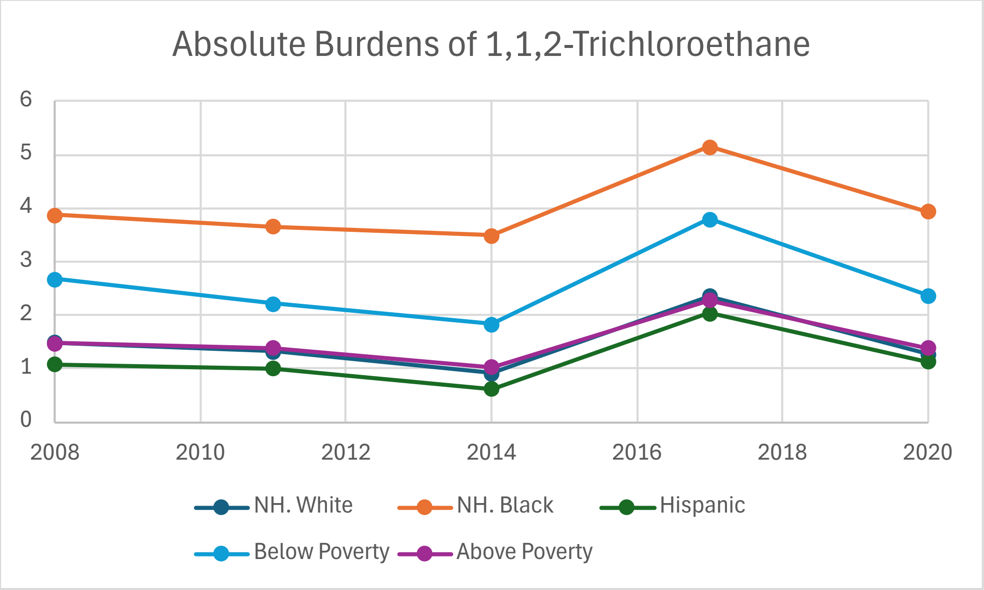

Figure B1. Absolute burdens of 1,1,2-trichloroethane from 2008 to 2020 by population subgroup.

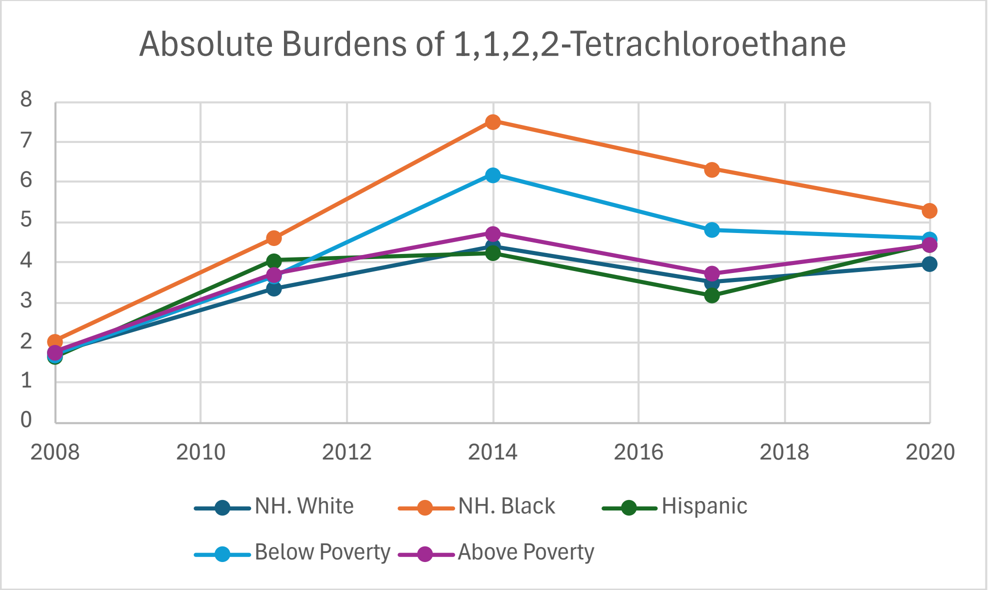

Figure B2. Absolute burdens of 1,1,2,2-tetrachloroethane from 2008 to 2020 by population subgroup.

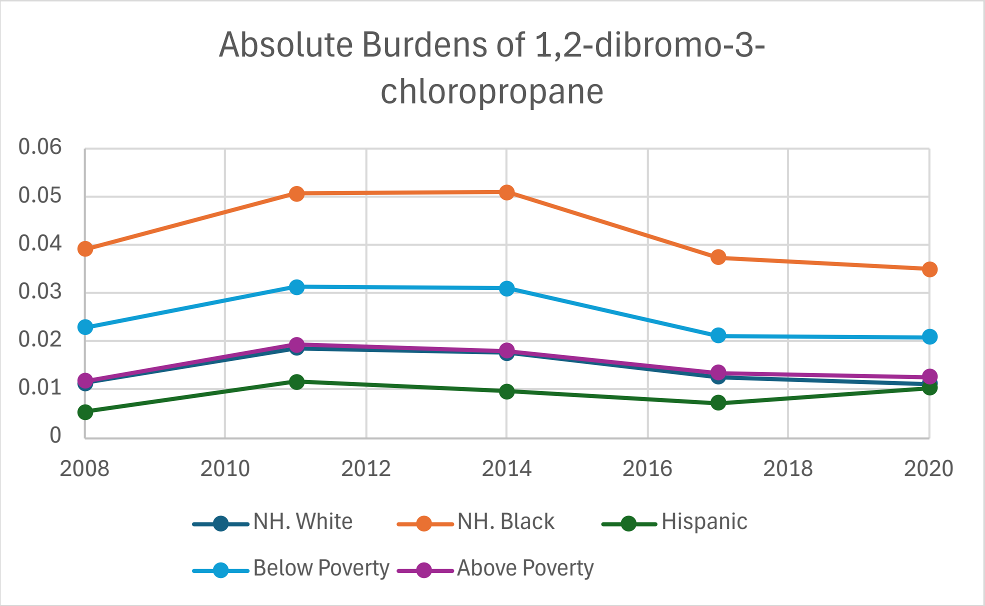

Figure B3. Absolute burdens of 1,2-dibromo-3-chloropropane from 2008 to 2020 by population subgroup.

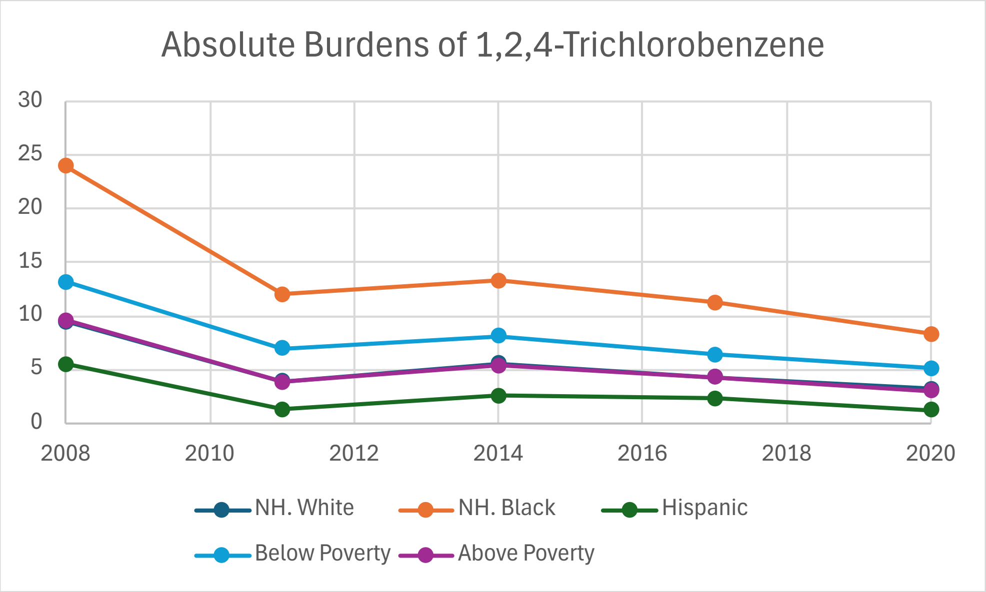

Figure B4. Absolute burdens of 1,2,4-trichlorobenzene from 2008 to 2020 by population subgroup.

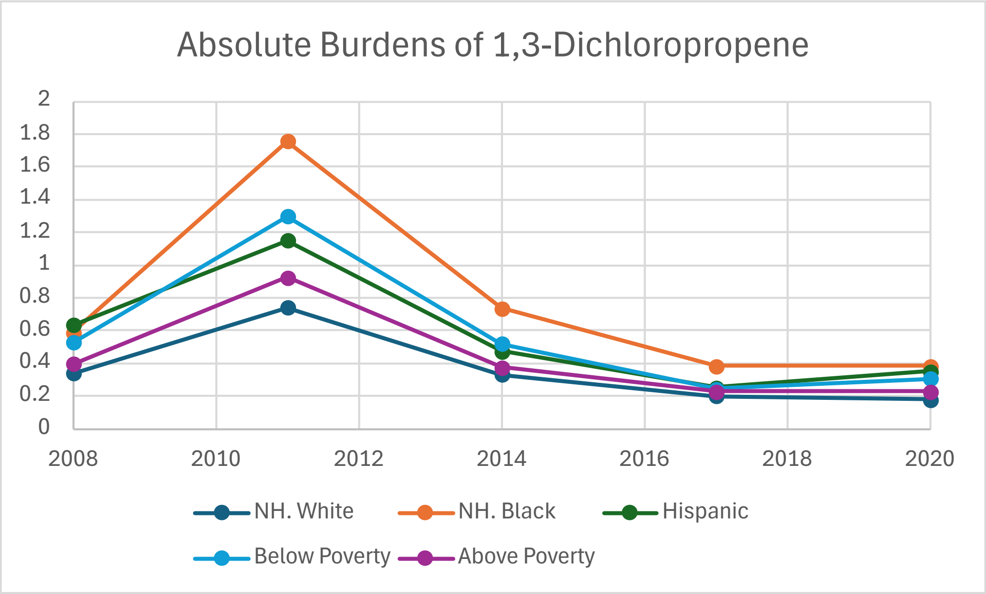

Figure B5. Absolute burdens of 1,3-dichloropropene from 2008 to 2020 by population subgroup.

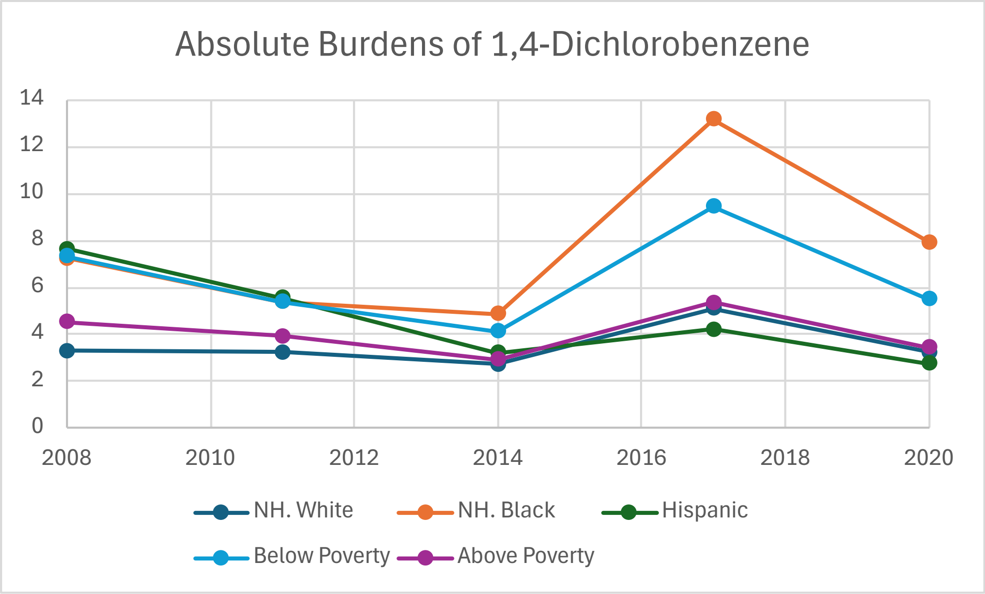

Figure B6. Absolute burdens of 1,4-dichlorobenzene from 2008 to 2020 by population subgroup.

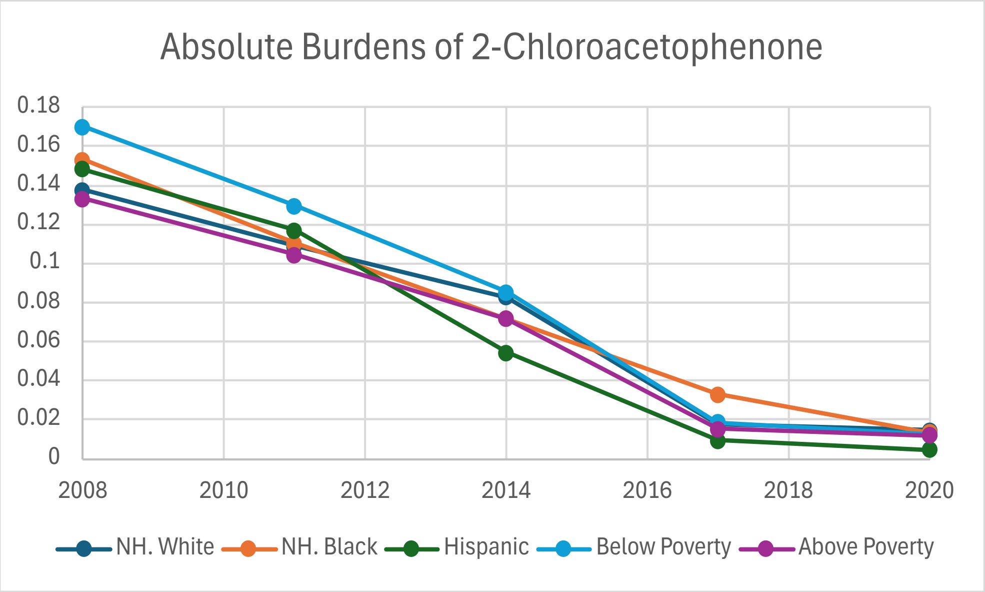

Figure B7. Absolute burdens of 2-chloroacetophenone from 2008 to 2020 by population subgroup.

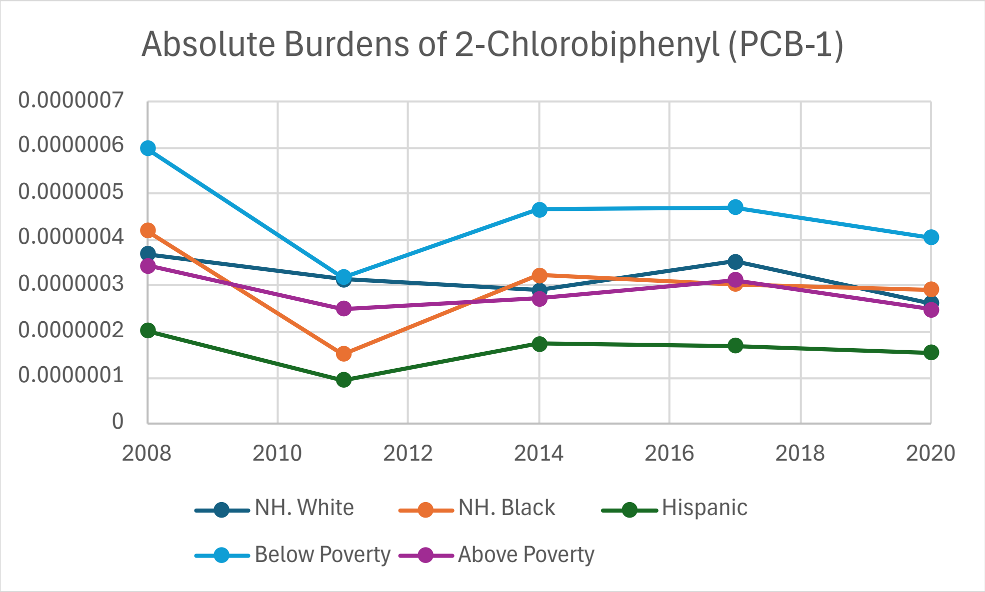

Figure B8. Absolute burdens of 2-chlorobiphenyl (PCB-1) from 2008 to 2020 by population subgroup.

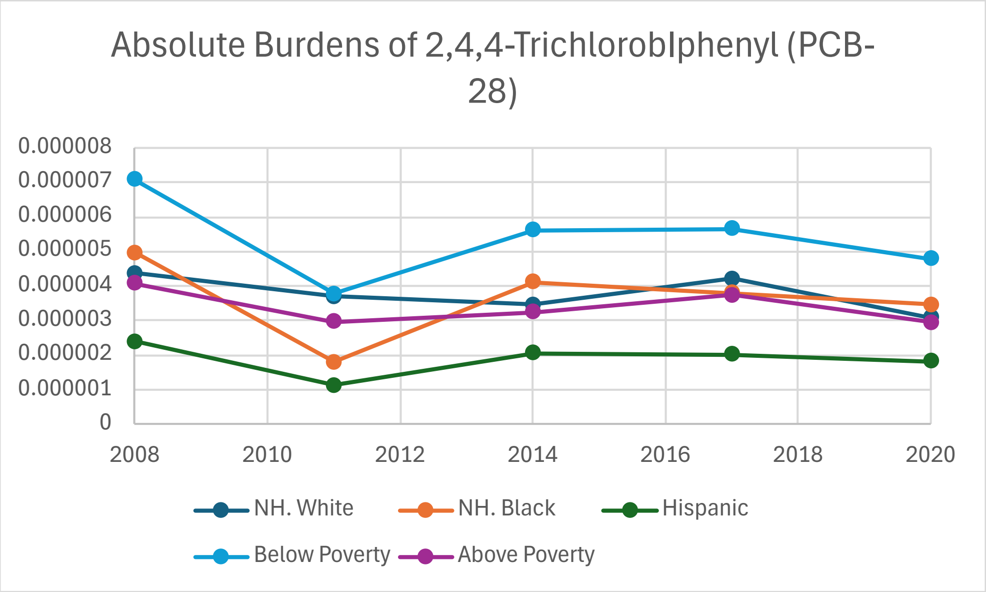

Figure B9. Absolute burdens of 2,4,4-trichlorobiphenyl (PCB-28) from 2008 to 2020 by population subgroup.

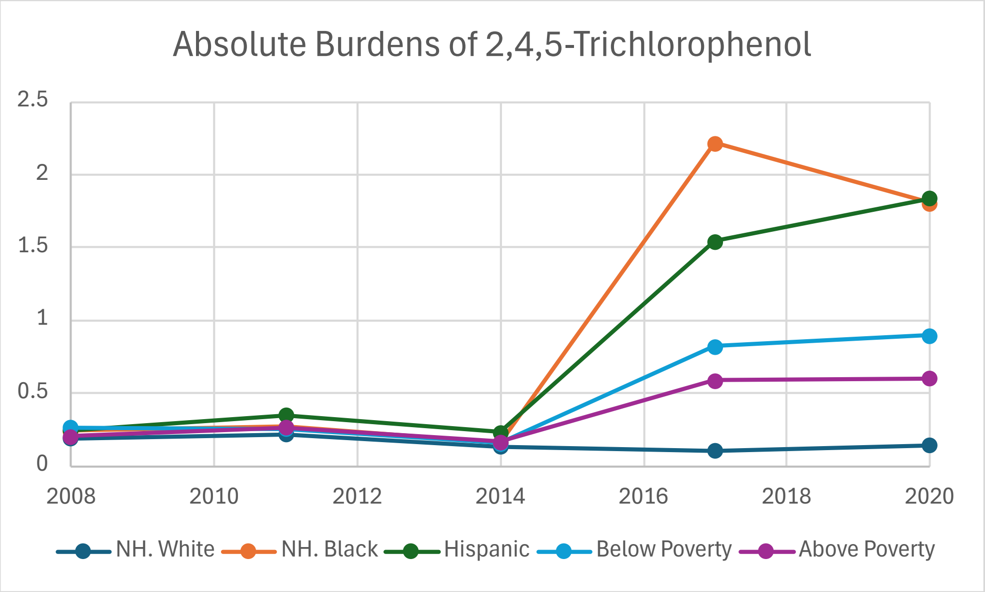

Figure B10. Absolute burdens of 2,4,5-trichlorophenol from 2008 to 2020 by population subgroup.

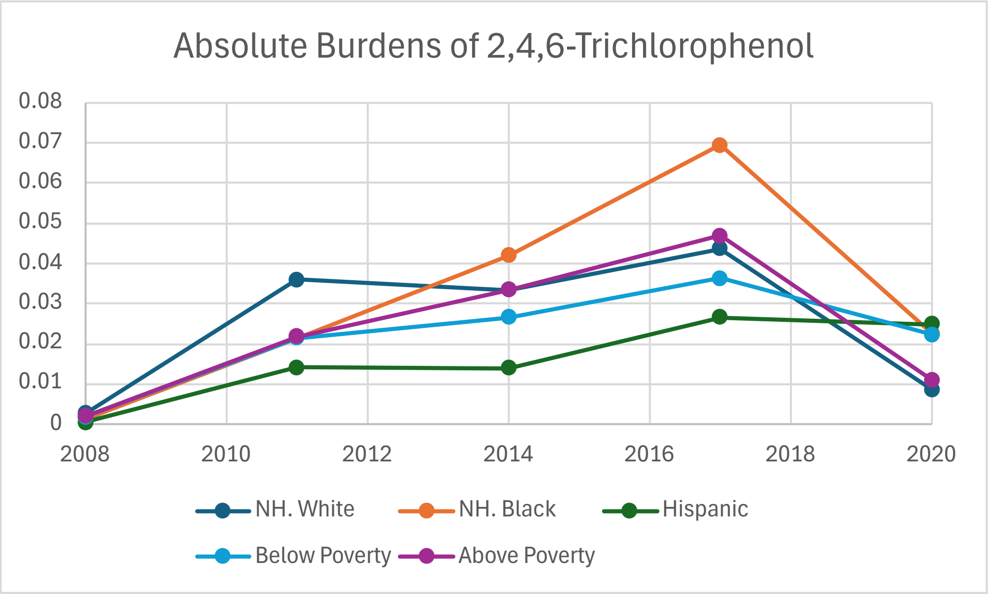

Figure B11. Absolute burdens of 2,4,6-trichlorophenol from 2008 to 2020 by population subgroup.

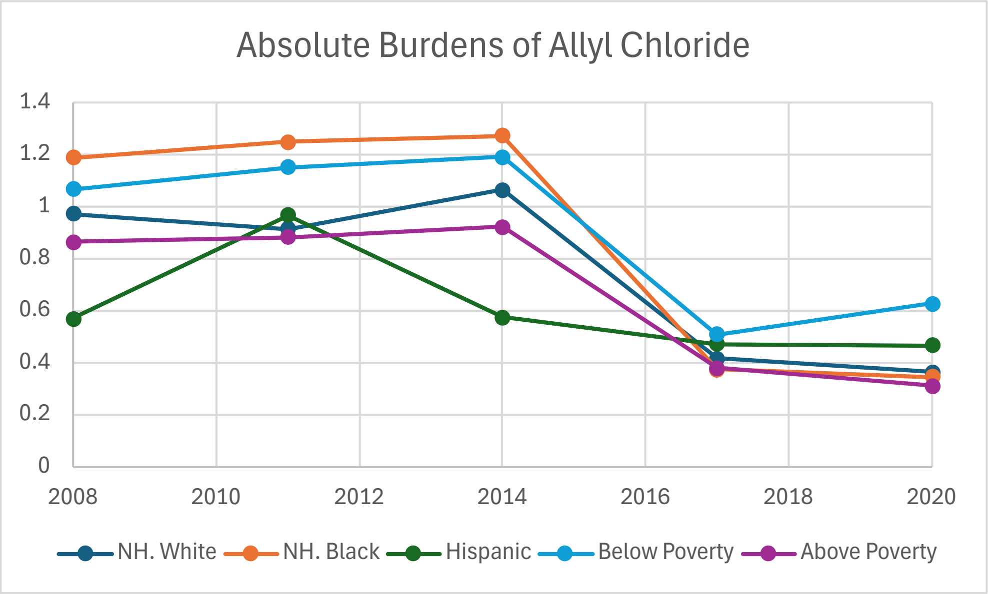

Figure B12. Absolute burdens of allyl chloride from 2008 to 2020 by population subgroup.

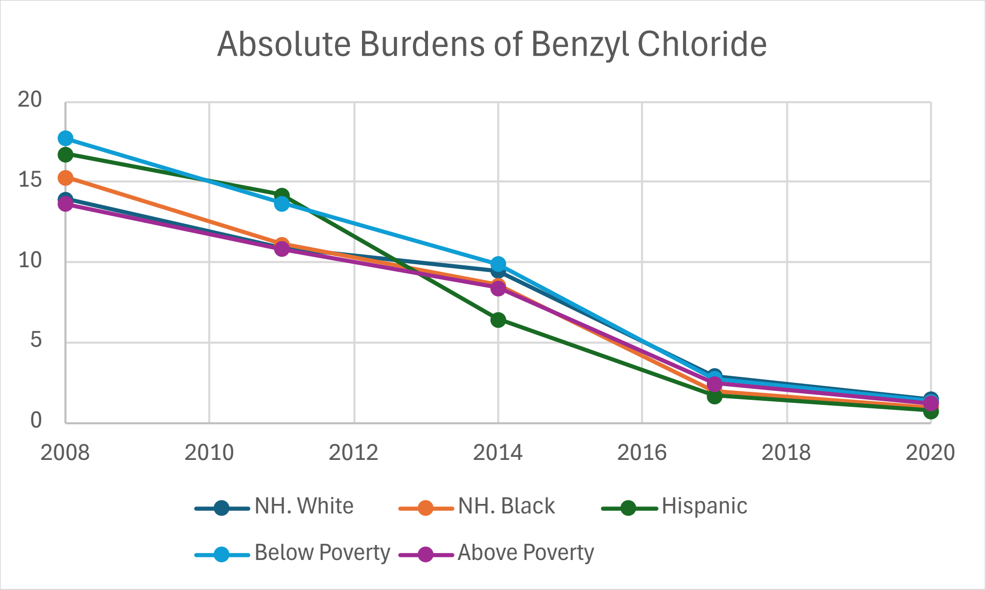

Figure B13. Absolute burdens of benzyl chloride from 2008 to 2020 by population subgroup.

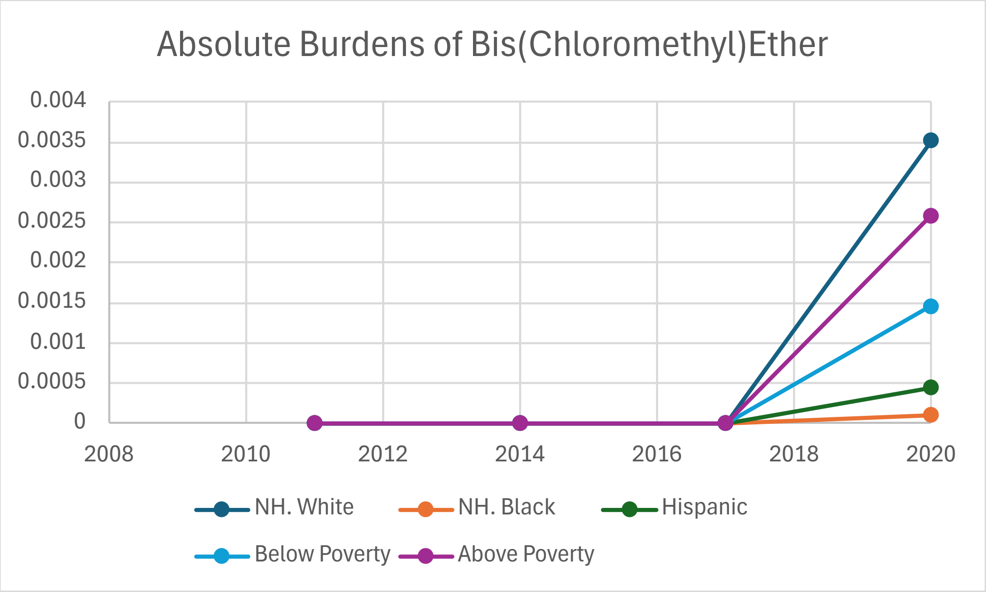

Figure B14. Absolute burdens of bis(chloromethyl)ether from 2008 to 2020 by population subgroup. Note. 2008 was not available within NEI data.

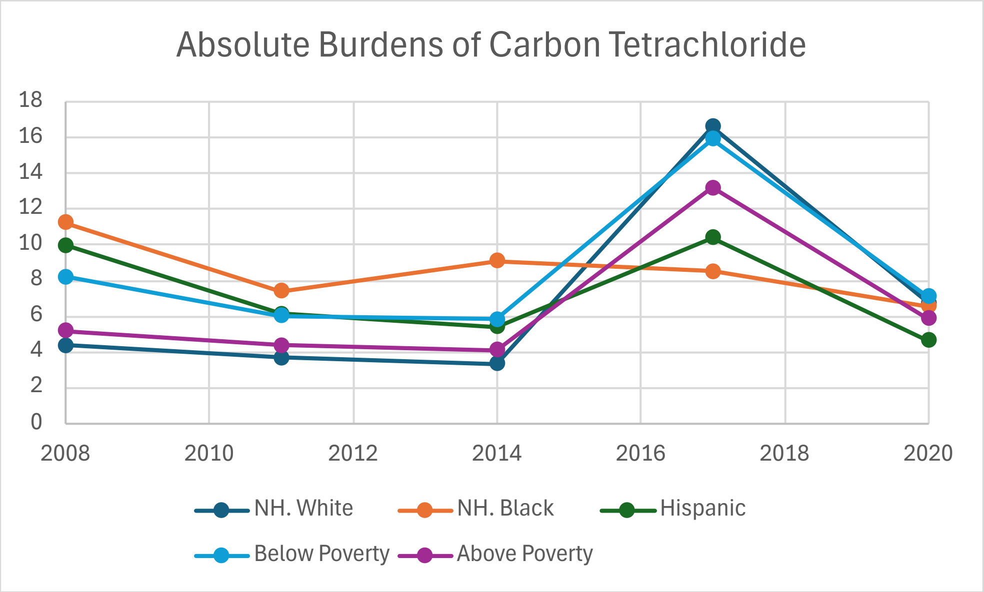

Figure B15. Absolute burdens of carbon tetrachloride from 2008 to 2020 by population subgroup.

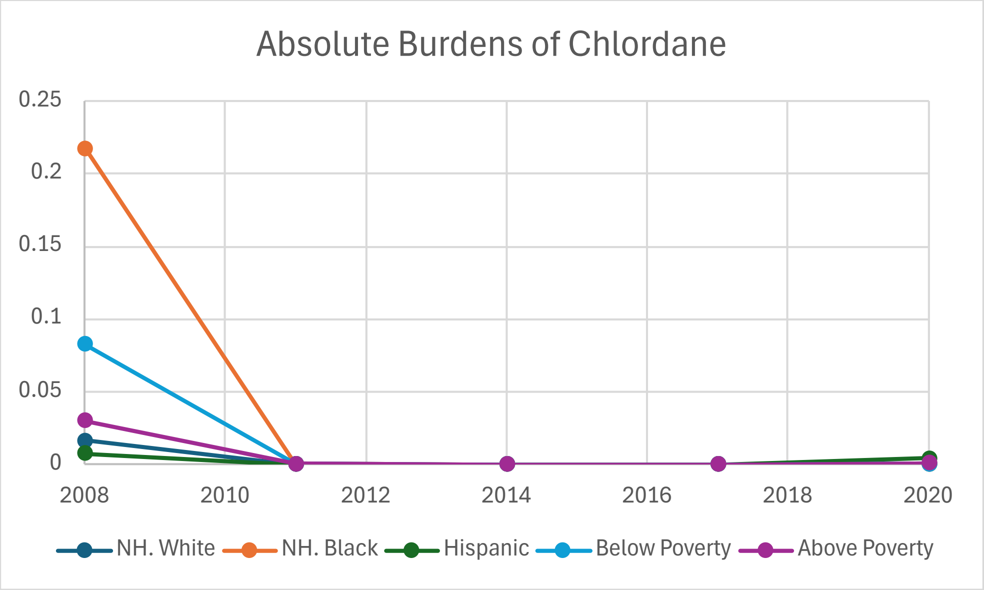

Figure B16. Absolute burdens of chlordane from 2008 to 2020 by population subgroup.

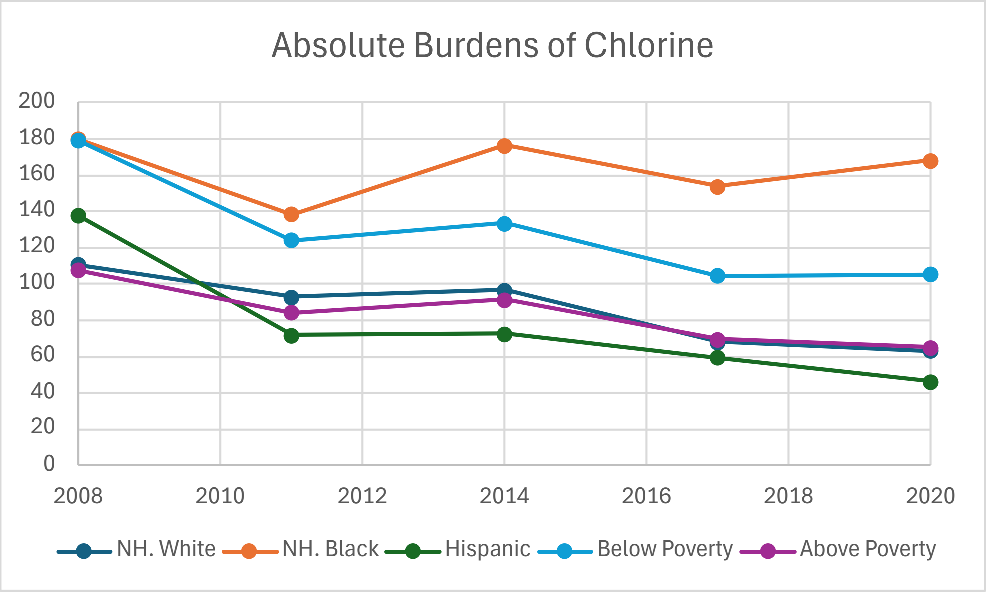

Figure B17. Absolute burdens of chlorine from 2008 to 2020 by population subgroup.

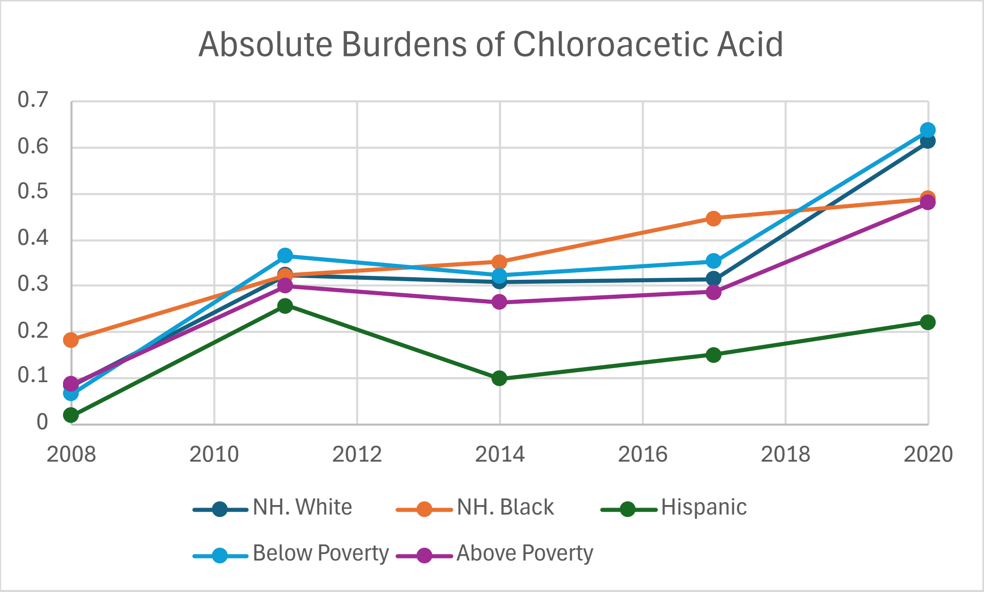

Figure B18. Absolute burdens of chloroacetic acid from 2008 to 2020 by population subgroup.

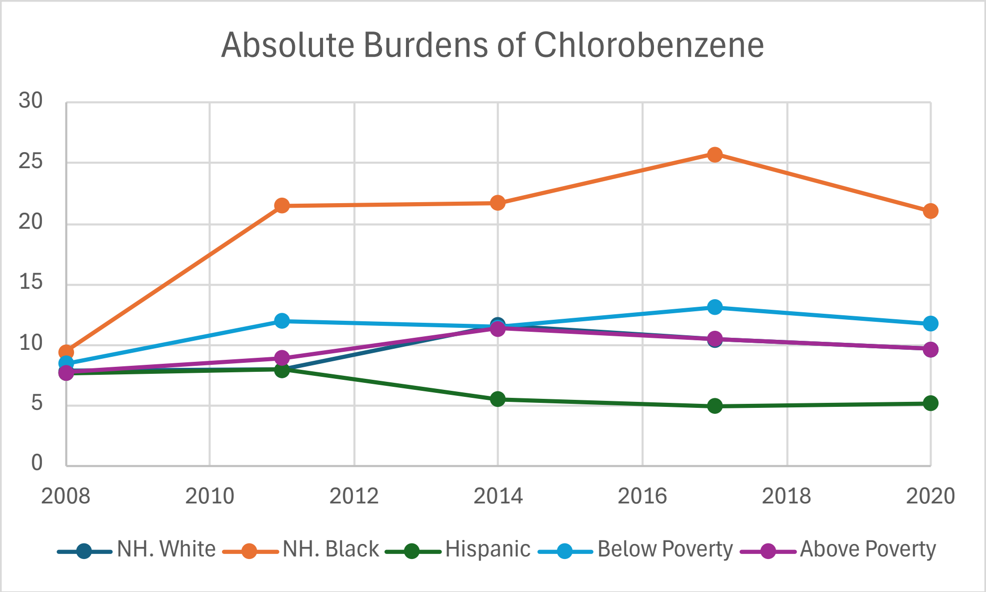

Figure B19. Absolute burdens of chlorobenzene from 2008 to 2020 by population subgroup.

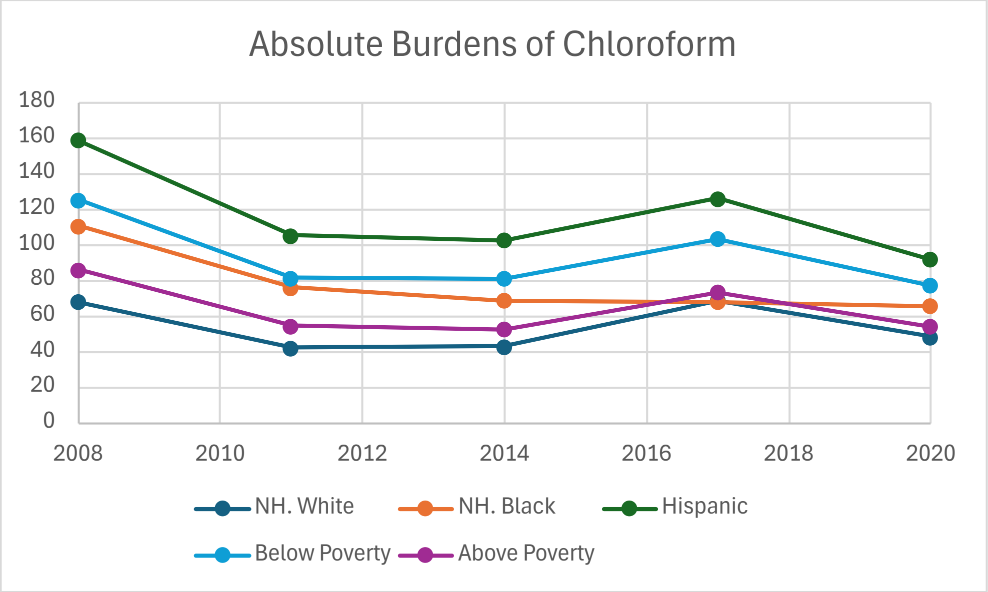

Figure B20. Absolute burdens of chloroform from 2008 to 2020 by population subgroup.

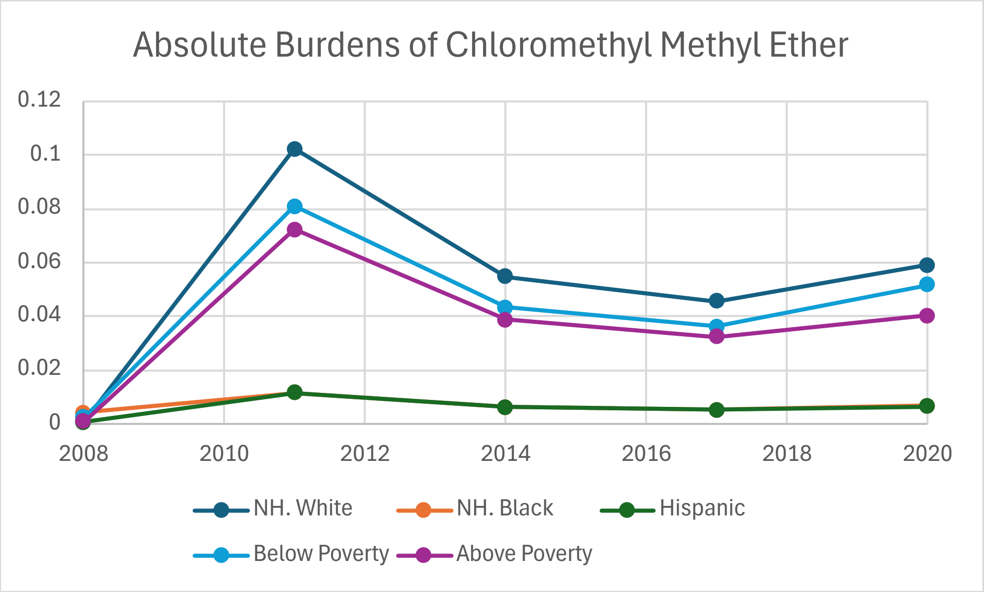

Figure B21. Absolute burdens of chloromethyl methyl ether from 2008 to 2020 by population subgroup.

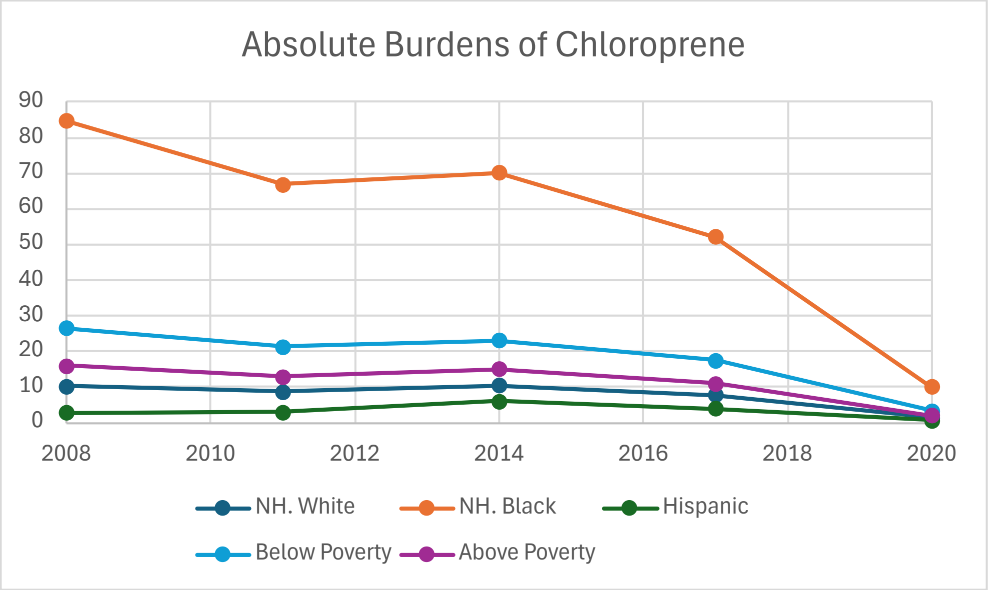

Figure B22. Absolute burdens of chloroprene from 2008 to 2020 by population subgroup.

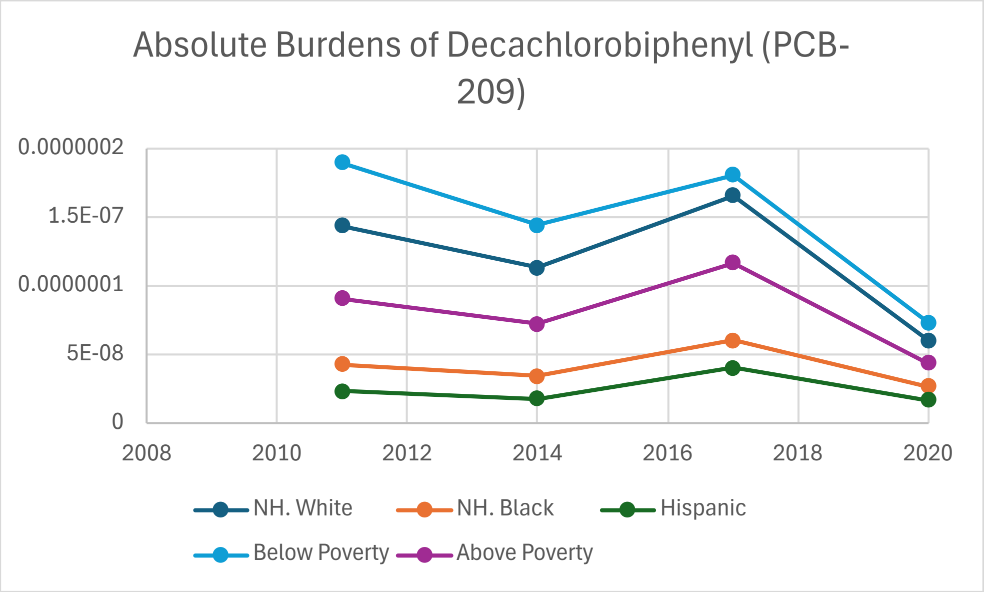

Figure B23. Absolute burdens of decachlorobiphenyl (PCB-209) from 2008 to 2020 by population subgroup. Note. 2008 was not available within NEI data.

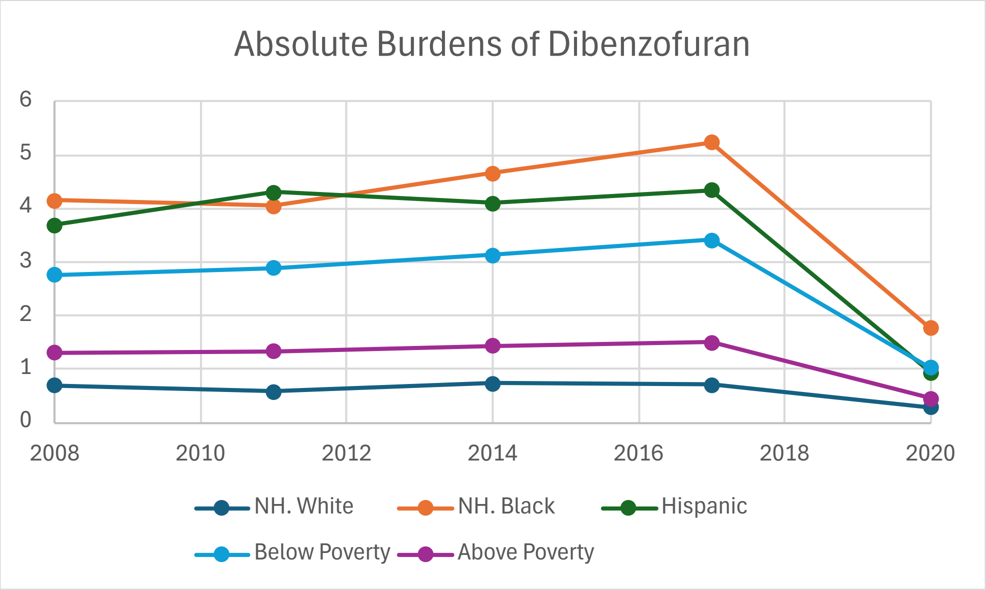

Figure B24. Absolute burdens of dibenzofuran from 2008 to 2020 by population subgroup.

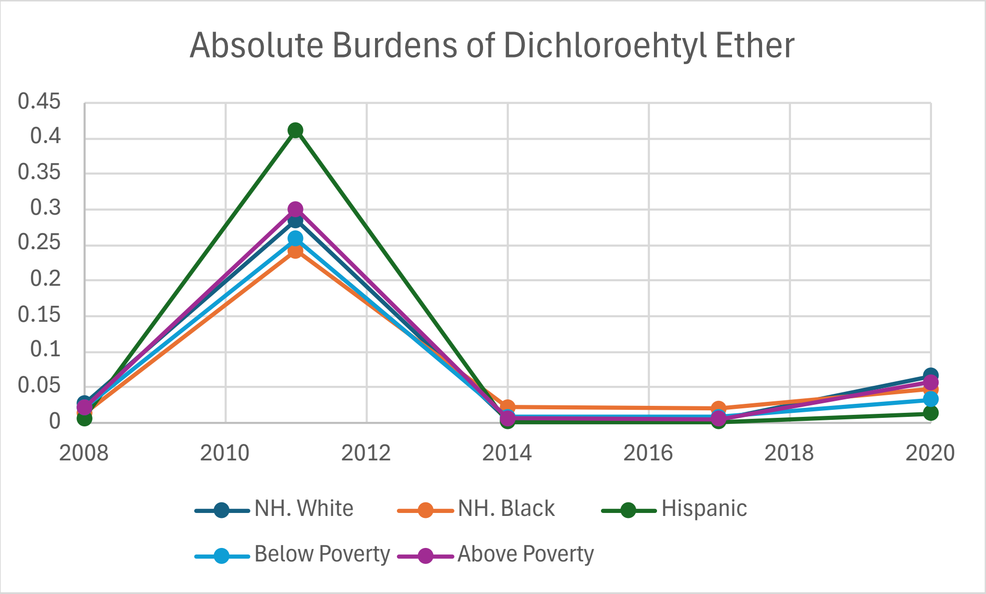

Figure B25. Absolute burdens of dichloroethyl ether from 2008 to 2020 by population subgroup.

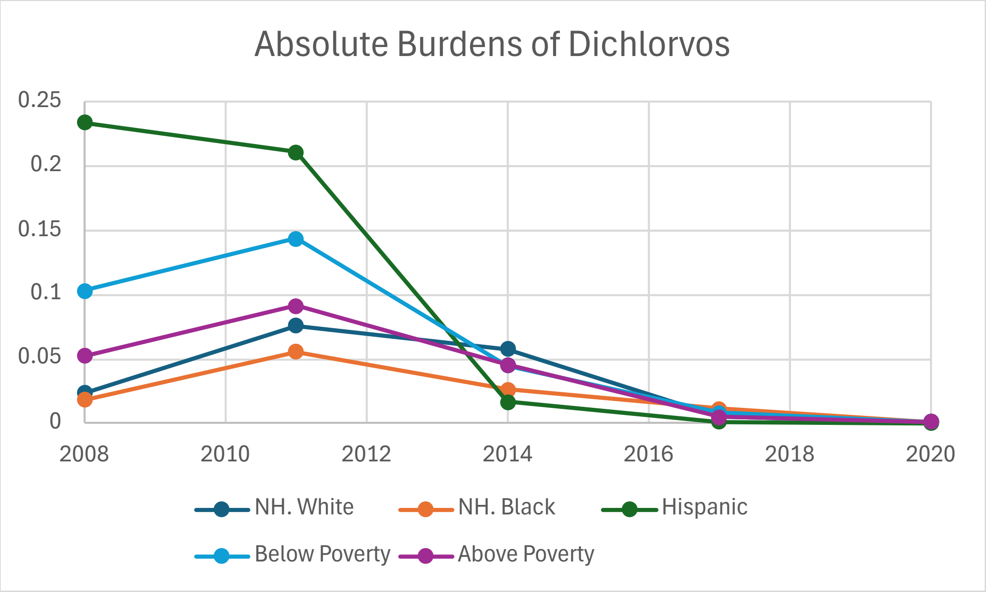

Figure B26. Absolute burdens of dichlorvos from 2008 to 2020 by population subgroup.

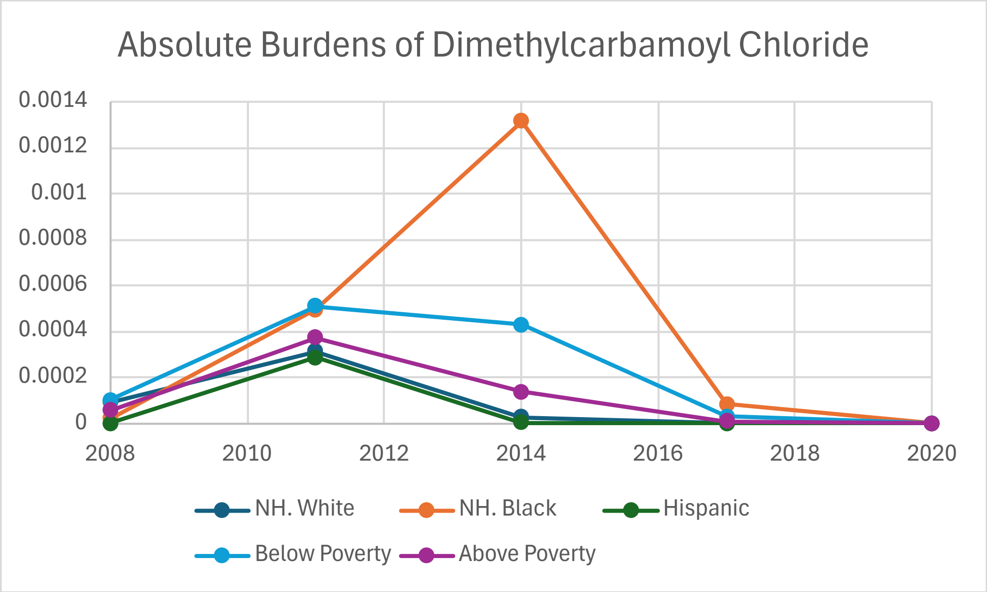

Figure B27. Absolute burdens of dimethylcarbamoyl chloride from 2008 to 2020 by population subgroup.

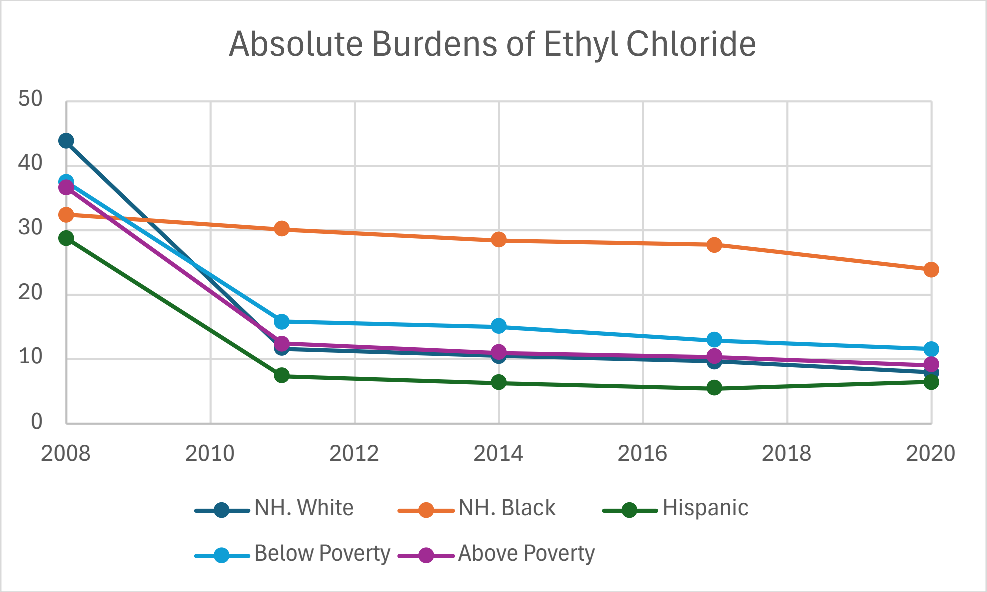

Figure B28. Absolute burdens of ethyl chloride from 2008 to 2020 by population subgroup.

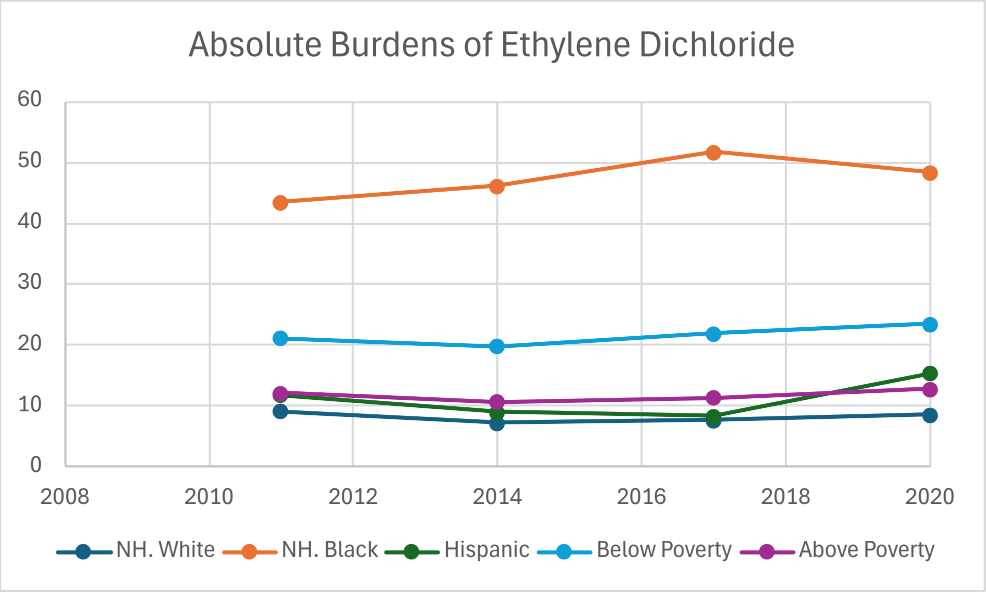

Figure B29. Absolute burdens of ethylene dichloride from 2008 to 2020 by population subgroup. Note. 2008 was not available with NEI data.

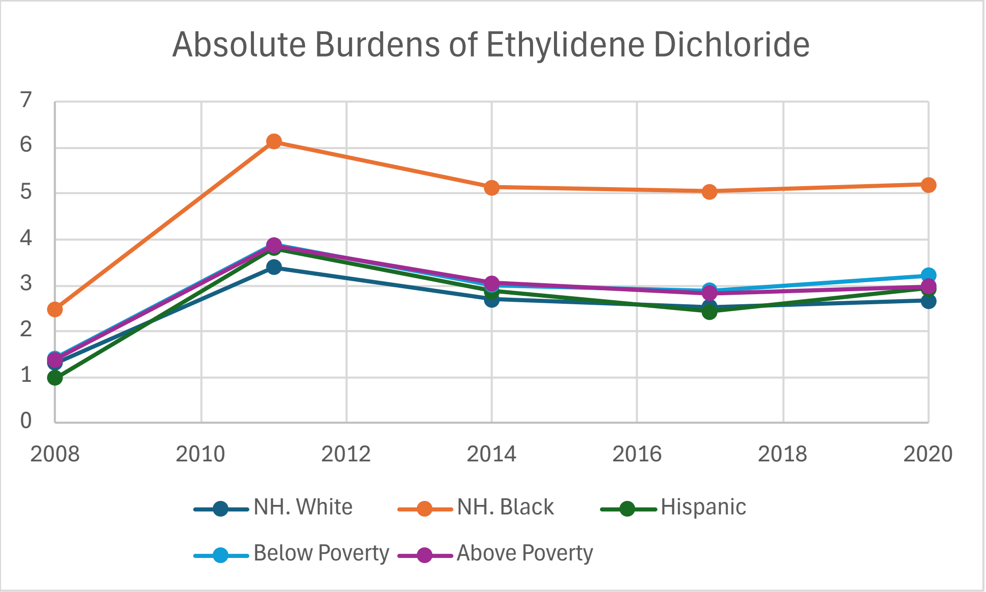

Figure B30. Absolute burdens of ethylidene dichloride from 2008 to 2020 by population subgroup.

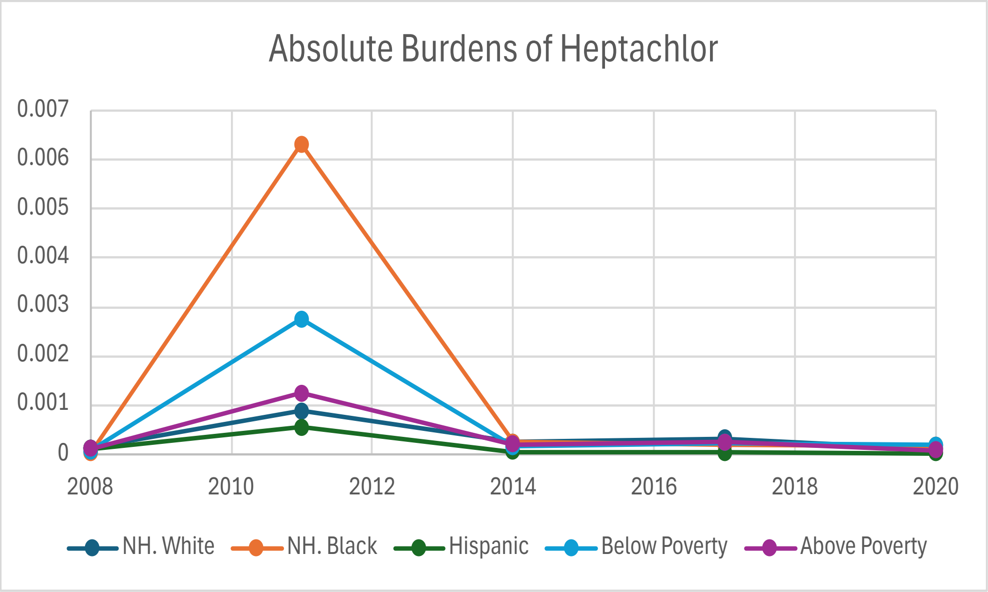

Figure B31. Absolute burdens of heptachlor from 2008 to 2020 by population subgroup.

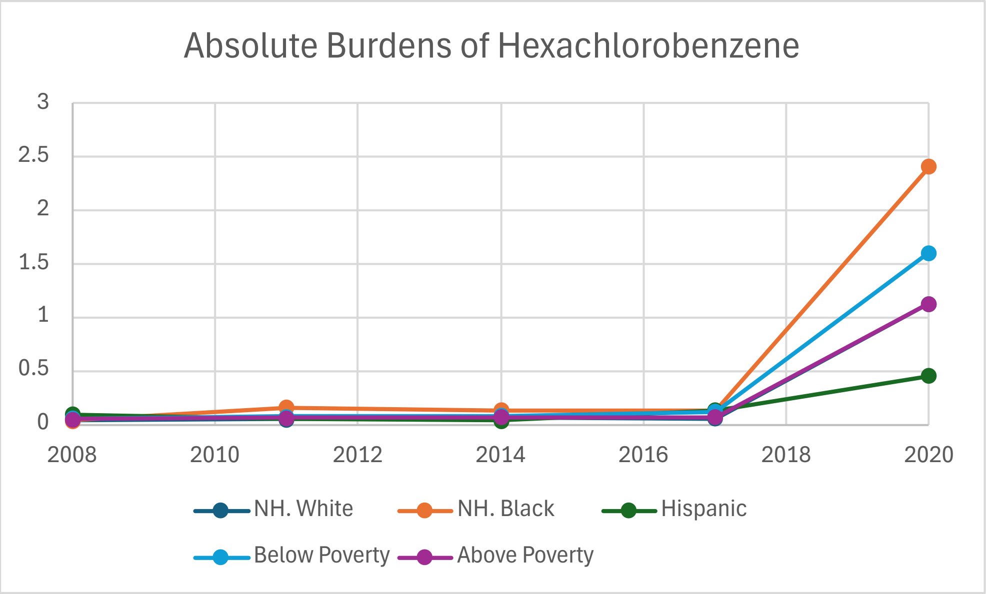

Figure B32. Absolute burdens of hexachlorobenzene from 2008 to 2020 by population subgroup.

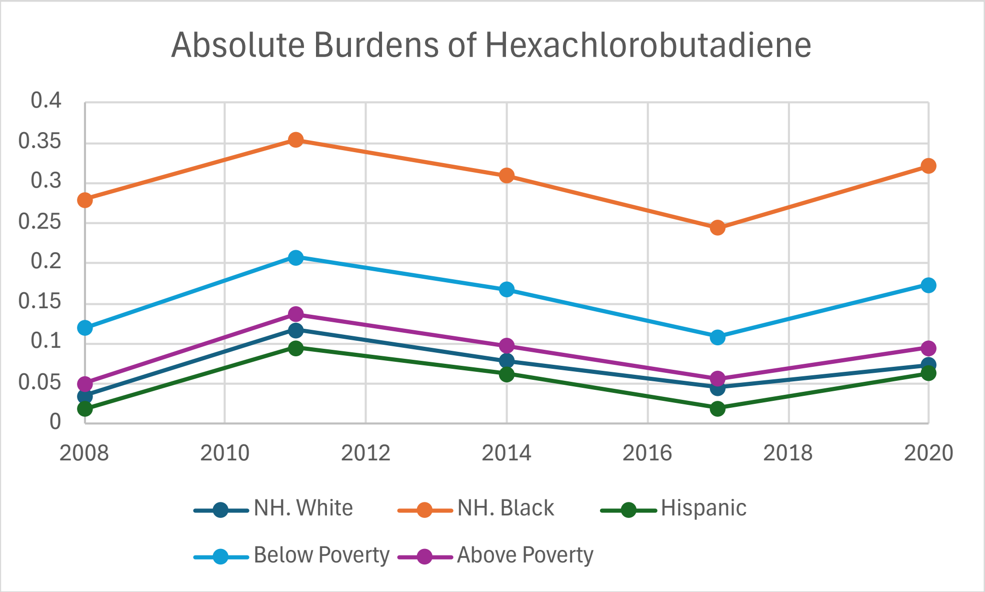

Figure B33. Absolute burdens of hexachlorobutadiene from 2008 to 2020 by population subgroup.

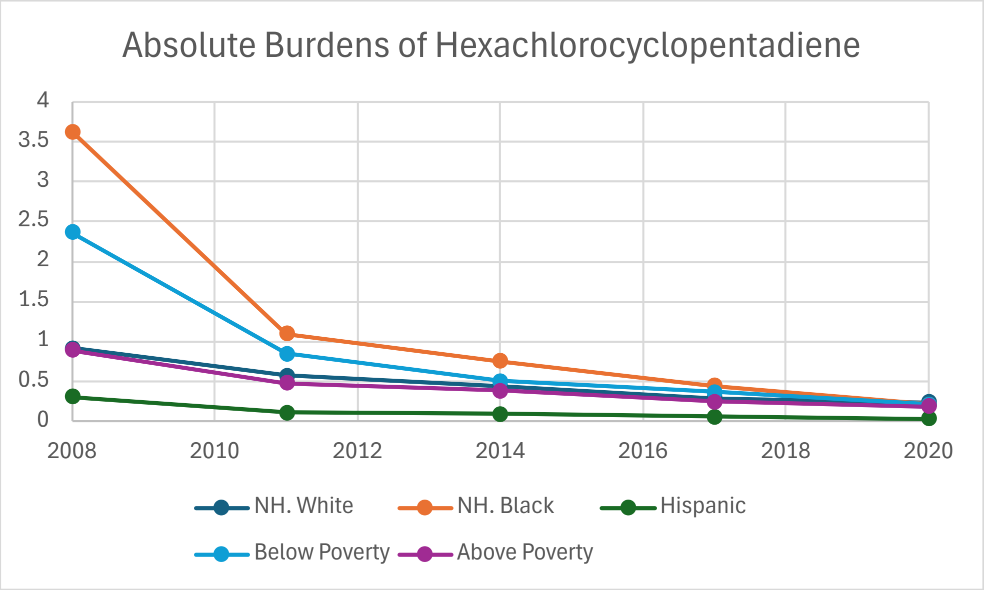

Figure B34. Absolute burdens of hexachlorocyclopentadiene from 2008 to 2020 by population subgroup.

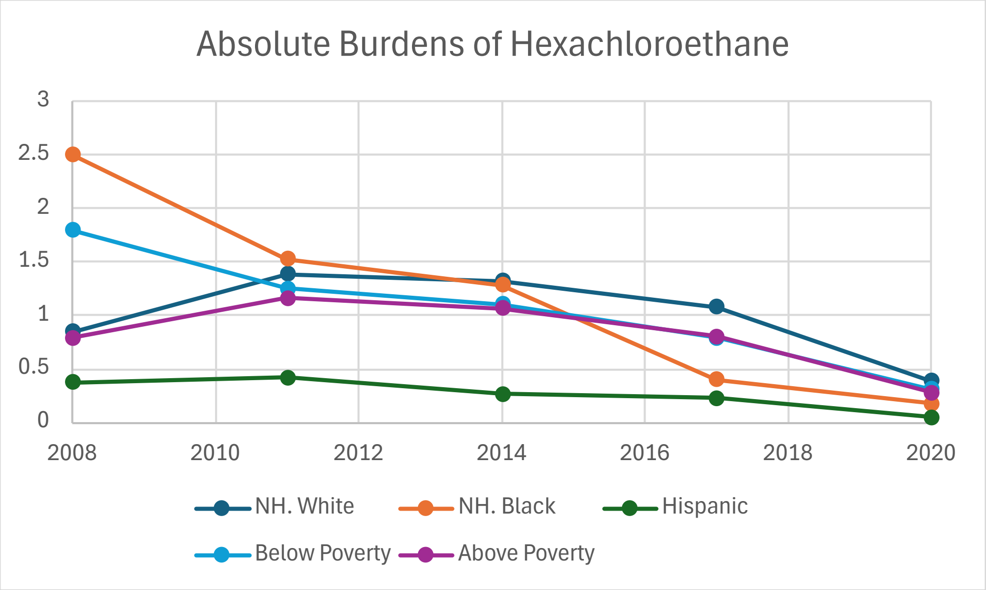

Figure B35. Absolute burdens of hexachloroethane from 2008 to 2020 by population subgroup.

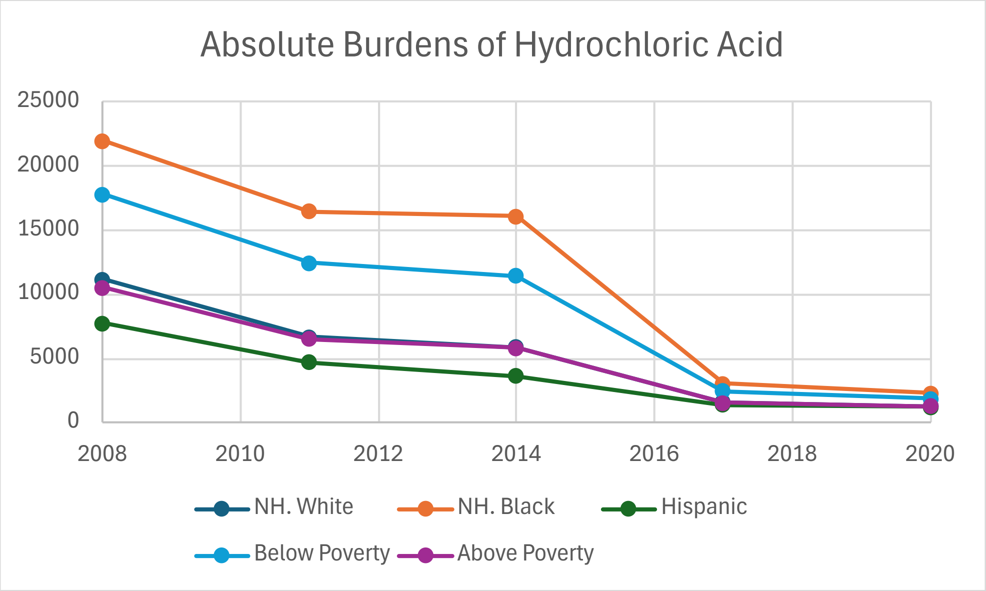

Figure B36. Absolute burdens of hydrochloric acid from 2008 to 2020 by population subgroup.

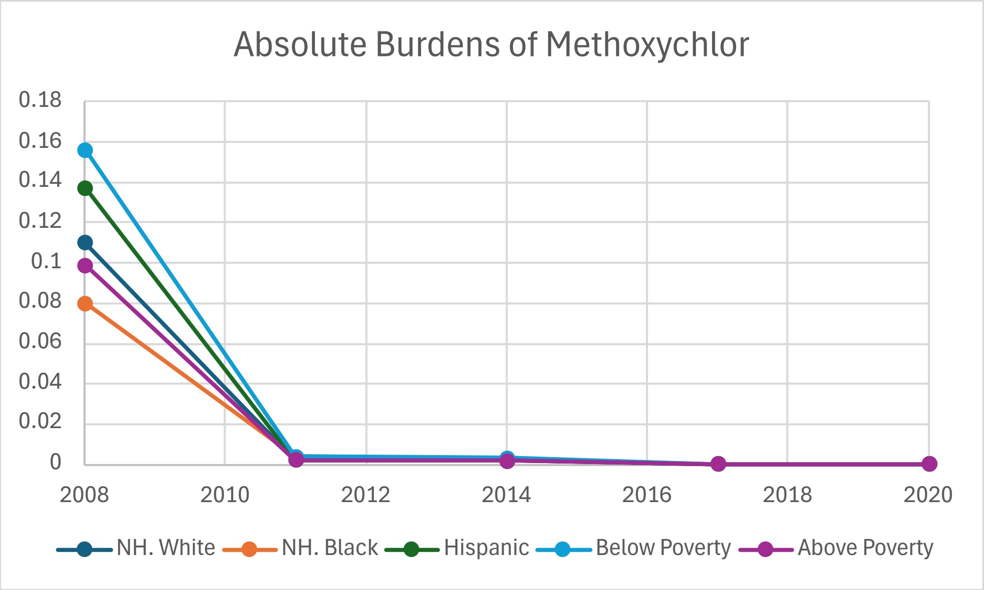

Figure B37. Absolute burdens of methoxychlor from 2008 to 2020 by population subgroup.

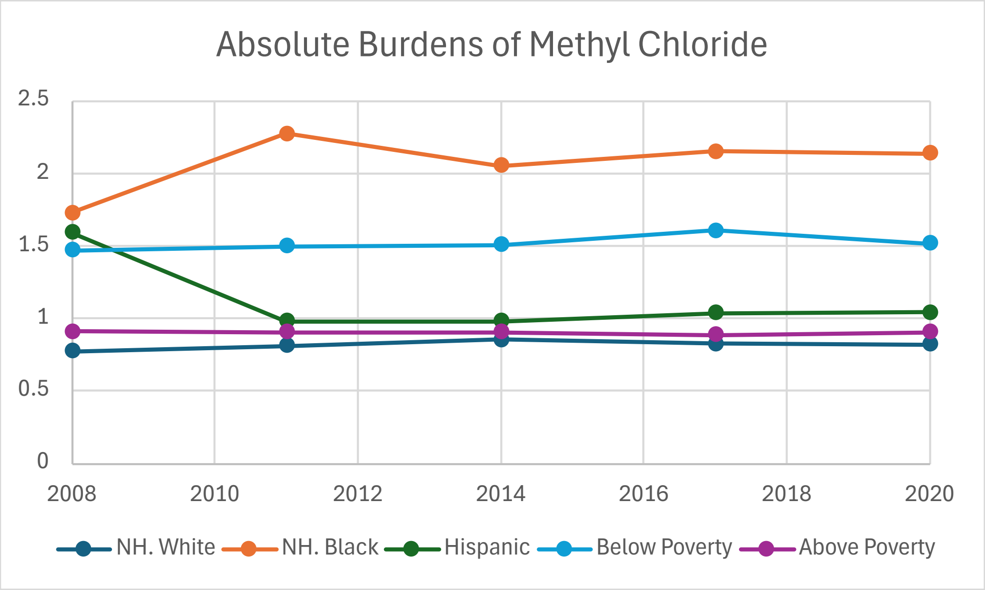

Figure B38. Absolute burdens of methyl chloride from 2008 to 2020 by population subgroup.

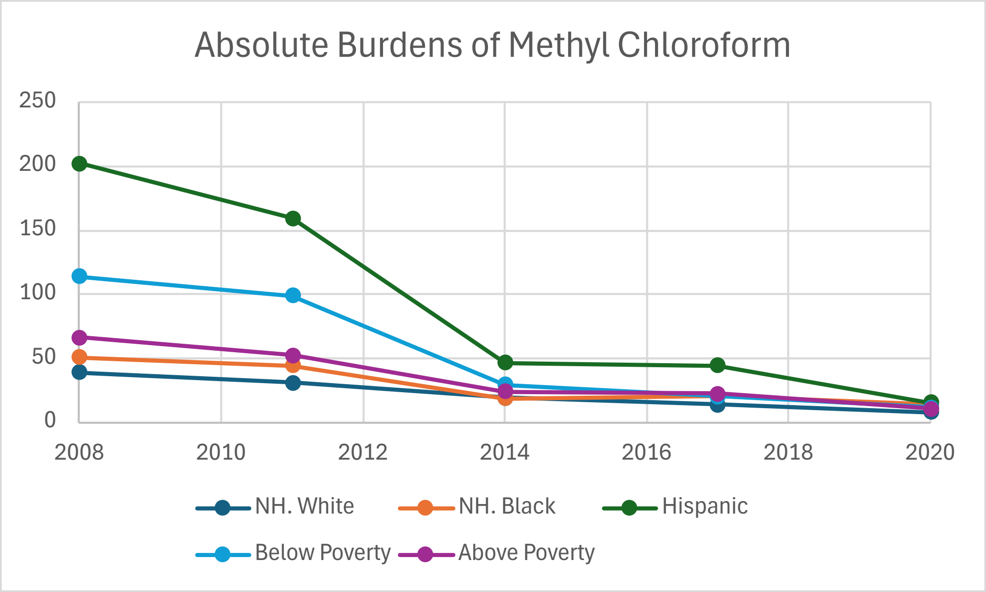

Figure B39. Absolute burdens of methyl chloroform from 2008 to 2020 by population subgroup.

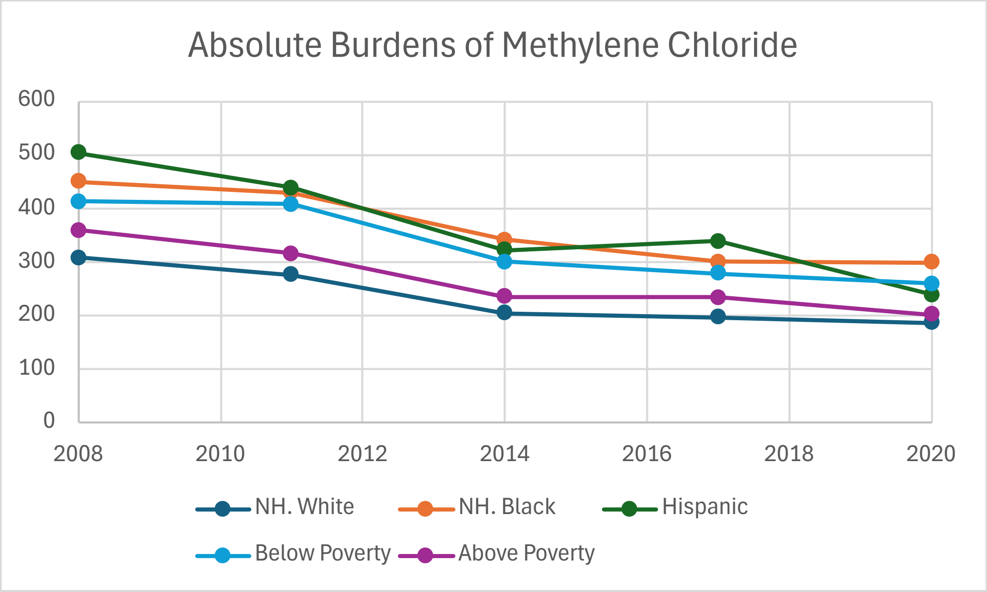

Figure B40. Absolute burdens of methylene chloride from 2008 to 2020 by population subgroup.

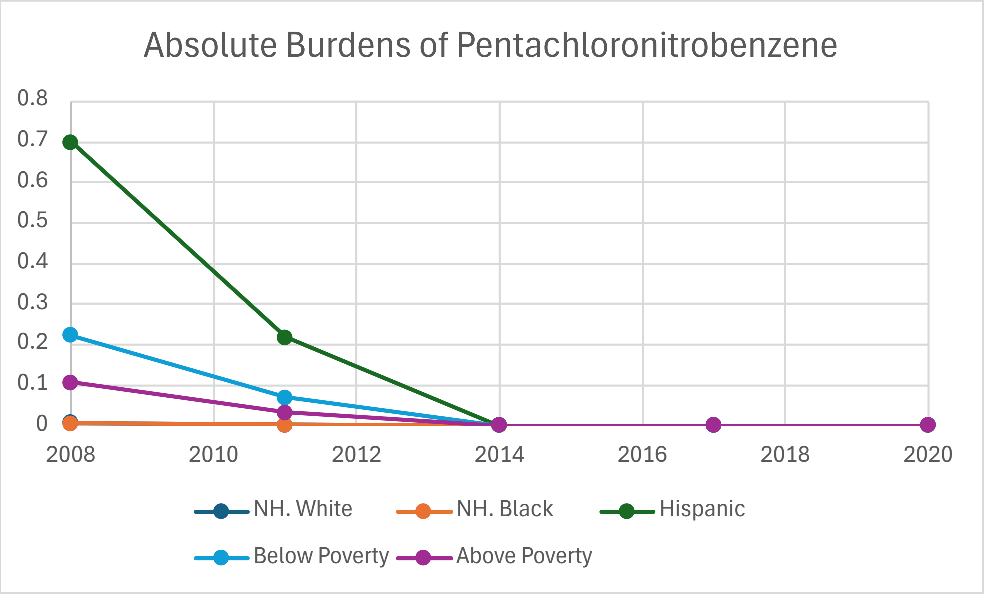

Figure B41. Absolute burdens of pentachloronitrobenzene from 2008 to 2020 by population subgroup.

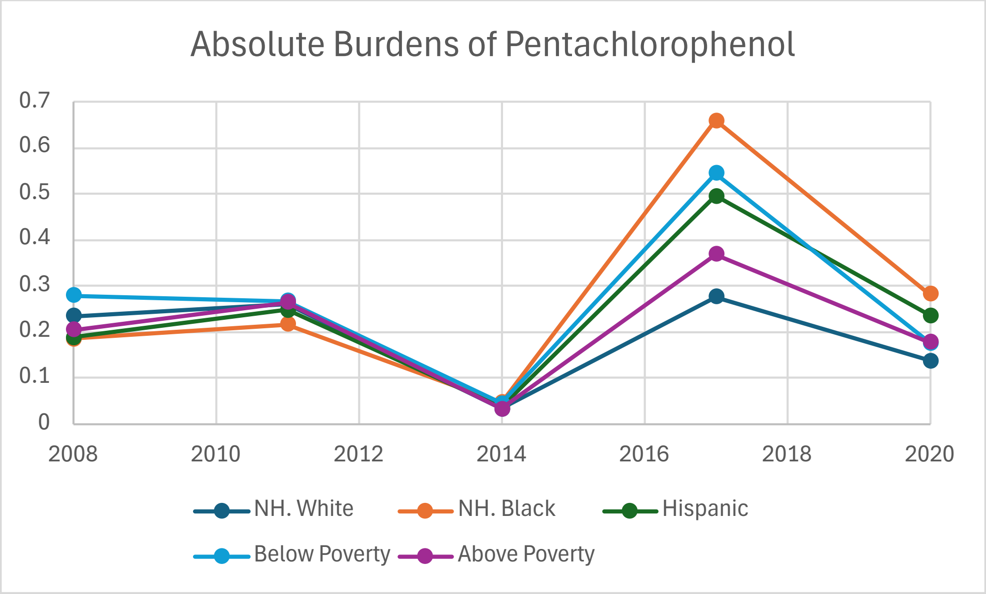

Figure B42. Absolute burdens of pentachlorophenol from 2008 to 2020 by population subgroup.

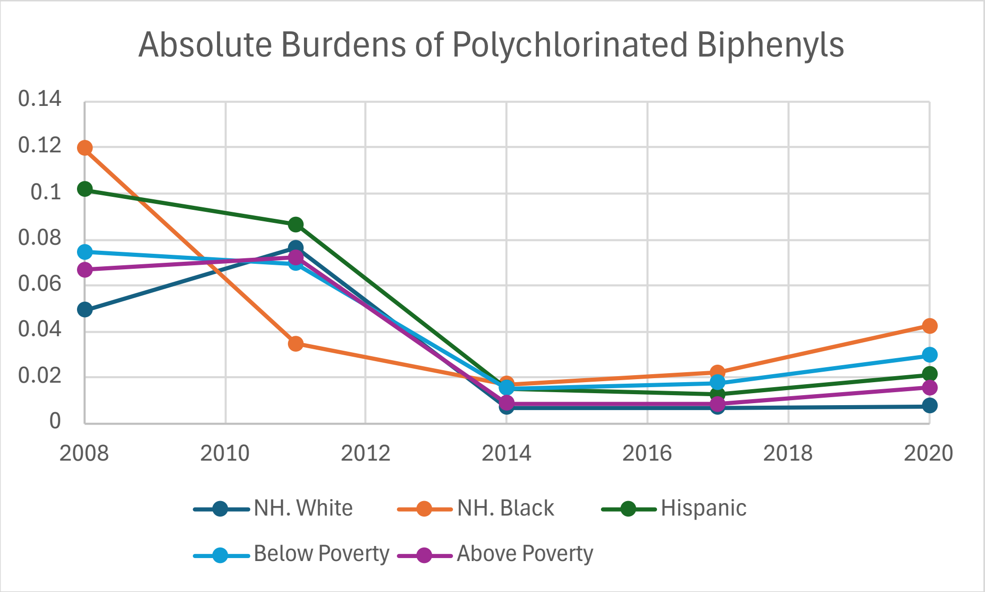

Figure B43. Absolute burdens of polychlorinated biphenyls from 2008 to 2020 by population subgroup.

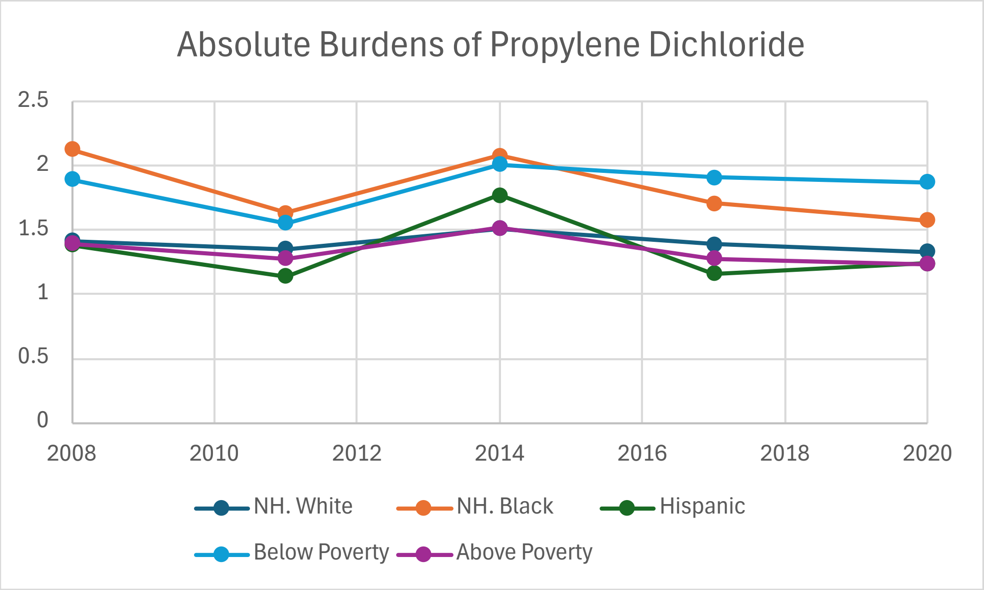

Figure B44. Absolute burdens of propylene dichloride from 2008 to 2020 by population subgroup.

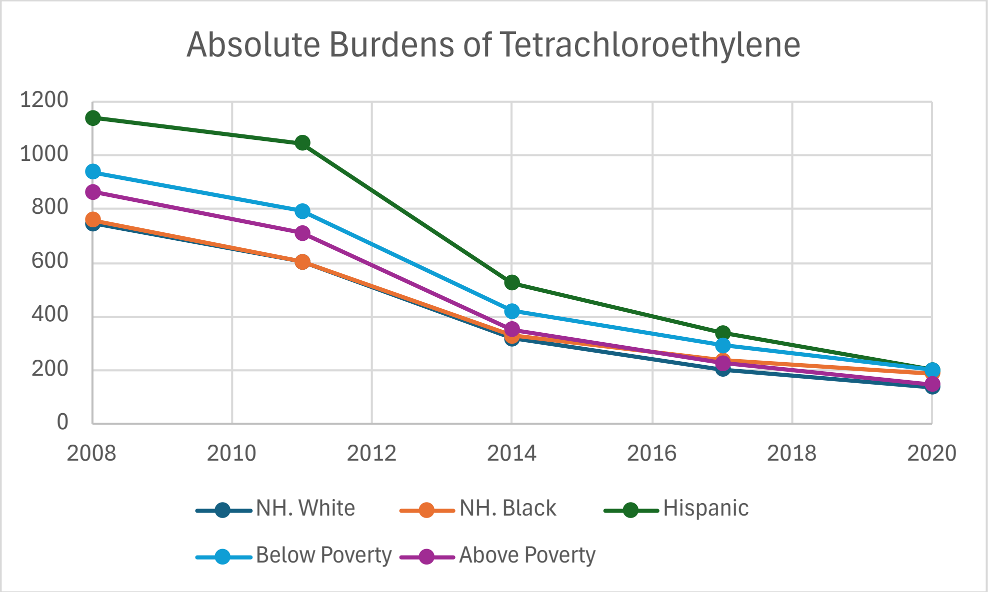

Figure B45. Absolute burdens of tetrachloroethylene from 2008 to 2020 by population subgroup.

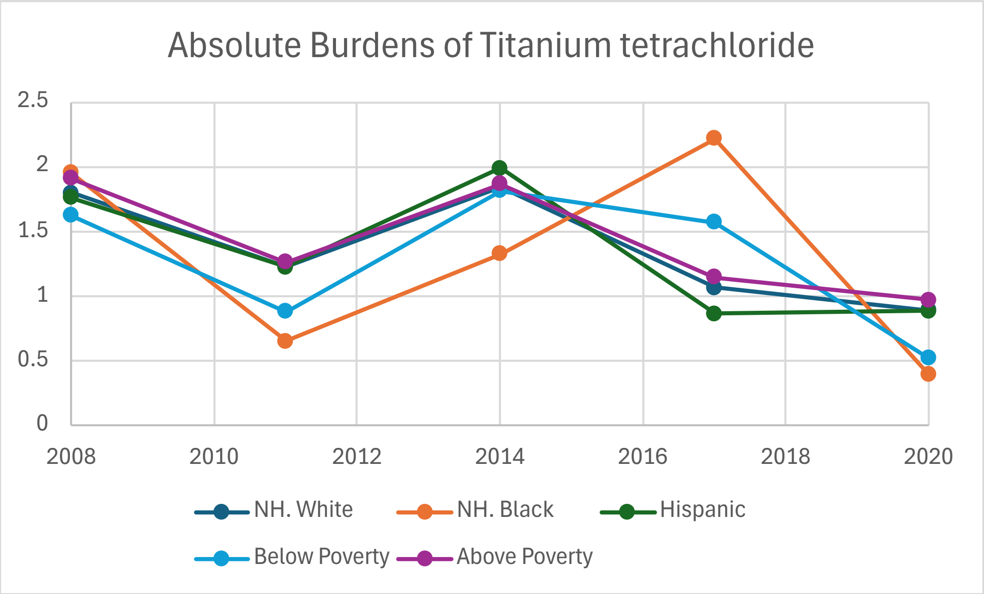

Figure B46. Absolute burdens of titanium tetrachloride from 2008 to 2020 by population subgroup.

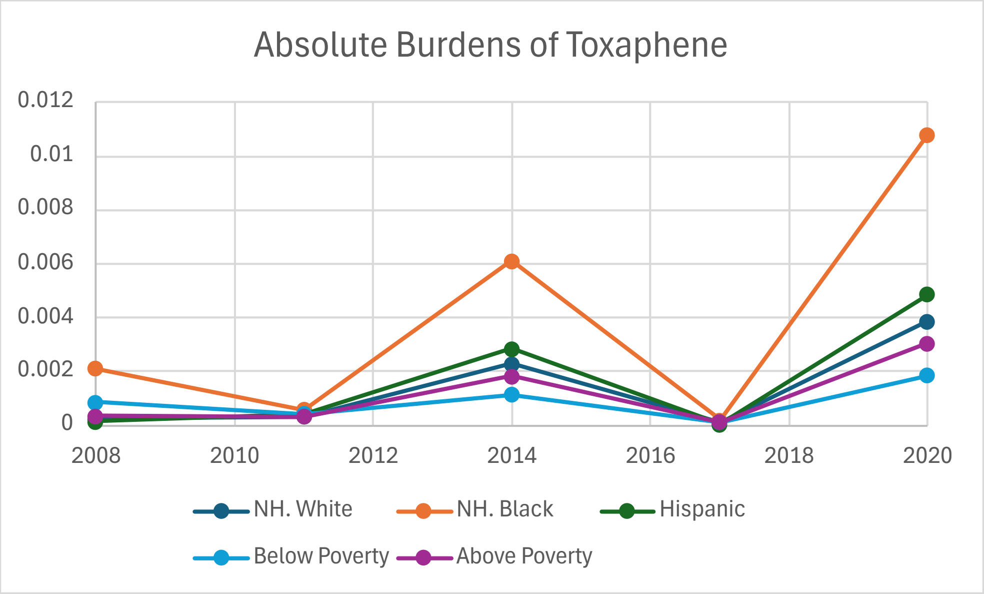

Figure B47. Absolute burdens of toxaphene from 2008 to 2020 by population subgroup.

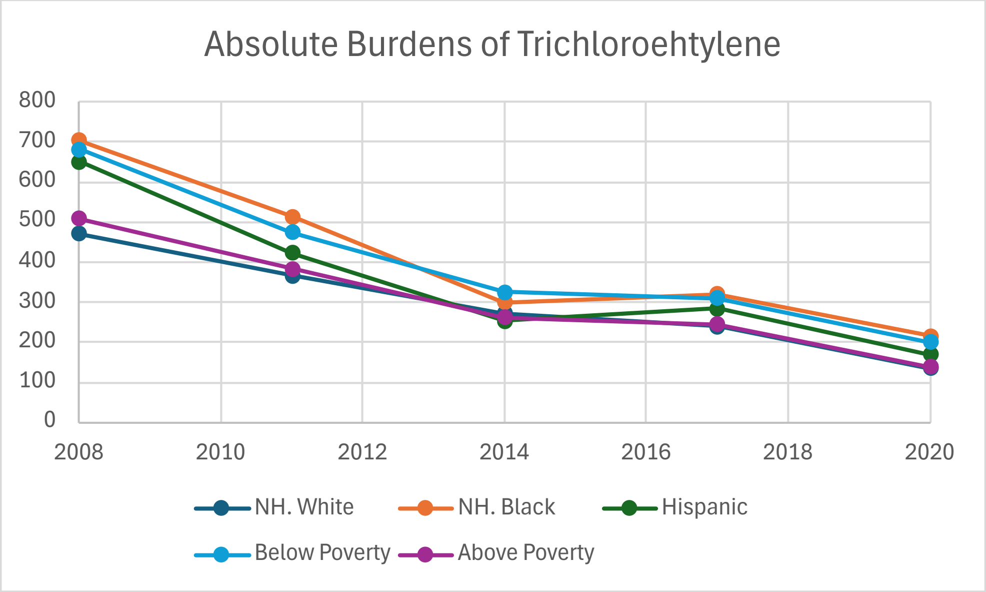

Figure B48. Absolute burdens of trichloroethylene from 2008 to 2020 by population subgroup.

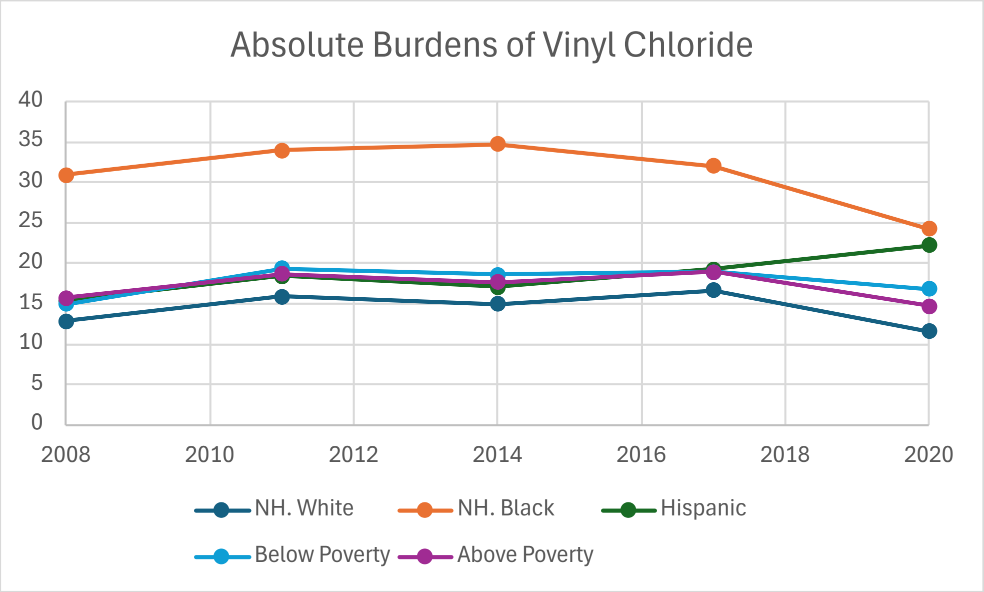

Figure B49. Absolute burdens of vinyl chloride from 2008 to 2020 by population subgroup.

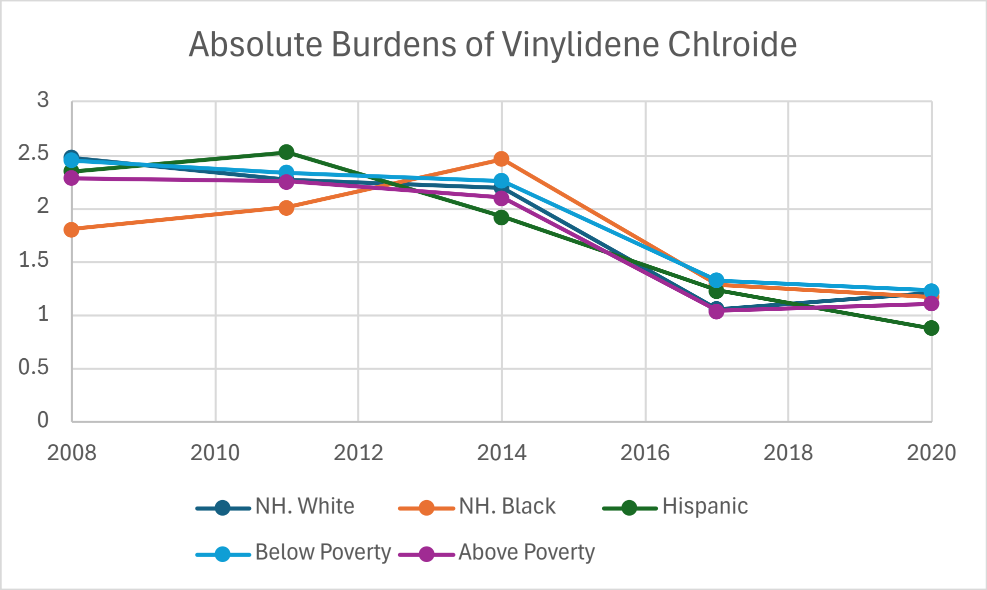

Figure B50. Absolute burdens of vinylidene chloride from 2008 to 2020 by population subgroup.

Appendix C

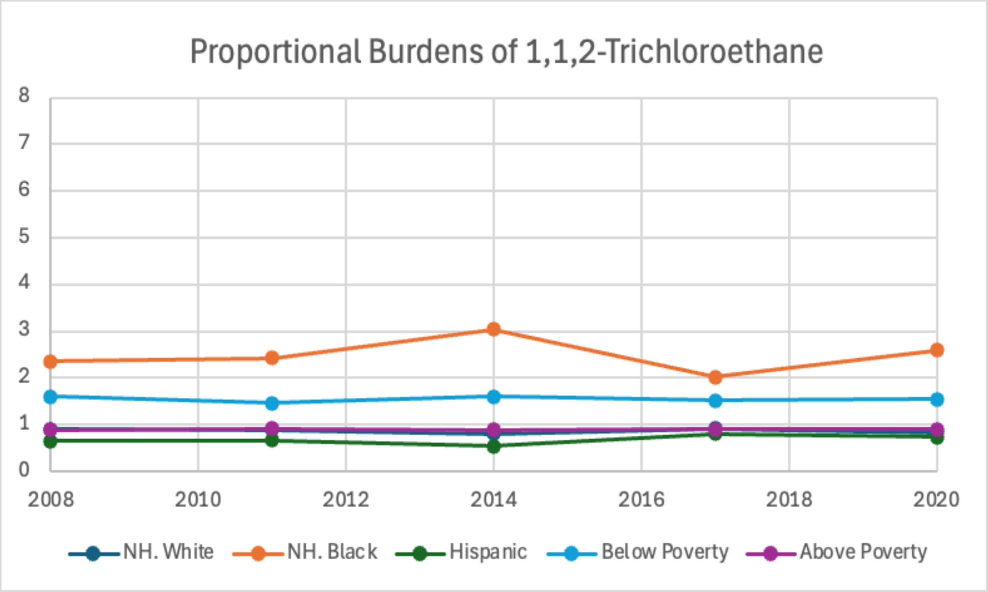

Figure C1. Absolute burdens of 1,1,2-trichloroethane from 2008 to 2020 by population subgroup.

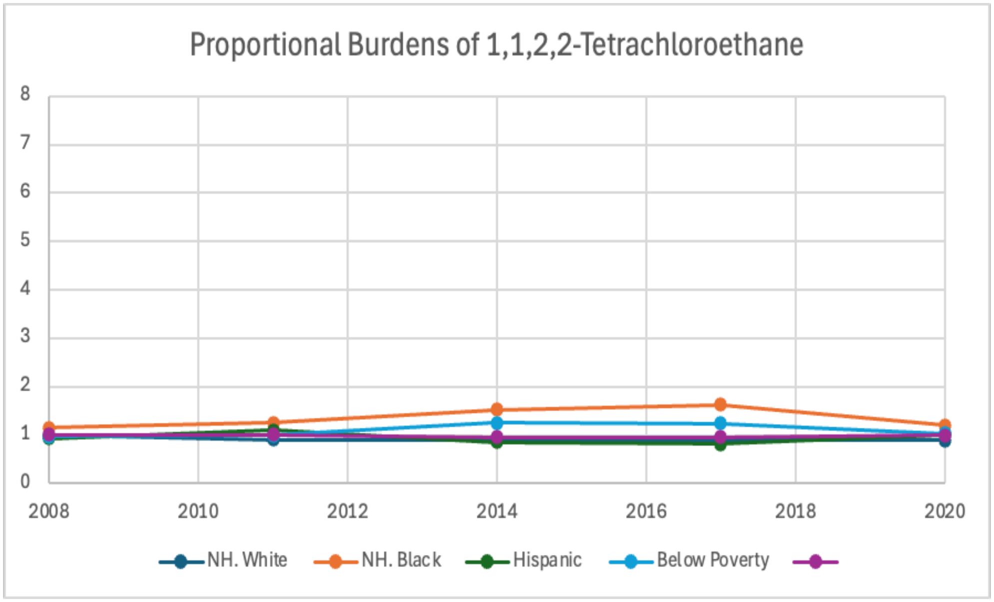

Figure C2. Absolute burdens of 1,1,2,2-tetrachloroethane from 2008 to 2020 by population subgroup.

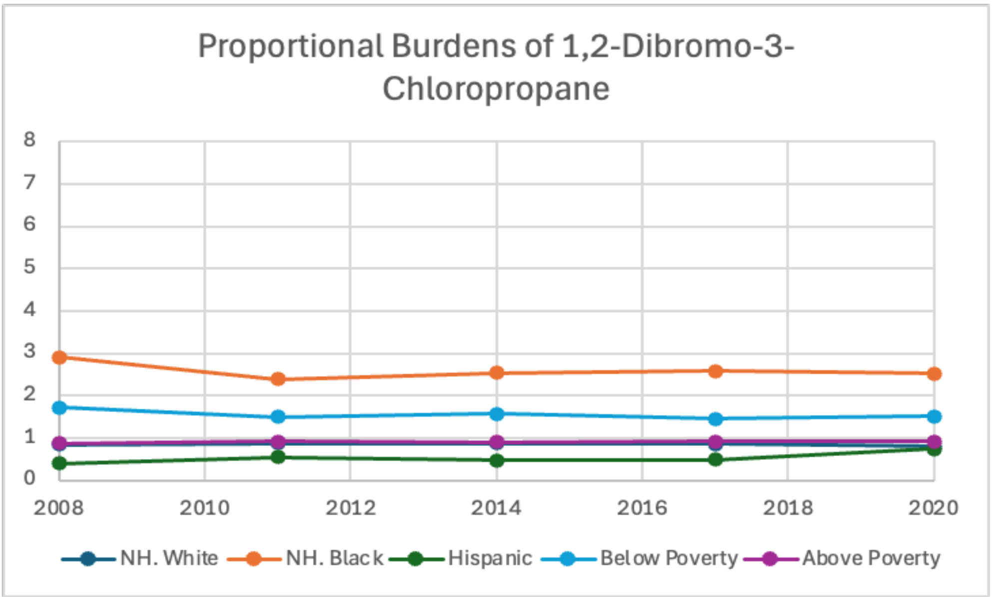

Figure C3. Absolute burdens of 1,2-dibromo-3-chloropropane from 2008 to 2020 by population subgroup.

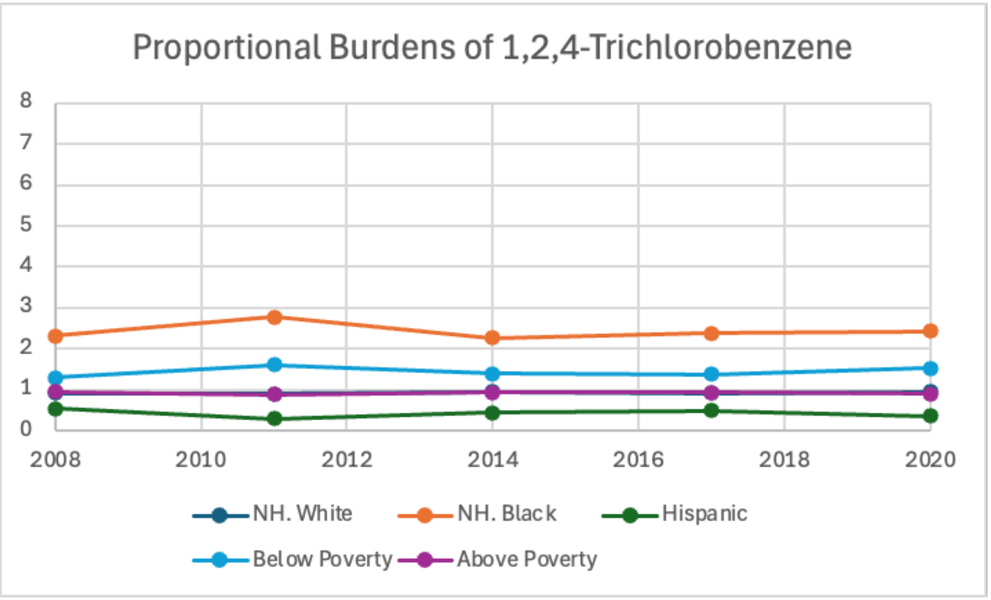

Figure C4. Absolute burdens of 1,2,4-trichlorobenzene from 2008 to 2020 by population subgroup.

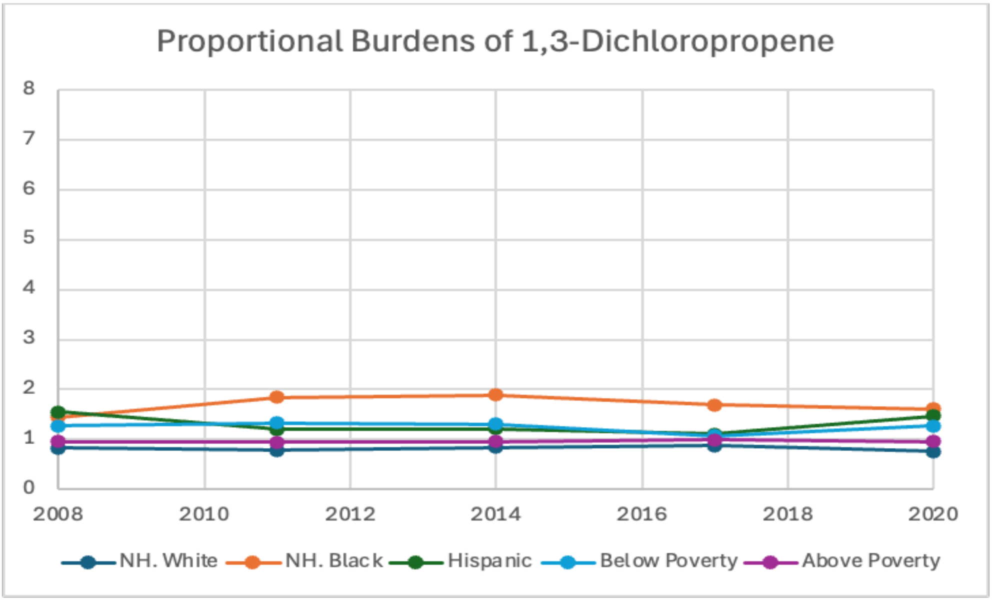

Figure C5. Absolute burdens of 1,3-dichloropropene from 2008 to 2020 by population subgroup.

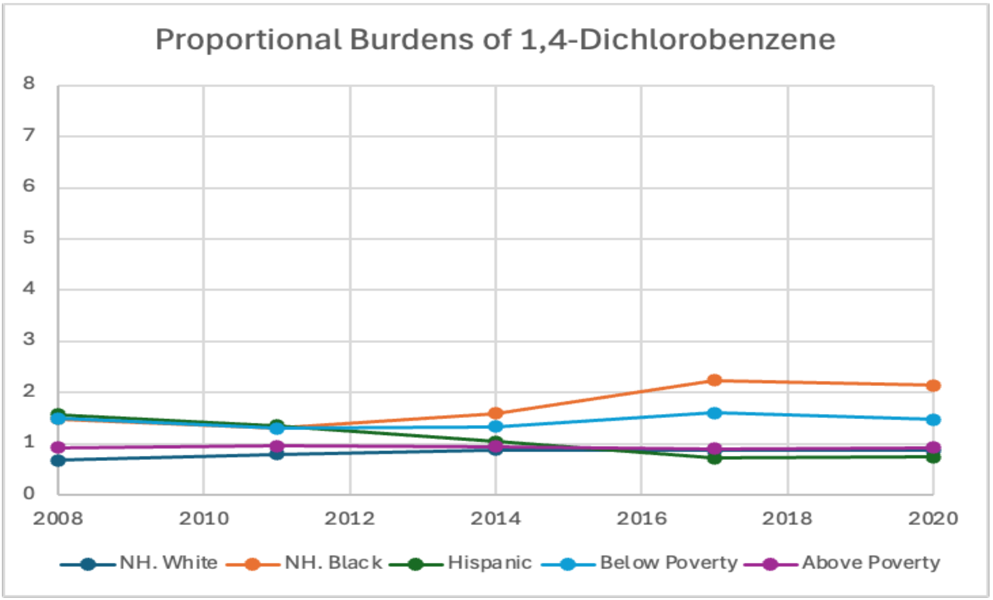

Figure C6. Absolute burdens of 1,4-dichlorobenzene from 2008 to 2020 by population subgroup.

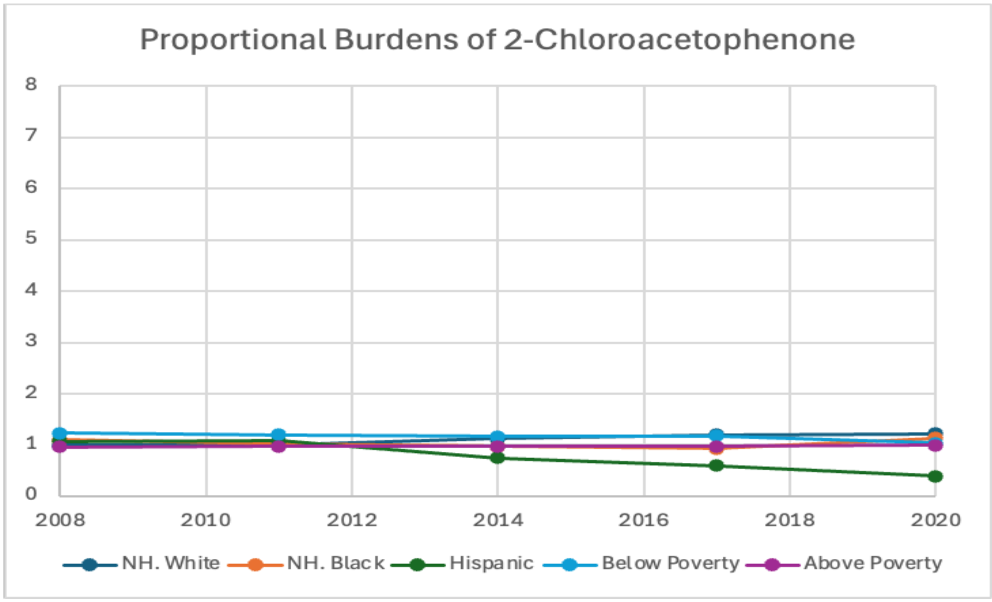

Figure C7. Absolute burdens of 2-chloroacetophenone from 2008 to 2020 by population subgroup.

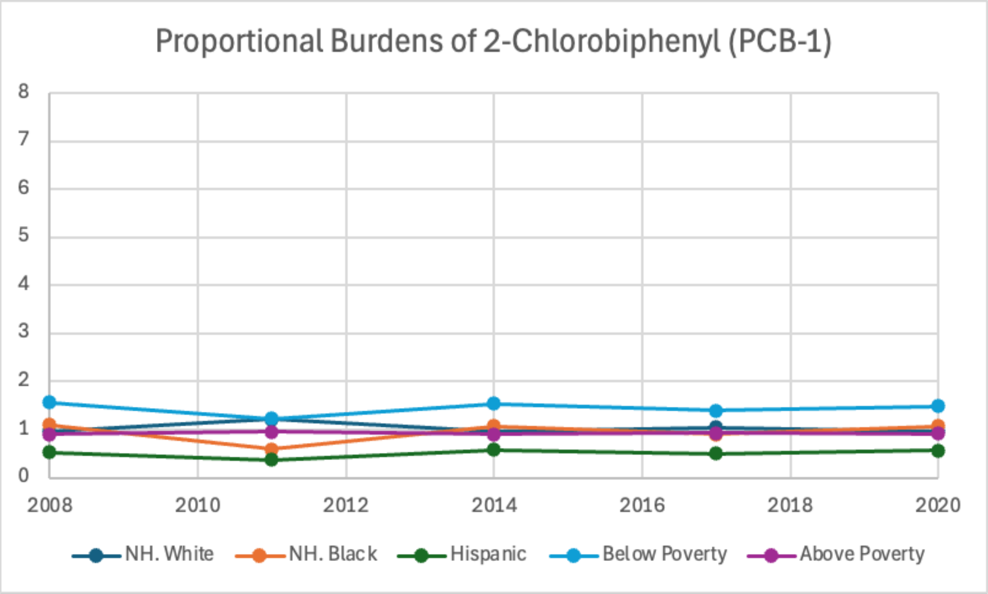

Figure C8. Absolute burdens of 2-chlorobiphenyl (PCB-1) from 2008 to 2020 by population subgroup.

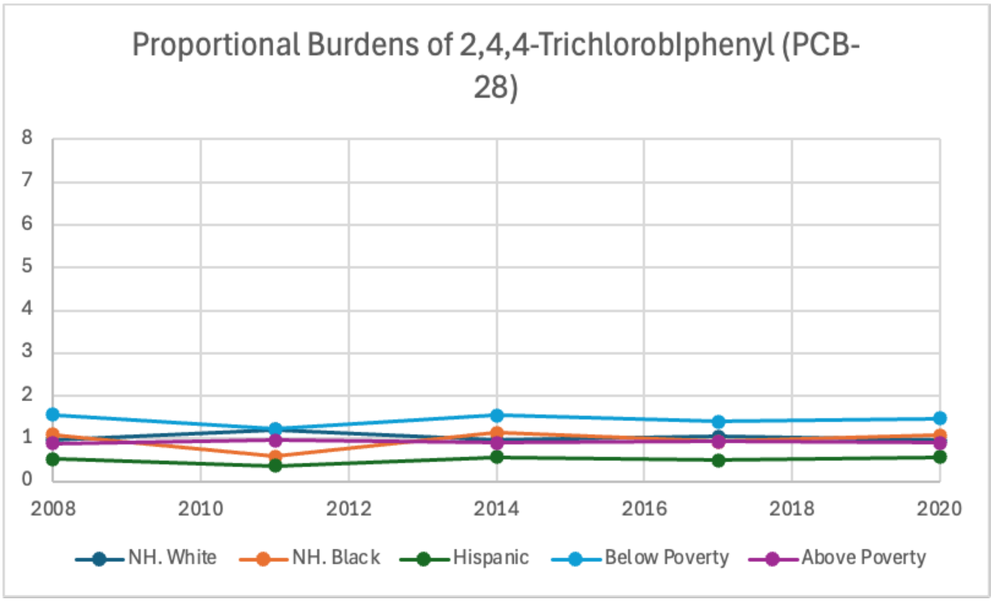

Figure C9. Absolute burdens of 2,4,4-trichlorobiphenyl (PCB-28) from 2008 to 2020 by population subgroup.

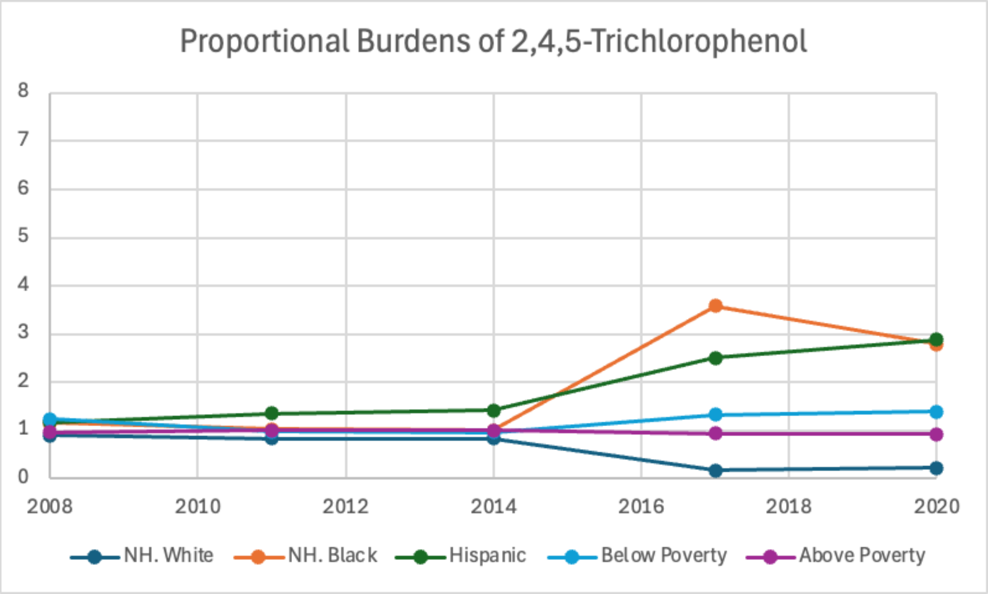

Figure C10. Absolute burdens of 2,4,5-trichlorophenol from 2008 to 2020 by population subgroup.

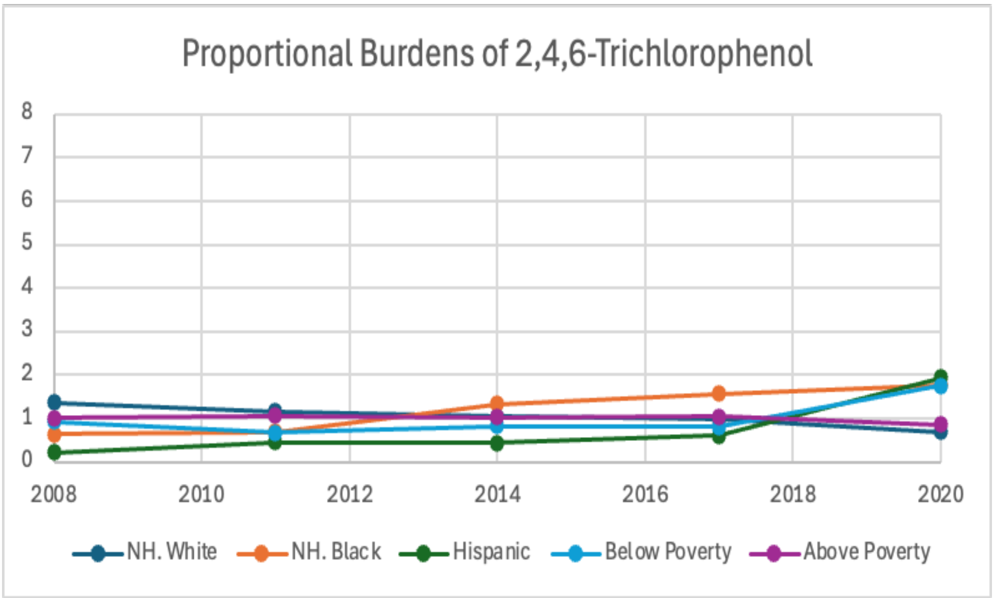

Figure C11. Absolute burdens of 2,4,6-trichlorophenol from 2008 to 2020 by population subgroup.

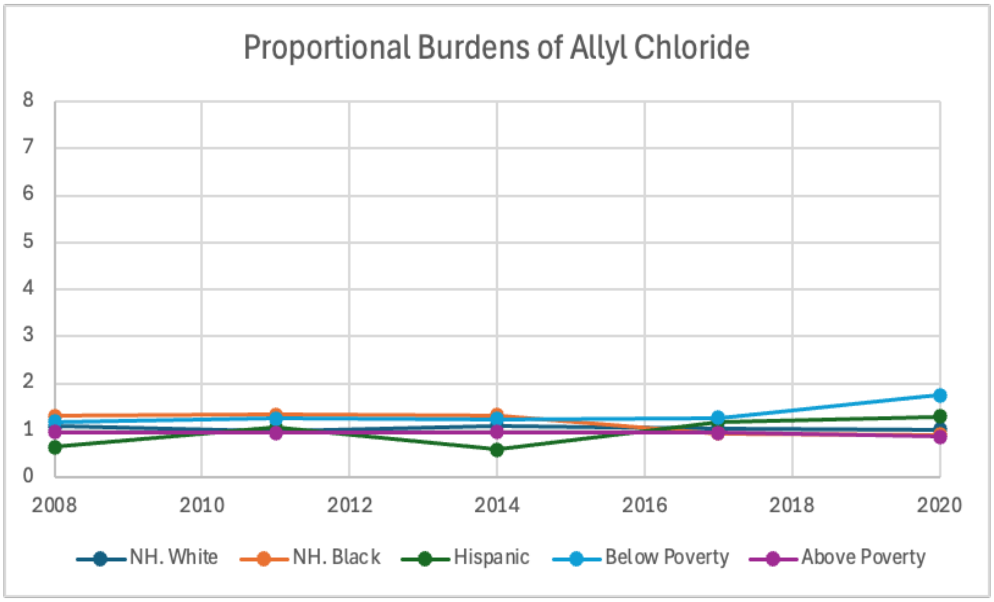

Figure C12. Absolute burdens of allyl chloride from 2008 to 2020 by population subgroup.

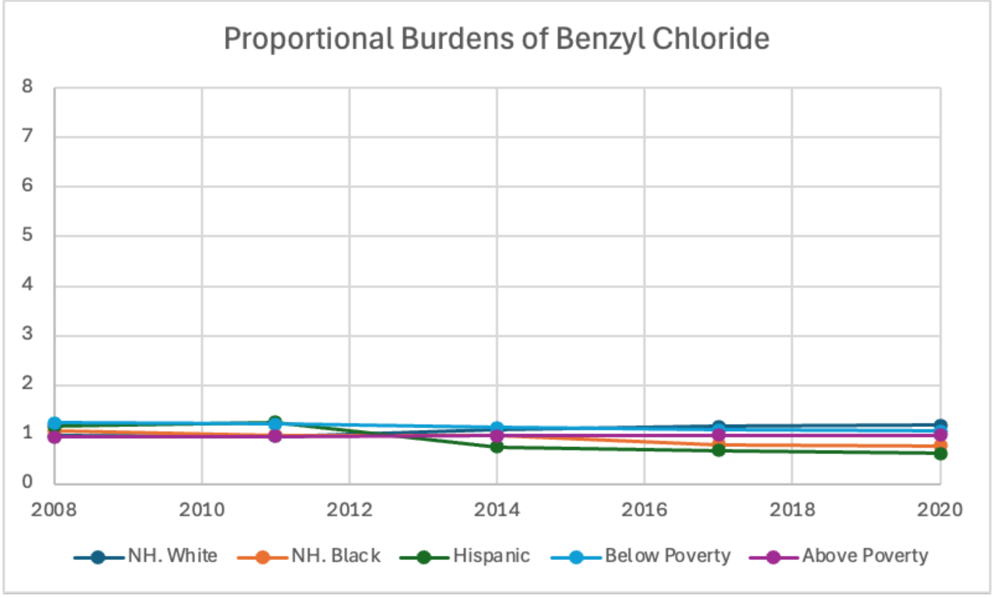

Figure C13. Absolute burdens of benzyl chloride from 2008 to 2020 by population subgroup.

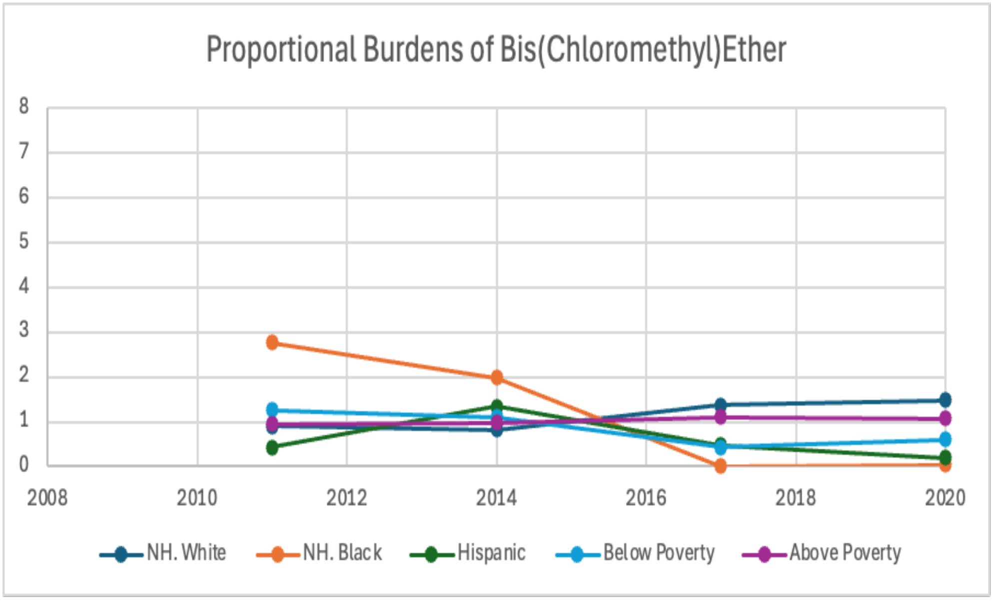

Figure C14. Absolute burdens of bis(chloromethyl)ether from 2008 to 2020 by population subgroup. Note. 2008 was not available within NEI data.

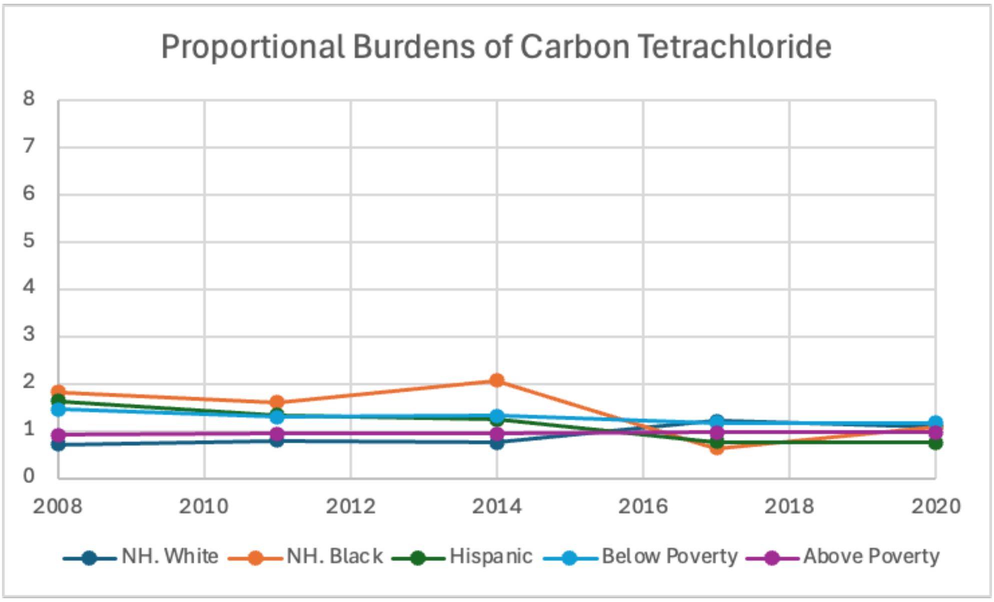

Figure C15. Absolute burdens of carbon tetrachloride from 2008 to 2020 by population subgroup.

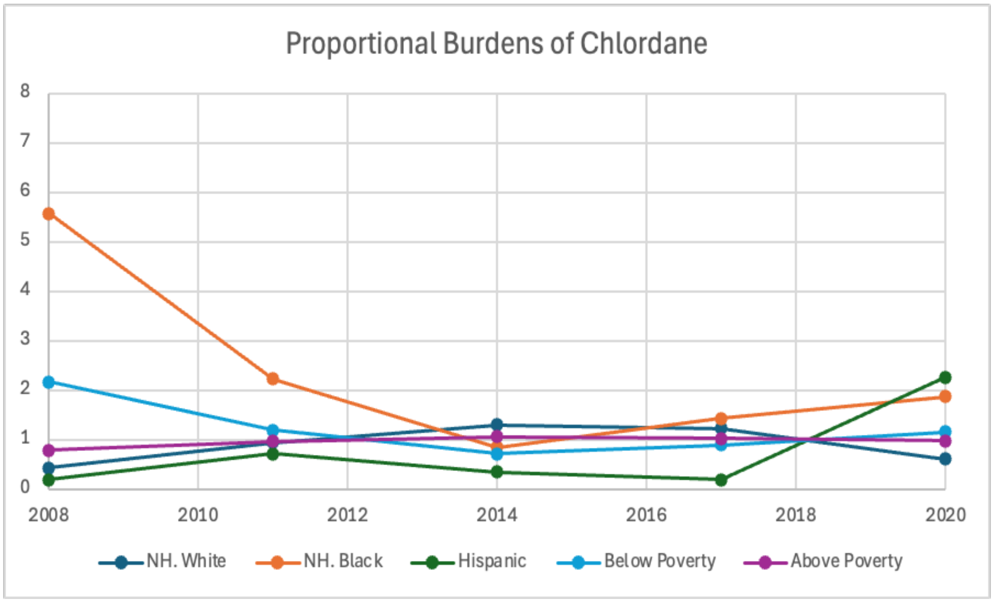

Figure C16. Absolute burdens of chlordane from 2008 to 2020 by population subgroup.

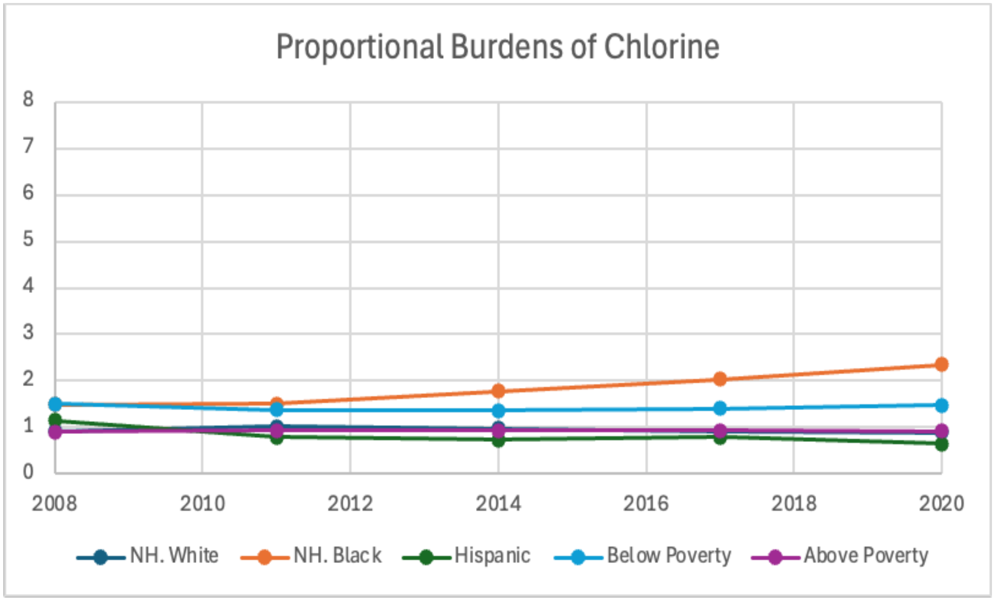

Figure C17. Absolute burdens of chlorine from 2008 to 2020 by population subgroup.

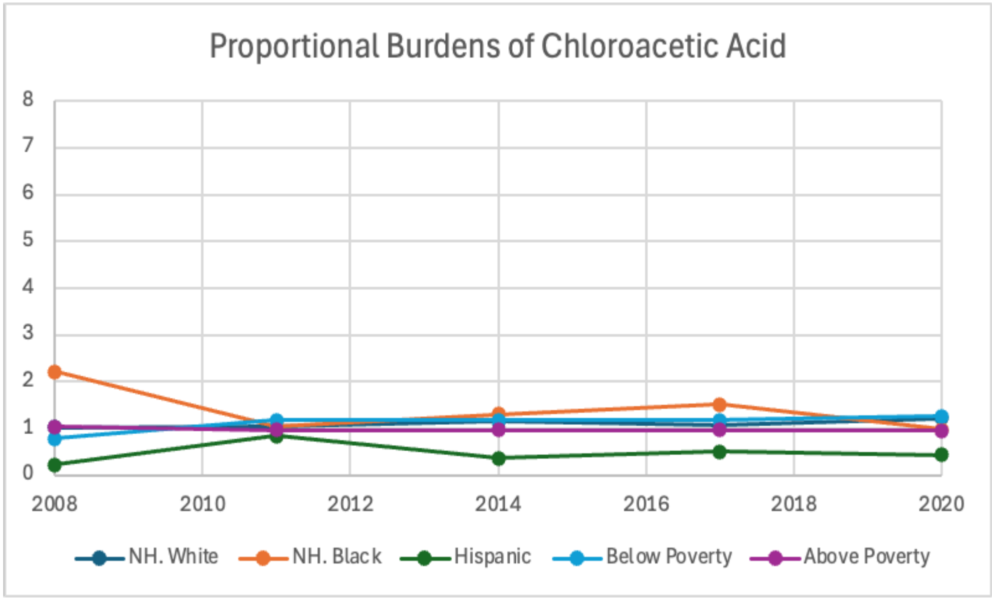

Figure C18. Absolute burdens of chloroacetic acid from 2008 to 2020 by population subgroup.

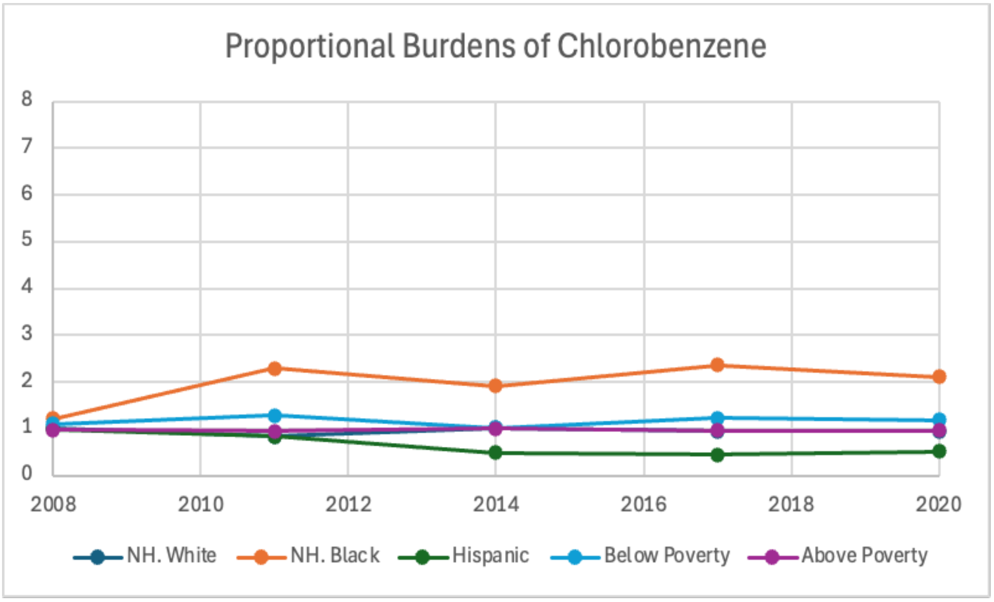

Figure C19. Absolute burdens of chlorobenzene from 2008 to 2020 by population subgroup.

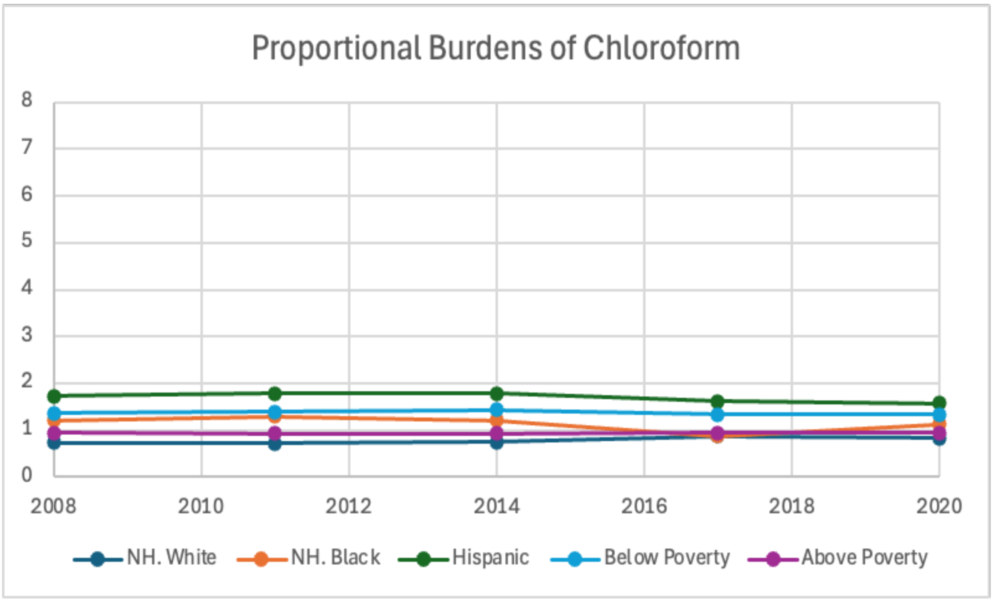

Figure C20. Absolute burdens of chloroform from 2008 to 2020 by population subgroup.

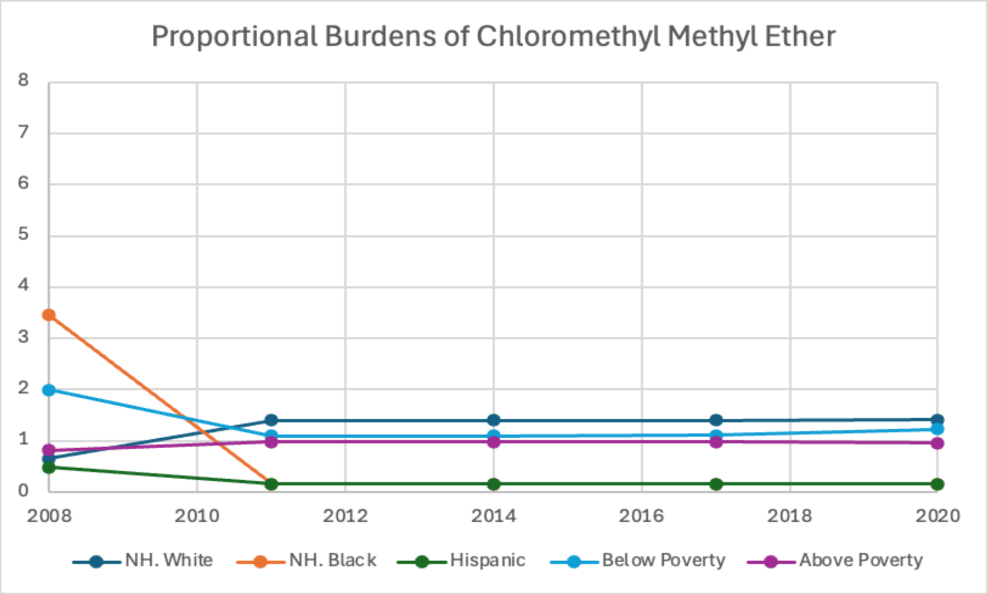

Figure C21. Absolute burdens of chloromethyl methyl ether from 2008 to 2020 by population subgroup.

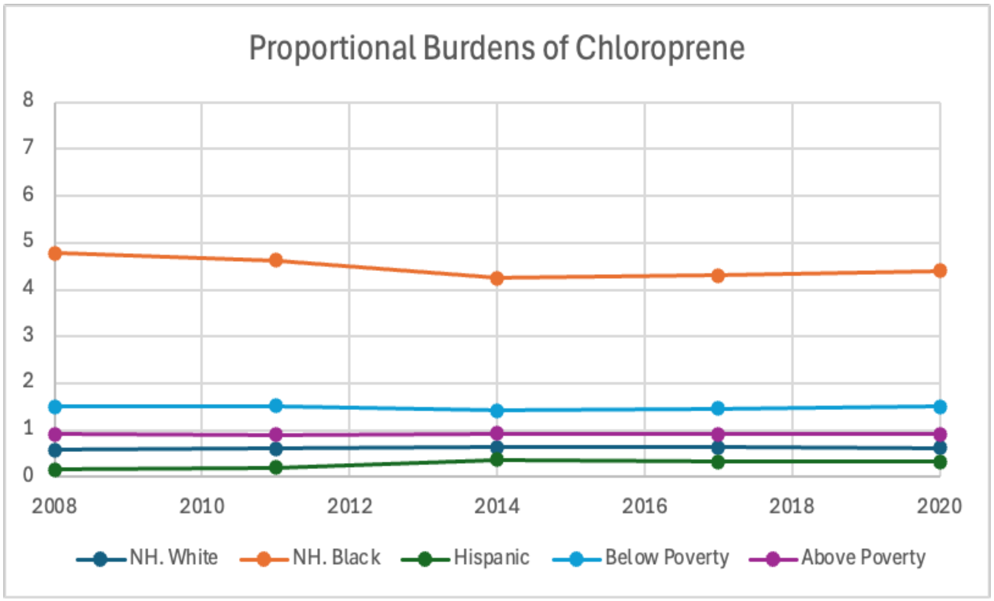

Figure C22. Absolute burdens of chloroprene from 2008 to 2020 by population subgroup.

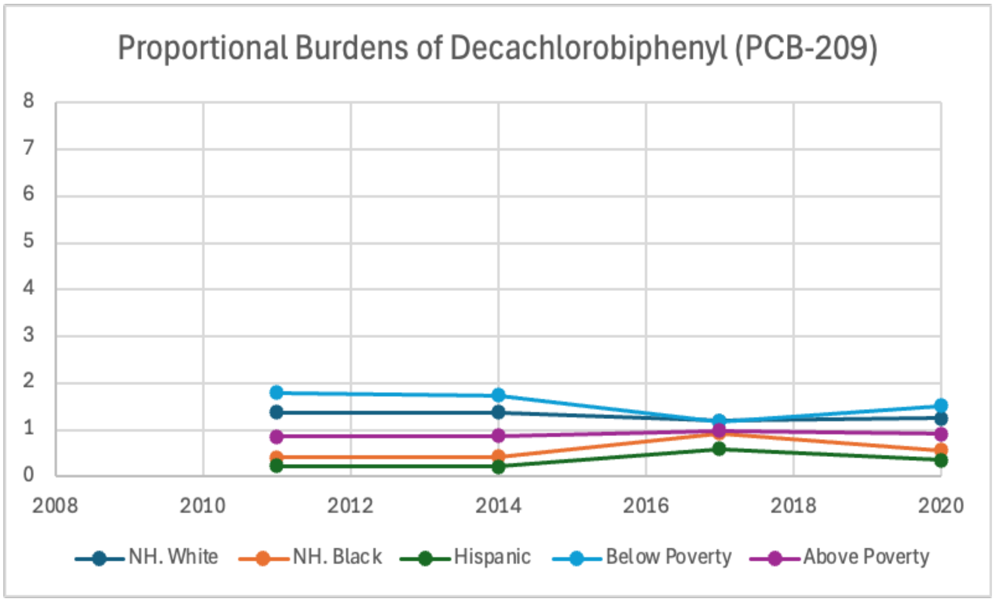

Figure C23. Absolute burdens of decachlorobiphenyl (PCB-209) from 2008 to 2020 by population subgroup. Note. 2008 was not available within NEI data.

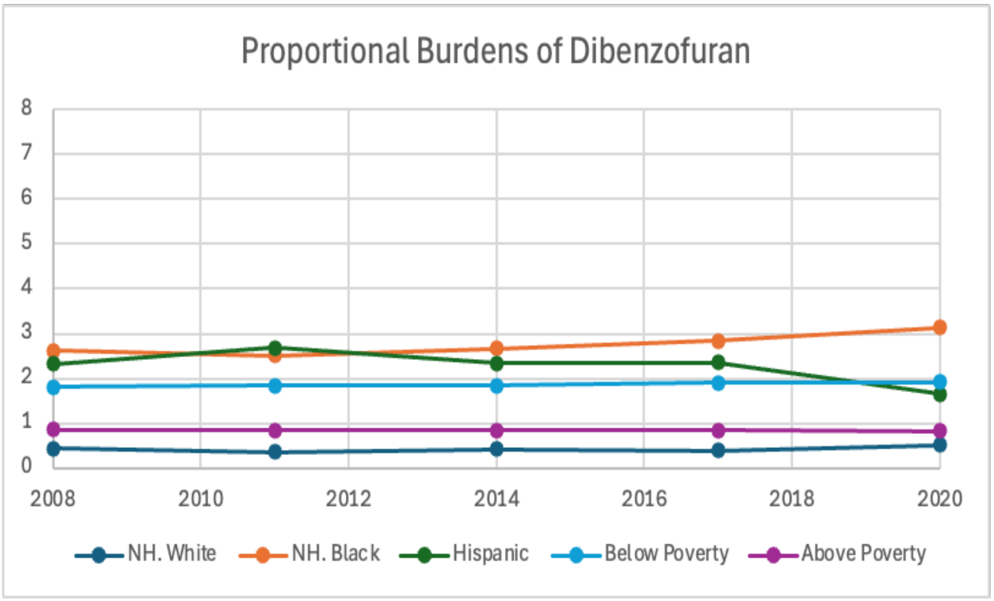

Figure C24. Absolute burdens of dibenzofuran from 2008 to 2020 by population subgroup.

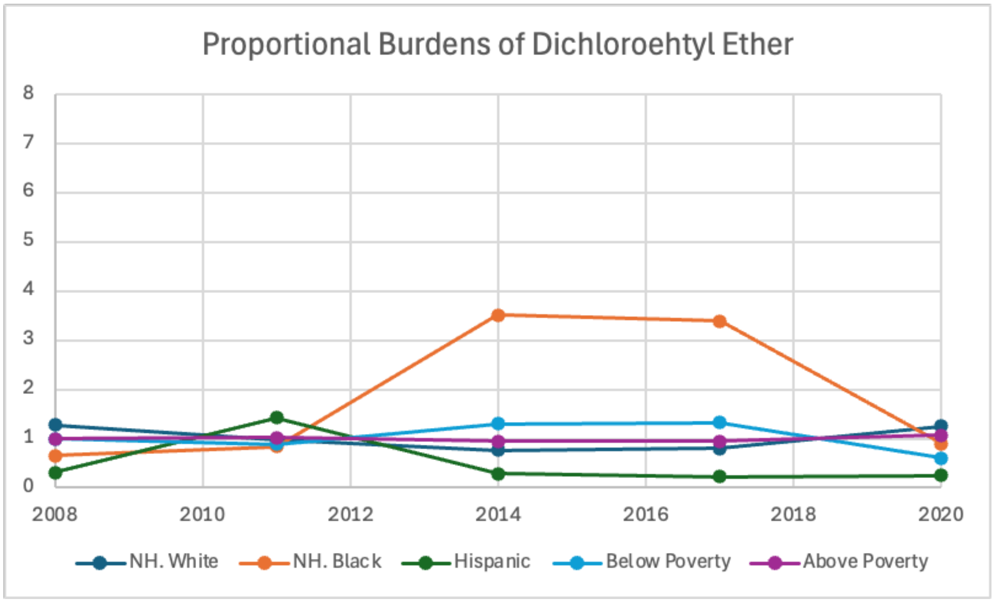

Figure C25. Absolute burdens of dichloroethyl ether from 2008 to 2020 by population subgroup.

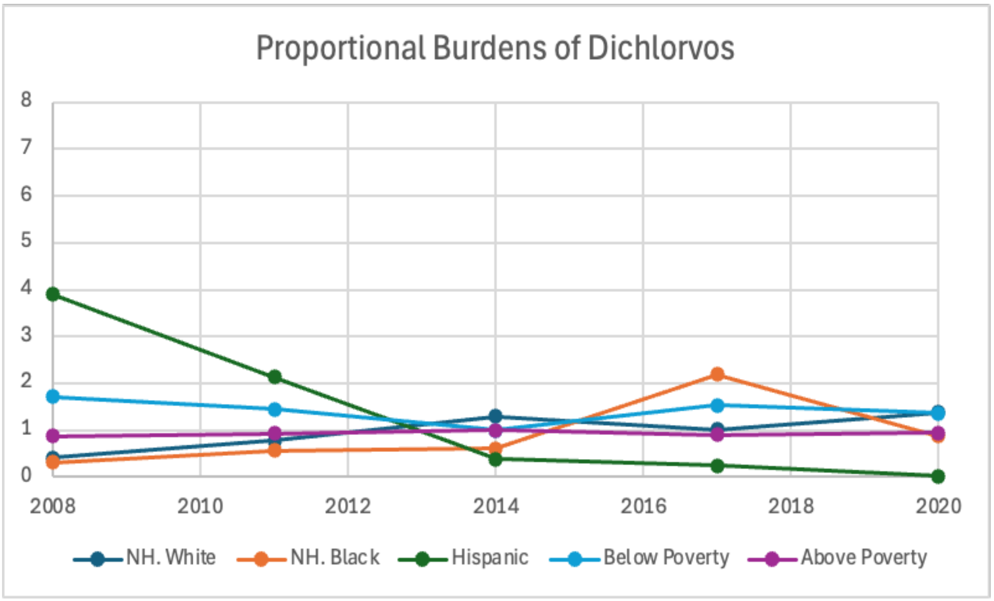

Figure C26. Absolute burdens of dichlorvos from 2008 to 2020 by population subgroup.

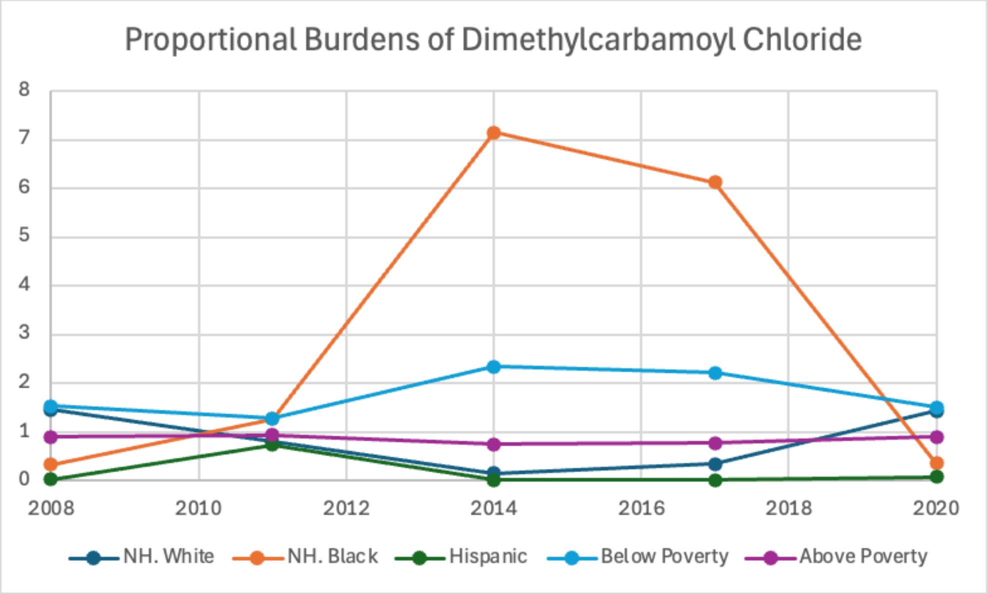

Figure C27. Absolute burdens of dimethylcarbamoyl chloride from 2008 to 2020 by population subgroup.

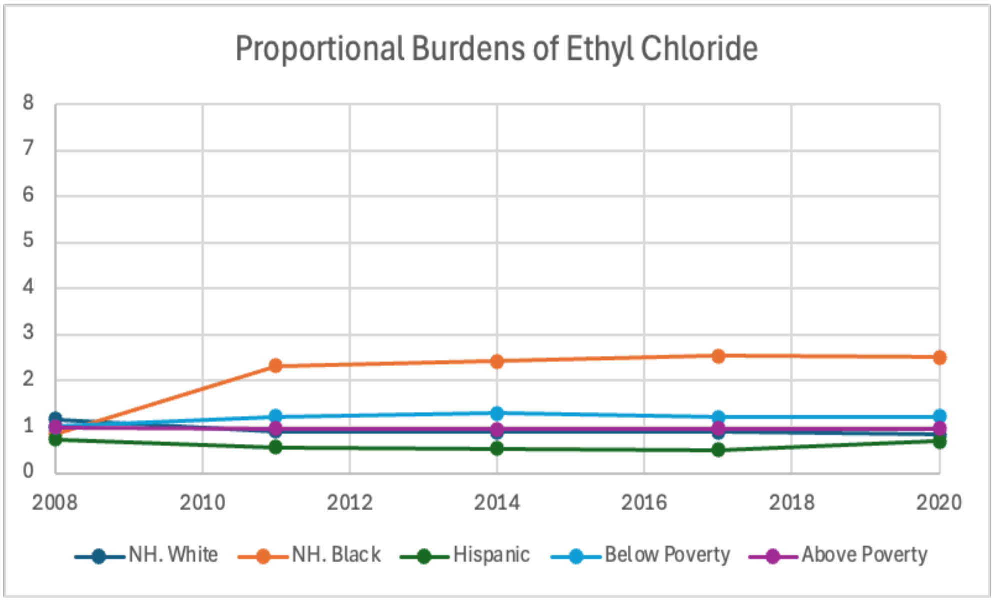

Figure C28. Absolute burdens of ethyl chloride from 2008 to 2020 by population subgroup.

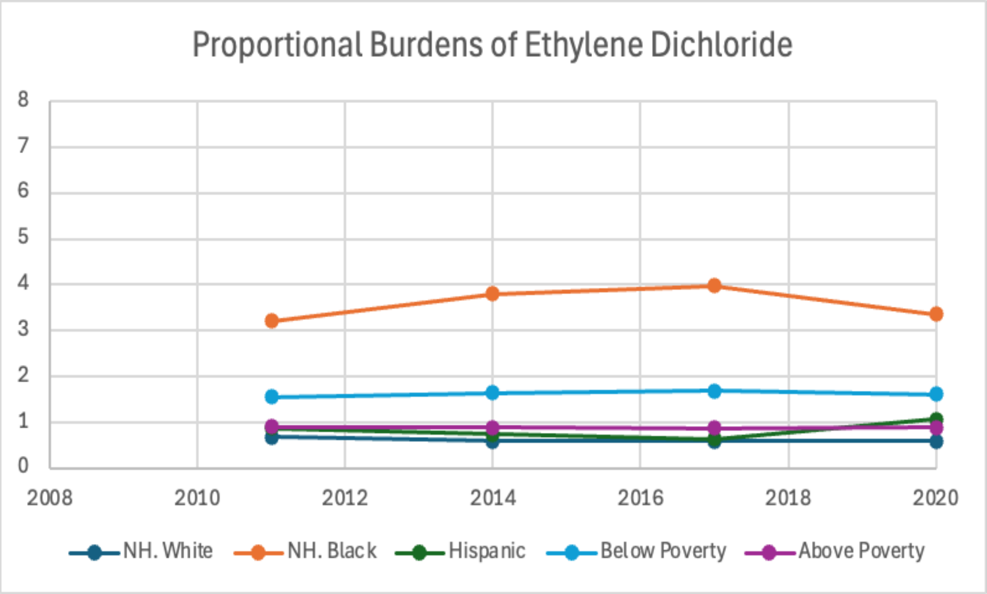

Figure C29. Absolute burdens of ethylene dichloride from 2008 to 2020 by population subgroup. Note. 2008 was not available with NEI data.

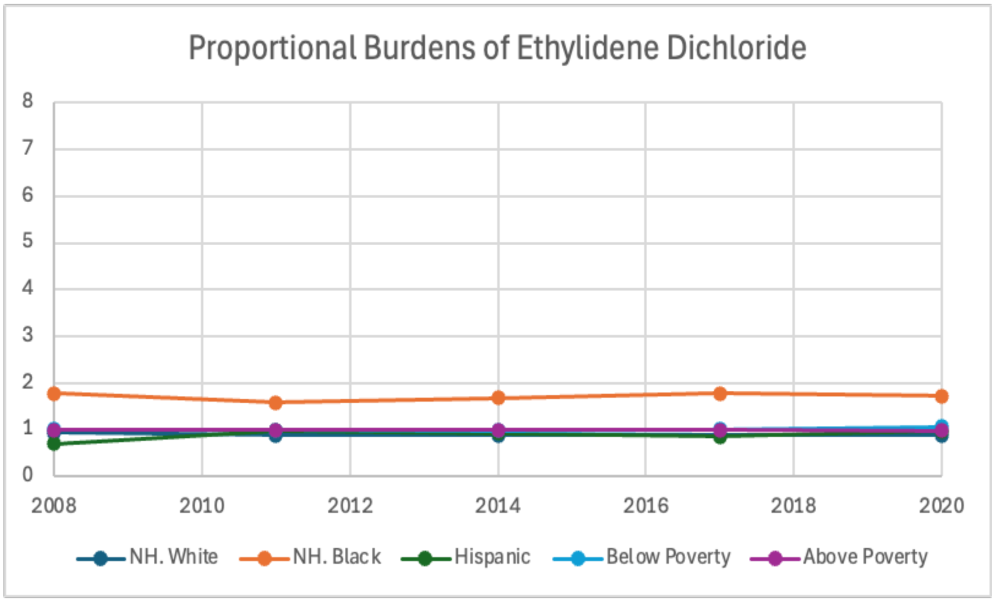

Figure C30. Absolute burdens of ethylidene dichloride from 2008 to 2020 by population subgroup.

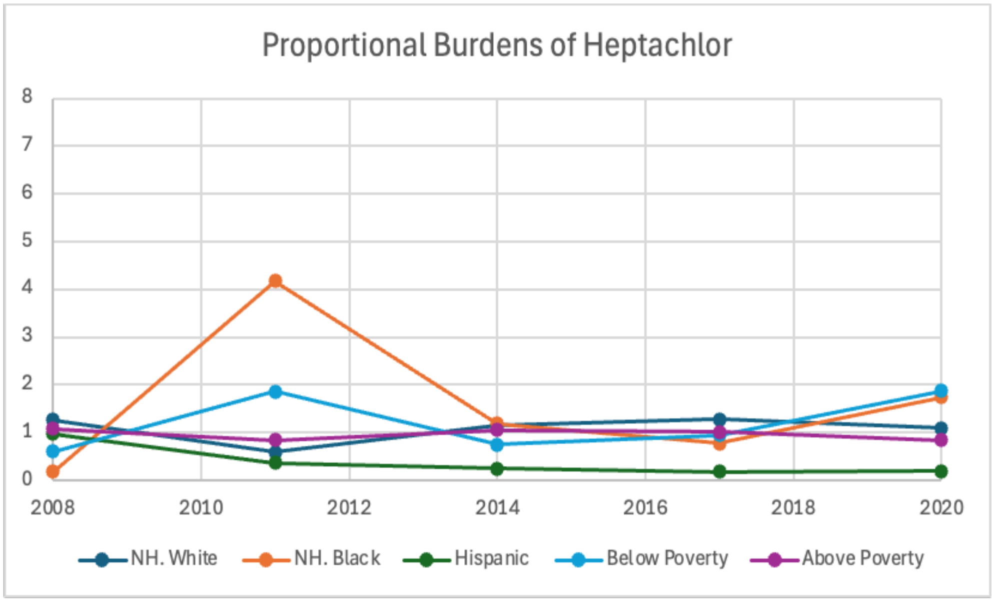

Figure C31. Absolute burdens of heptachlor from 2008 to 2020 by population subgroup.

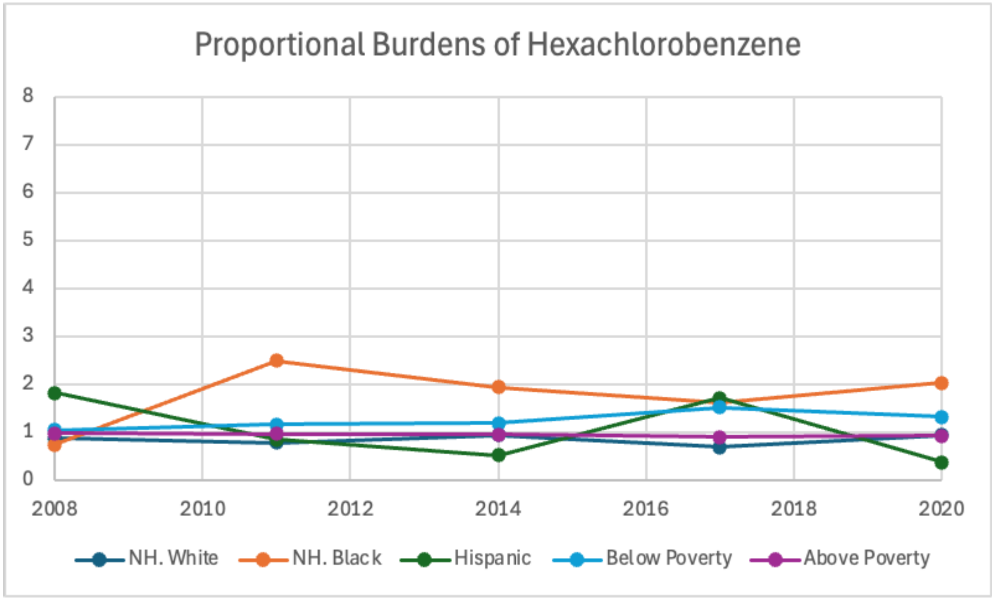

Figure C32. Absolute burdens of hexachlorobenzene from 2008 to 2020 by population subgroup.

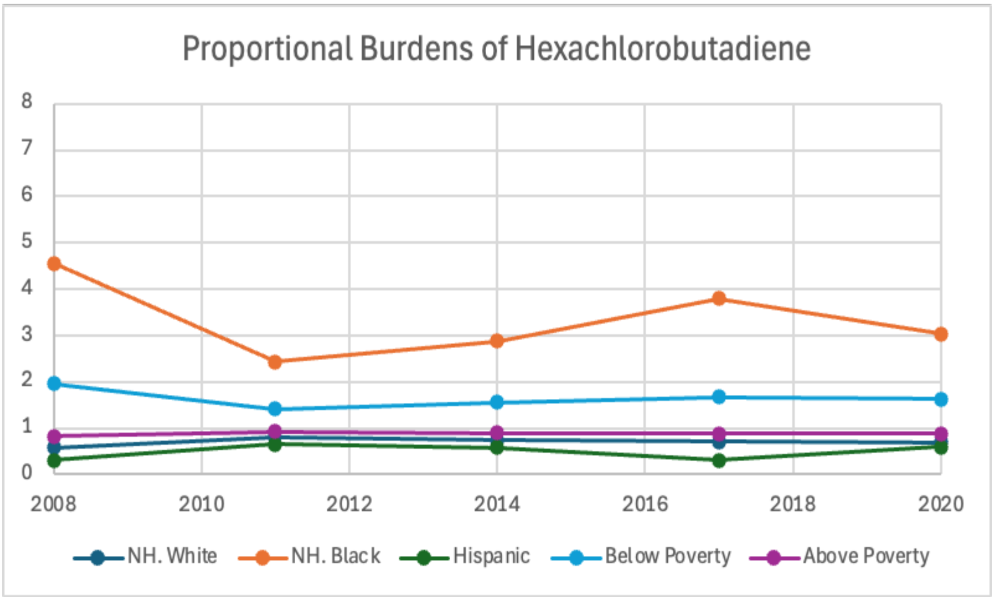

Figure C33. Absolute burdens of hexachlorobutadiene from 2008 to 2020 by population subgroup.

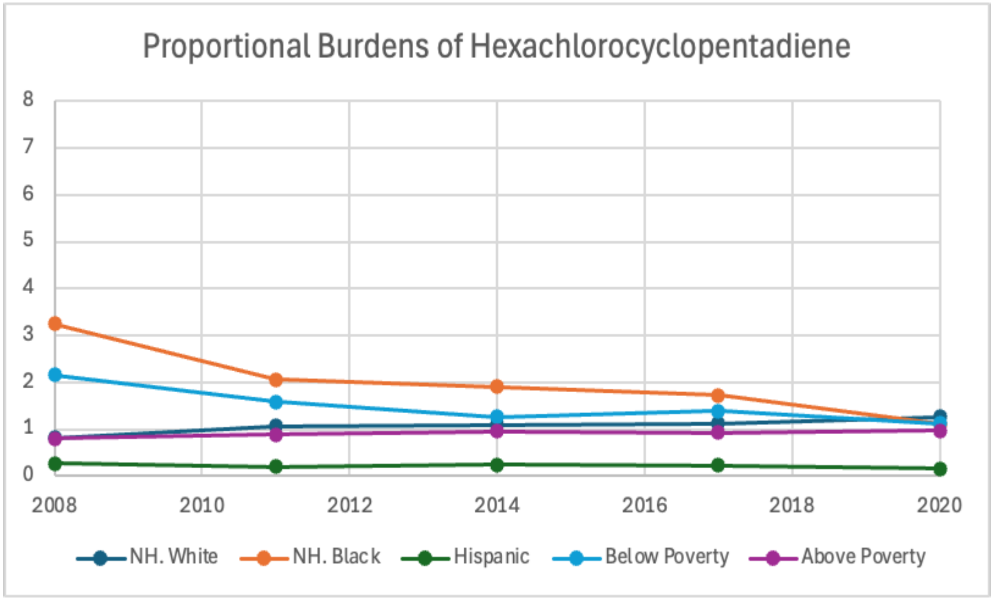

Figure C34. Absolute burdens of hexachlorocyclopentadiene from 2008 to 2020 by population subgroup.

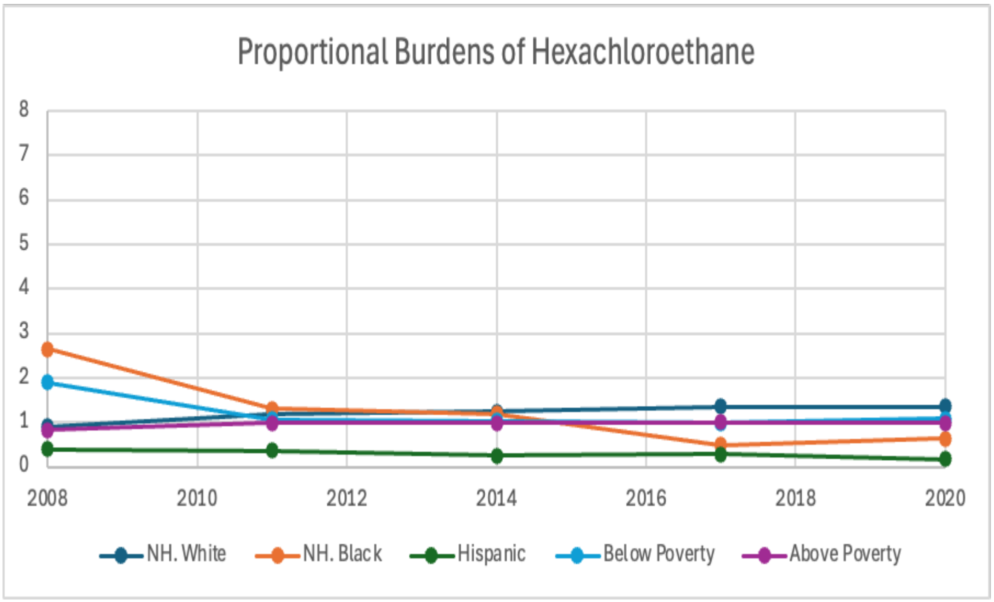

Figure C35. Absolute burdens of hexachloroethane from 2008 to 2020 by population subgroup.

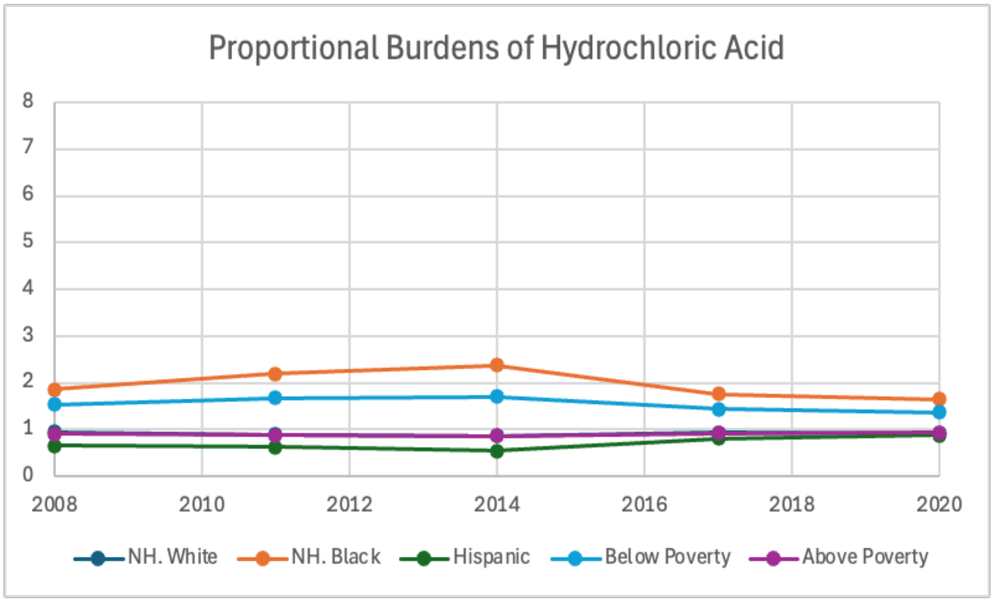

Figure C36. Absolute burdens of hydrochloric acid from 2008 to 2020 by population subgroup.

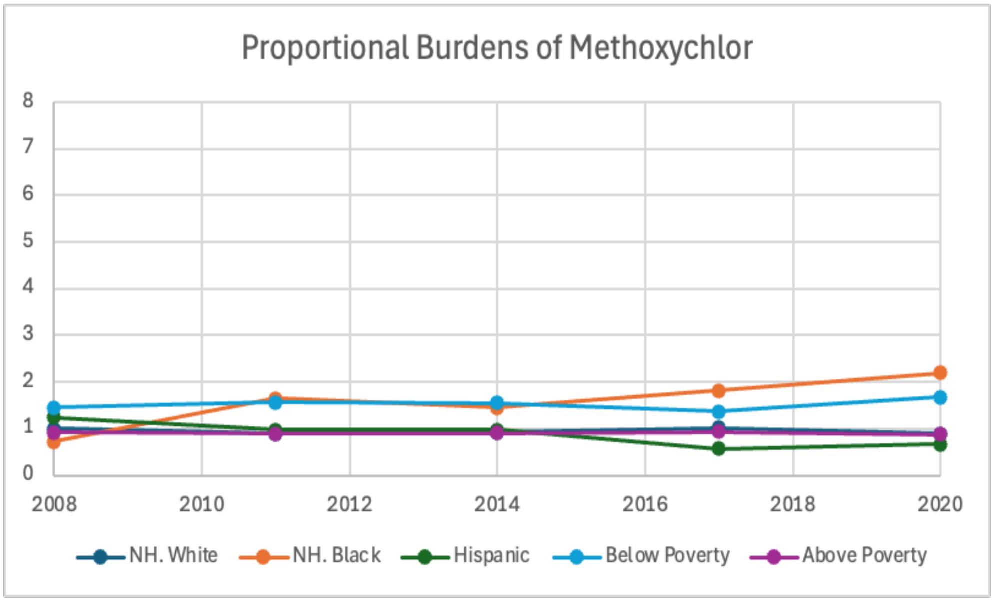

Figure C37. Absolute burdens of methoxychlor from 2008 to 2020 by population subgroup.

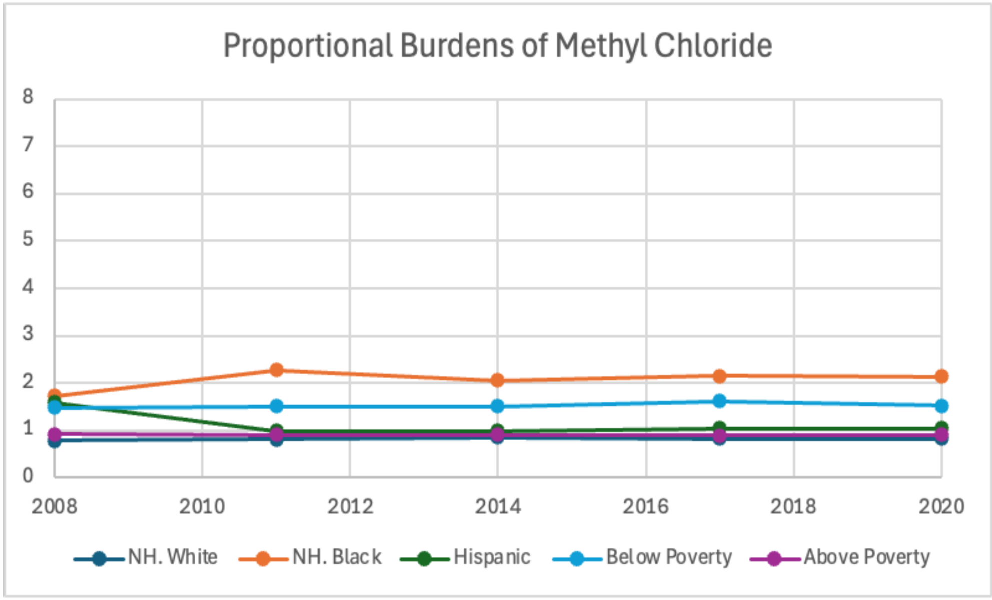

Figure C38. Absolute burdens of methyl chloride from 2008 to 2020 by population subgroup.

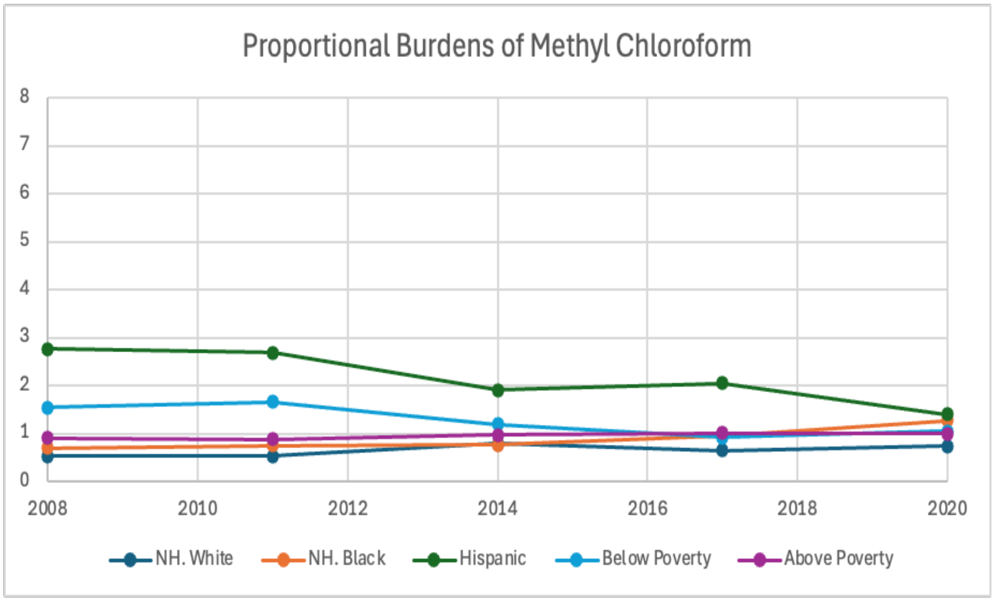

Figure C39. Absolute burdens of methyl chloroform from 2008 to 2020 by population subgroup.

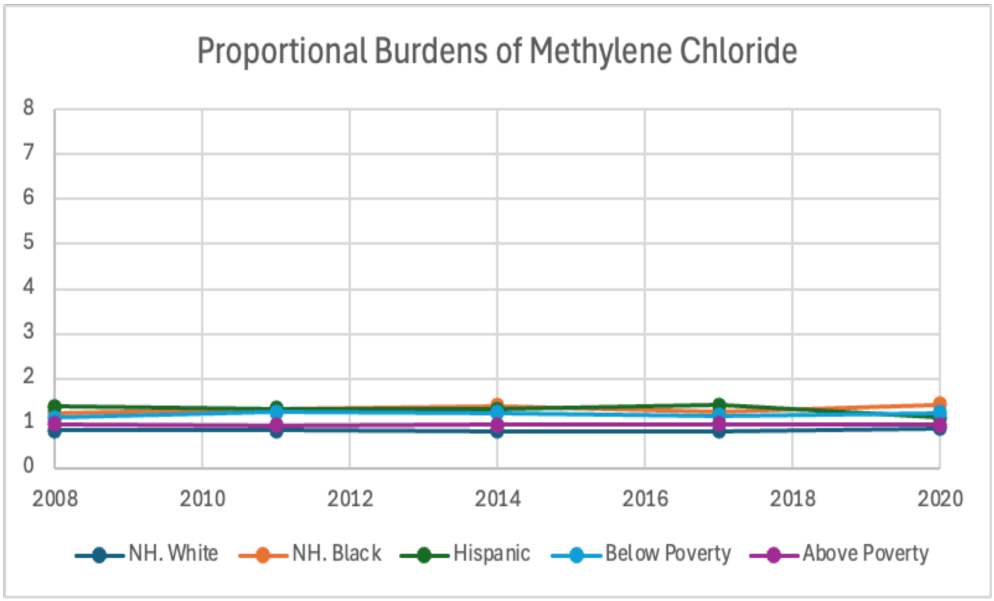

Figure C40. Absolute burdens of methylene chloride from 2008 to 2020 by population subgroup.

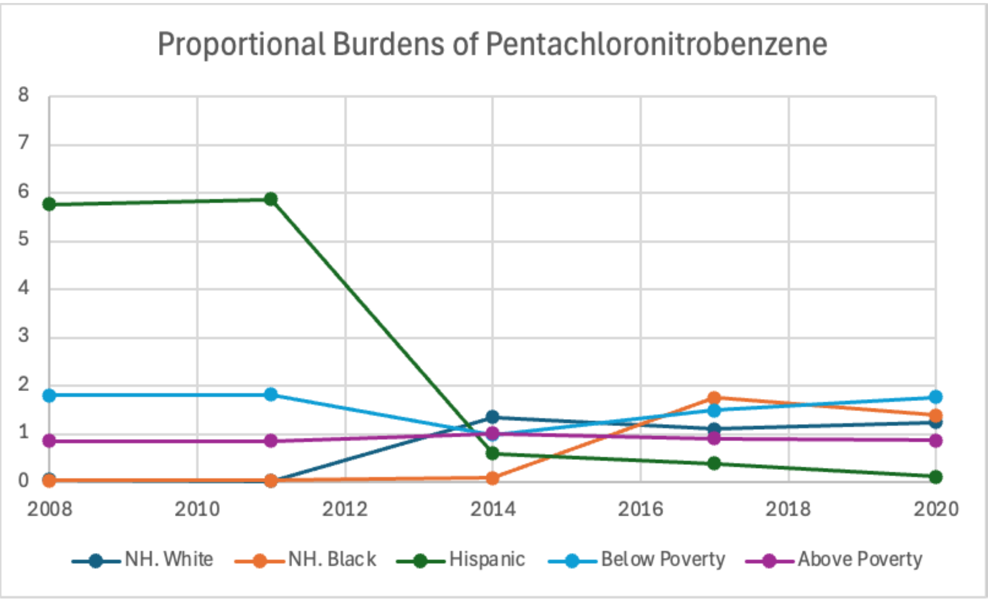

Figure C41. Absolute burdens of pentachloronitrobenzene from 2008 to 2020 by population subgroup.

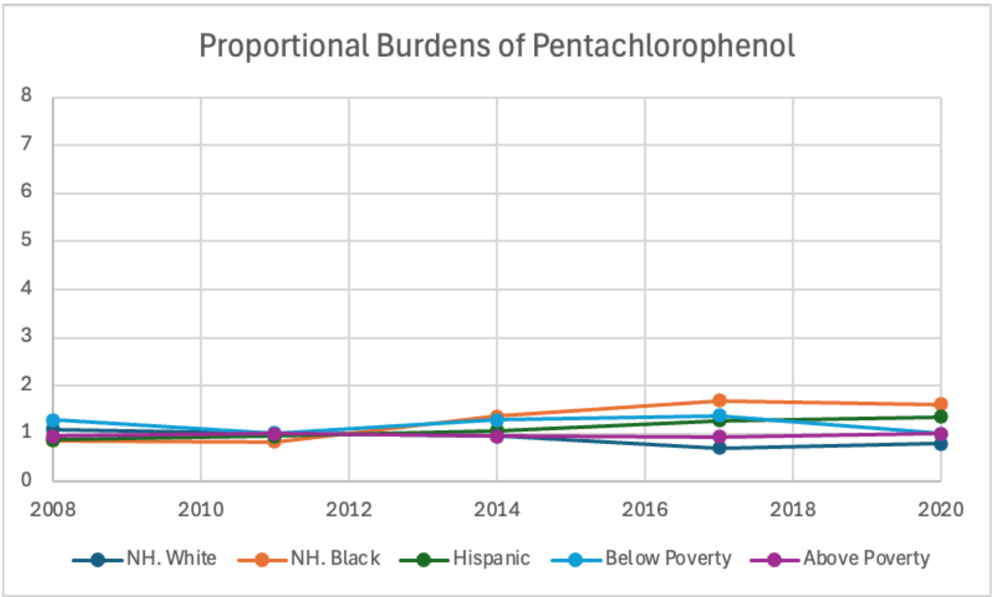

Figure C42. Absolute burdens of pentachlorophenol from 2008 to 2020 by population subgroup.

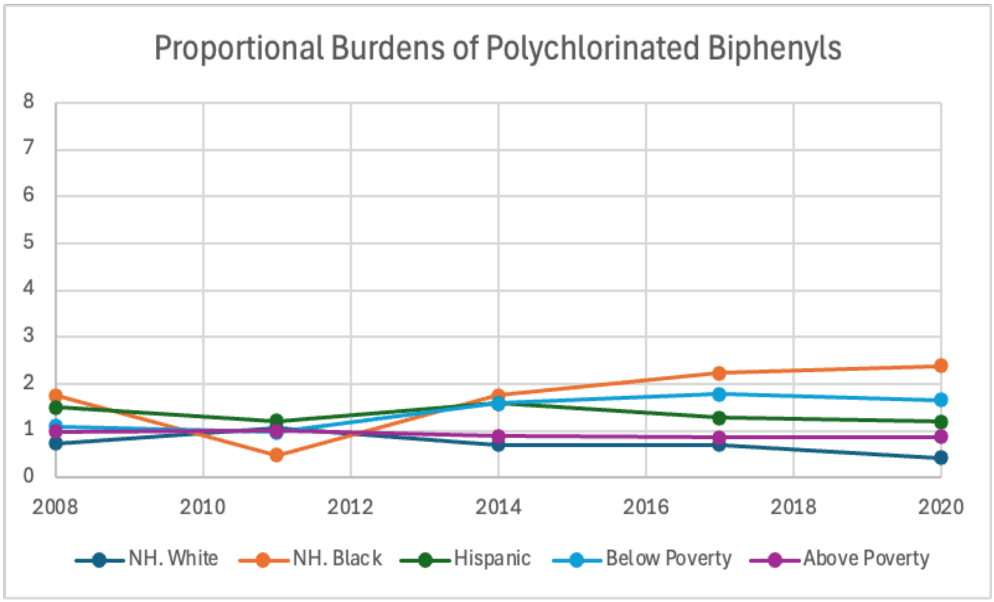

Figure C43. Absolute burdens of polychlorinated biphenyls from 2008 to 2020 by population subgroup.

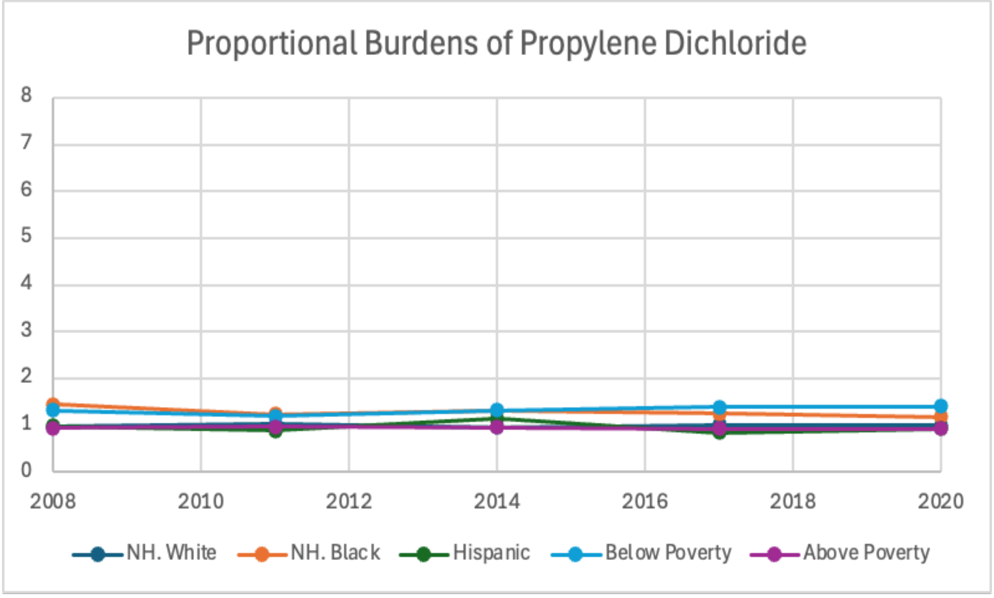

Figure C44. Absolute burdens of propylene dichloride from 2008 to 2020 by population subgroup.

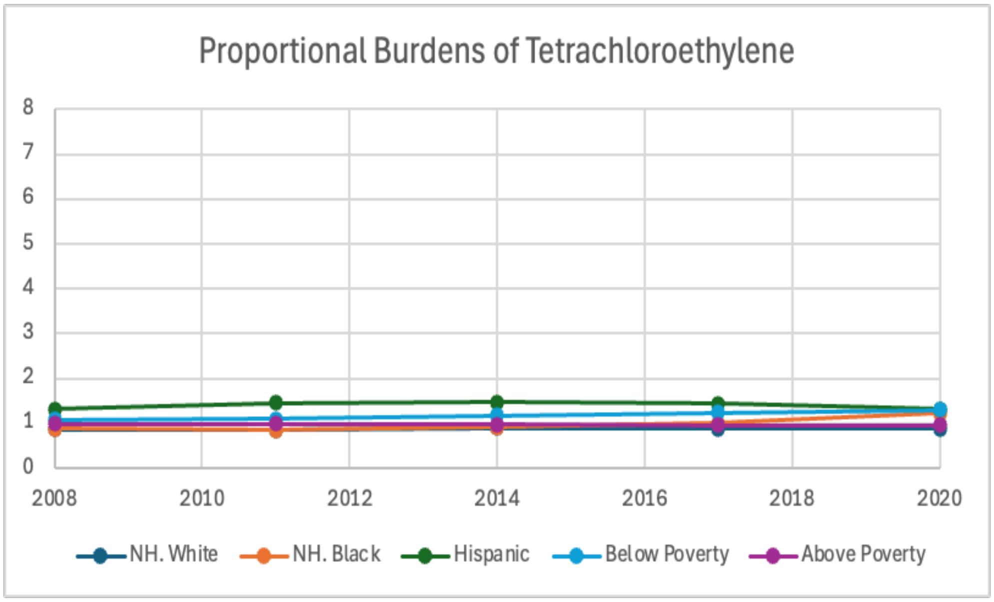

Figure C45. Absolute burdens of tetrachloroethylene from 2008 to 2020 by population subgroup.

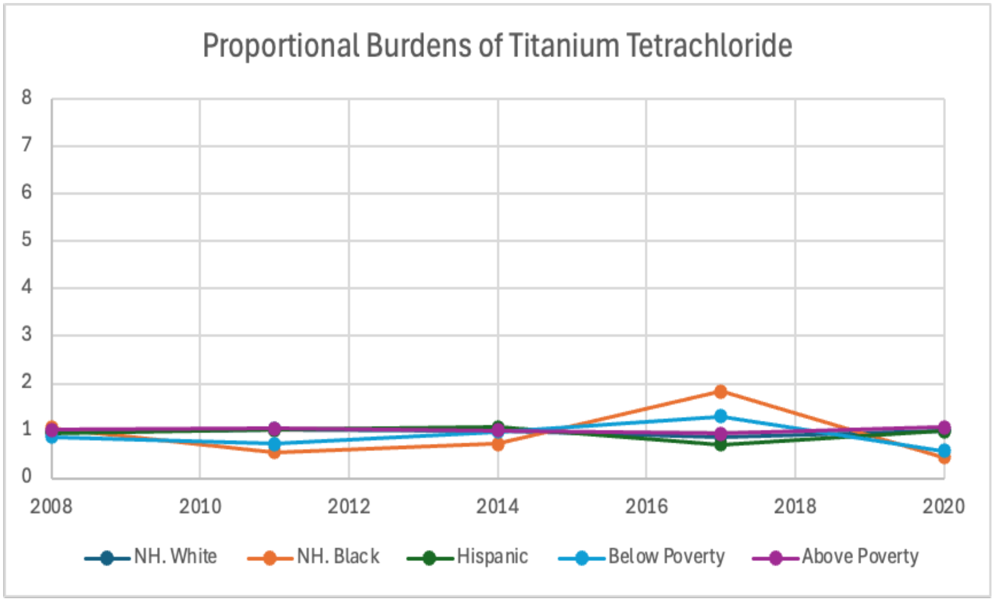

Figure C46. Absolute burdens of titanium tetrachloride from 2008 to 2020 by population subgroup.

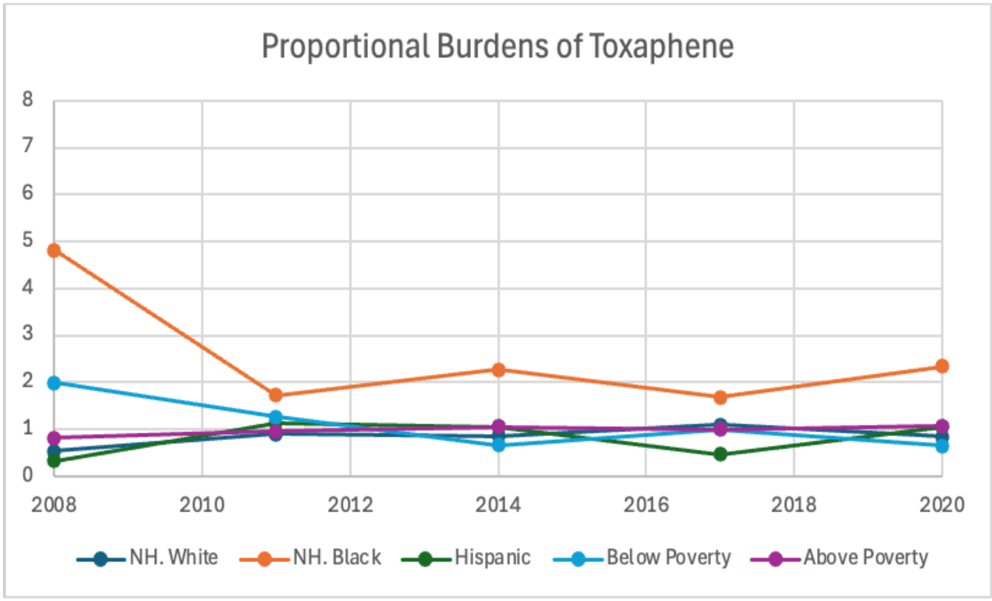

Figure C47. Absolute burdens of toxaphene from 2008 to 2020 by population subgroup.

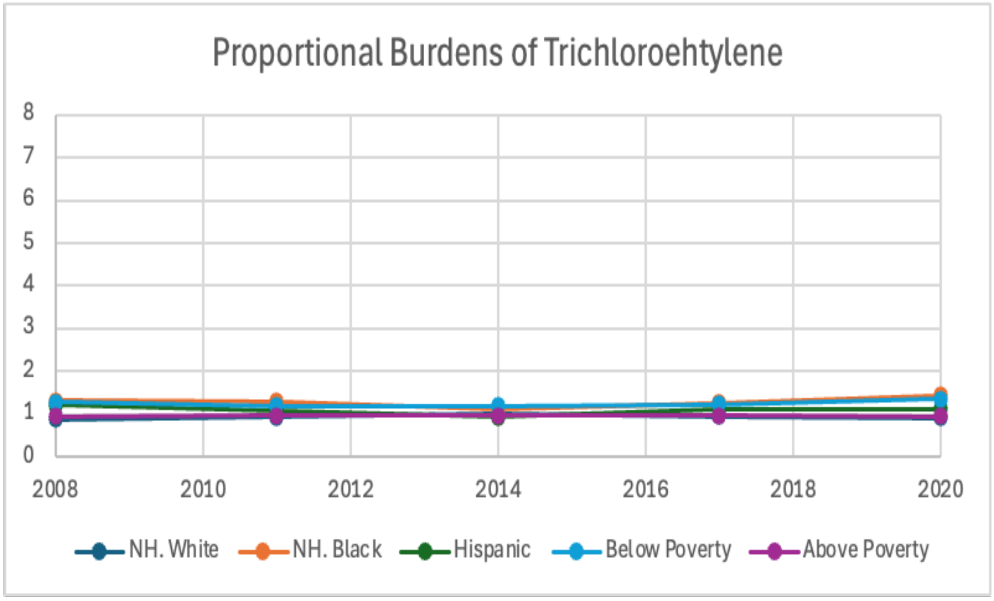

Figure C48. Absolute burdens of trichloroethylene from 2008 to 2020 by population subgroup.

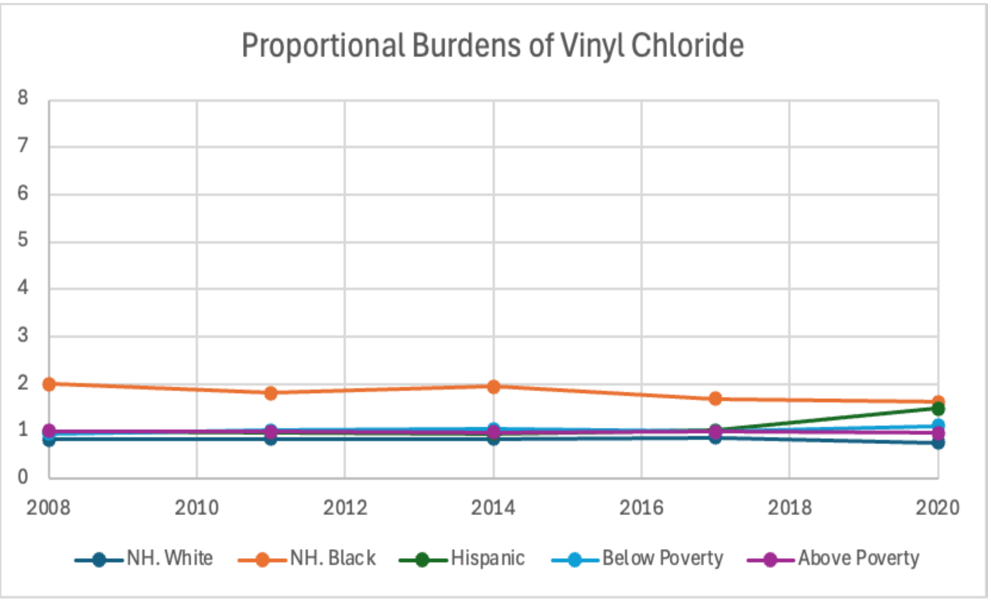

Figure C49. Absolute burdens of vinyl chloride from 2008 to 2020 by population subgroup.

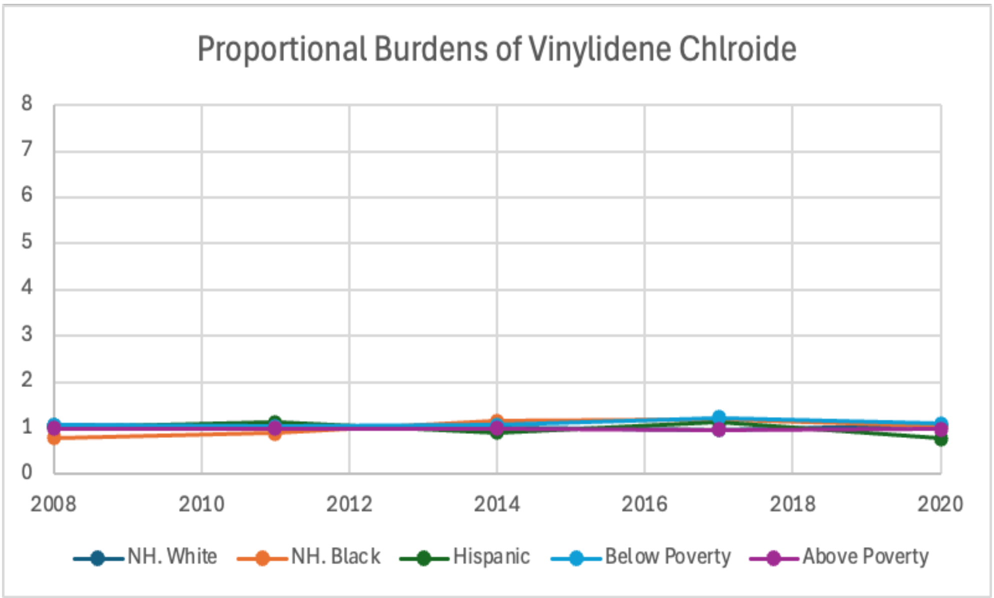

Figure C50. Absolute burdens of vinylidene chloride from 2008 to 2020 by population subgroup.