RESEARCH ARTICLE

Review on the Relationship between Biodiesel and Environment

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Abstract: Biodiesel is a fuel made from vegetable oils or waste grease. While there is a considerable body of evidence on the negative health effects of petroleum diesel exhaust exposures in occupational and urban settings, there has been little research examining the impact of biodiesel fuel on occupational and environmental exposures. This dissertation combined a collaborative exposure assessment of B20 (20% soy-based biodiesel/80% diesel) at a rural recycling center with a policy intervention to deliberate the results of this analysis and potential policy outcomes. Researchers and undergraduate students from Keene State College and employees from the City of Keene Department of Public Works quantitatively estimated diesel and biodiesel exposure profiles for particulate matter (< 2.5 microns diameter), elemental carbon, organic carbon, and nitrogen dioxide using standard occupational and environmental air monitoring methods. I collected qualitative data to examine the genesis, evolution and outcomes of the Biodiesel Working Group. Integrating analysis and deliberation led to a number of positive outcomes related to local use of B20 in nonroad engines. Particulate matter and elemental carbon concentrations were significantly reduced (60% and 22% respectively) during B20 use at the field site. Organic carbon levels were significantly higher (370%) during B20 use. Although NO_2 levels were 19% higher, this increase was not statistically significant. Connecting the analysis with deliberation improved the quality of the exposure assessment, increased dissemination of the research results in the local community, and catalyzed novel policy outcomes, including the development of a unique public/private partnership to manufacture biodiesel locally from waste grease.
Keywords：Biodiesel；environmental exposures；waste grease

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1.Introduction

1.1 An Overview of the Problem of Diesel Exhaust

1.1.1 Use of Petroleum Diesel

Petroleum diesel fuel is the lifeblood of the American economy. Although the vast majority of passenger cars are fueled by gasoline, diesel engines are used in almost all heavy duty trucks, buses, railway engines, marine vessels, as well as countless other industrial and commercial applications. These applications range from the obvious, such as the use of diesel engines to power front loaders and bulldozers at construction sites, to the more obscure, such the use of diesel engines to power air compressors to make snow in New England ski resorts. Decker et al. (2003) effectively illustrate how diesel engines are embedded in the U.S. economy by describing the journey of a shipment of grain from a farm to international export. First, diesel tractors and diesel combines till, plant and harvest the grain, with diesel powered pumps providing irrigation water. Diesel trucks bring the grain to storage silos; from there, diesel powered trains bring the grain to shipping ports where it is loaded onto ocean ships by diesel powered equipment, with diesel electrical generators providing backup power as necessary (Decker et al. 2003). Simply put, diesel engines are the backbone of both the production and transportation of goods and people in this country.

There are about 6 million diesel engines on the road and almost 6 million non road engines in tractors, forklifts, locomotives, construction equipment and other applications (Weinhold 2002).
trucks were sold in the 1990's (EPA 2002a). In 2004, there were approximately 2.7 million trucks registered in Class 8 alone, and 2006 marked a new all time sales record for Class 8 trucks with over 284,000 sold. Most of the 600,000 school buses in the U.S. that transport nearly 24 million children daily are powered by diesel fuel (Wargo et al. 2002).

These diesel engines rely on enormous quantities of petroleum diesel fuel. Approximately 68% of all petroleum was used in the transportation sector in 2006, and 45% of this transportation petroleum is gasoline (Energy Information Administration 2007). While gasoline is clearly the primary petroleum product for the U.S. passenger vehicle fleet, over half of distillate fuel oil – more than 2 million barrels per day – is used as highway diesel fuel. Additionally, the annual gallons of diesel fuel consumed have been steadily increasing – from 29 billion gallons in 1996 to 35 billion in 2000, with annual increases of 2% per year expected into the foreseeable future (Weinhold 2002).

Gas and diesel engines operate differently and so require different types of fuel. The gas powered internal combustion engine car operates by capturing the energy from a spark induced reaction in a cylinder to move a piston. Diesel engines operate by compressing an air and fuel mixture in a cylinder and more efficiently capturing this energy to do useful work. Diesel engines are much more efficient than gas engines (45% versus 30%) (Weinhold 2002).

Increased efficiency means that diesel vehicles typically get better miles per gallon (MPG) when compared to equivalent gasoline vehicles. For example, a diesel powered 4 cylinder 2003 Volkswagen Jetta gets 40 MPG on the highway compared to 27 MPG for a similar sized gasoline powered Jetta (Department of Energy 2008).

Since diesel engines compress air to much higher pressures than gasoline engines, the cylinders in a diesel engine are designed to be more rugged and durable. Due to their better fuel efficiency, power, and engine durability, diesel engines are critical for heavy-duty applications. Many Class 8 engines can go to 1,000,000 miles before their first rebuild, and can be rebuilt several times (EPA 2002a). In addition to transportation applications, these powerful diesel engines have been adapted to a wide variety of non-road applications, such as construction and surface mining.

Due to emerging concerns regarding greenhouse gas emissions and climate change, the potential for diesel engines to get better mileage has focused attention on the difference between gasoline and diesel fuel. In 2003, the transportation sector accounted for about 27 percent of greenhouse gas emissions, up from 24.8 percent in 1990 (EPA 2006).

Although diesel engines are more fuel efficient, emissions of carbon dioxide are greater from combustion of diesel fuel than from gasoline. According to EPA (2007b), 22 pounds of carbon dioxide is emitted per gallon of diesel fuel, compared to 19.4 pounds per gallon of gasoline. It is not clear whether the higher carbon dioxide output offsets the higher efficiency of diesel engines as a way to reduce overall greenhouse gas emissions.

1.1.2 What are the Hazardous Components of Diesel Exhaust?

Although diesel engines have many attractive qualities, the environmental and occupational health effects caused by exposure to petroleum diesel exhaust are daunting. There is substantial scientific evidence of negative health effects associated with exposure to the whole mixture of diesel exhaust, as well as negative health effects associated with exposure to the separate components of diesel exhaust. These health effects range from asthma exacerbation to lung cancer. In this section, I will review the hazardous components that make up diesel exhaust and in subsequent sections examine the literature on health effects associated with exposure.

Diesel exhaust is a complex mixture of over 450 components in vapor and particulate form. The main approach to better understanding the impact of diesel exhaust mixtures on human health has been to focus on the individual components in the mixture and their associated human health impacts.

These burned and unburned products are released as gases or in particulate phase form. The vapor phase consists of carbon dioxide, carbon monoxide, nitrogen oxides (NOx), other inorganic gases, and numerous vapor phase hydrocarbon compounds like benzene and formaldehyde. Besides these gases, particles are emitted from the tailpipe. Primary particulate matter is emitted directly from the tailpipe and secondary particulate matter can form from the gaseous constituents transforming into particles (EPA 2002a).

Particles consist of an insoluble fraction and soluble fraction. The insoluble fraction is the elemental carbon core (EC) or soot and associated metals or ash that can't be dissolved in an organic solution. When diesel exhaust cools as it exits the tailpipe, the unburned fuel and oil condenses or adsorbs to the insoluble particle phase, forming a soluble organic fraction layer on the particle base (HEI 1995). The soluble organic fraction (SOF) is somewhat similar to the organic carbon content (OC) although SOF and OC are measured via different methods. The particles can undergo further atmospheric chemical processes such as oxidation or nitration, however there is limited knowledge on diesel exhaust's chemical and physical transformations in the atmosphere or the toxicological impact of these changes (EPA 2002a).

Inorganic and organic gases such as vapor phase hydrocarbons are not attached to the particulate matter and form their own hazard category. Then the DPM (diesel particulate matter) phase consists of two main fractions: insoluble and soluble. The insoluble components of diesel particulate matter include mainly solid carbon spheres or the aforementioned elemental carbon (EC), with some metals, sulfates, and other unknowns. EC is carbon that is stripped of its hydrogen; EC content can range from 50-75% of DPM mass, depending on fuel, engine operation, and other characteristics (EPA 2002a). Adsorbed to EC is the soluble organic fraction (SOF), or the organic portion of DPM that can be extracted from the particle matrix into solution (EPA 2002a). While SOF and OC represent the adsorbed/condensed material on the solid carbon core, measurement of SOF and OC are by very different methods.

1.1.2a Main Focus of this Study: PM 2.5, NO2, and EC/OC

Although diesel exhaust mixtures are chemically and physically complex and may vary due to engine type, load, operation, and chemical transformation in the atmosphere, there are critical components of diesel exhaust such as fine particulate matter and nitrogen oxides considered by public health scientists to be of primary health concern. This guided the selection of the air contaminants measured in this study. The key species measured were fine particulate matter (or particulate matter less than 2.5 micron in aerodynamic diameter), nitrogen dioxide, elemental carbon, and organic carbon. Fine particulate matter includes the soluble and insoluble fraction (solid carbon) of diesel particulate matter. These air contaminants were selected due to their environmental and occupational health policy relevance and the local expertise and resources available at Keene State College for this study. To demonstrate the health policy relevance, first I will review the scale of the problem of diesel engine emissions' contribution to total PM 2.5 and NOx inventories. Then I will summarize the major literature on human health effects from each pollutant.

1.1.2.b Scale of the Problem of Diesel Exhaust: Contribution of PM2.5, NOx, EC/OC to Ambient Air Pollution

Due to the widespread use of diesel engines, the scale of the problem of associated PM and NOx emissions is significant. Diesel particulate matter is estimated to contribute up to 35% of total annual levels of PM2.5 in some urban areas (EPA 2002a).

Approximately 90% of 2001 PM2.5 emissions from all mobile sources came from onroad and nonroad diesel engines (Decker et al. 2003). The graph shows 64% of PM2.5 came from nonroad diesel engines. By 2006, the total amount of PM2.5 emitted by all mobile sources decreased slightly, but the percent contribution of nonroad engines to the total PM2.5 emissions inventory increased to 69% (EPA 2007a).

Diesel engines are also large contributors to regional and national NOx pollution. Combining both onroad and nonroad diesel engines into one category results in the single largest source of NOx. In 2006, over 1.5 million short tons of NOx were emitted by diesel engines (EPA 2007a).

Determining national inventories of elemental or organic carbon or sources contributing these inventories is not possible at this time. Since PM2.5 and NOx are considered criteria air pollutants regulated by the Clean Air Act, there is extensive monitoring and inventory data available for these contaminants. Elemental and organic carbon data (EC and OC) have been measured by researchers at local scales like the workplace and community. For example, a study of air quality in Harlem neighborhoods determined local EC levels ranging from 1.5 to 6.2 µg/m3 (Kinney et al. 2000). EC can account for up to 90% of total DPM mass (HEI 2002), although in general EC accounts for about 50%-75% of the mass of DPM (EPA 2002a; Ramachadran and Watts 2003). Since most elemental carbon from vehicles is linked to diesel exhaust and not gasoline exhaust, EC is often considered a surrogate measure of total diesel particulate matter, especially in the workplace in the absence of other combustion sources (Cantrell and Watts 1997; Ramachandran and Watts 2003).

1.1.2.c Summary of Human Health Effects of Diesel Exhaust, PM2.5, NOx, and EC/OC

U.S. regulatory agencies have determined that petroleum diesel exhaust is a "potential occupational carcinogen" (NIOSH 1988), and "likely to be carcinogenic to humans by inhalation" from environmental exposures (EPA 2002a). The extensive Multiple Air Toxics Exposure study (also known as MATES-II) conducted in southern California determined that 70% of the air pollution cancer risk for residents of the Los Angeles area was due to diesel particulate emissions (South Coast Air Quality Management District 2000). Exposure to diesel exhaust is also associated with a number of acute and chronic non-cancer health effects, ranging from nasal/eye irritation, decreased lung function, and increased cough to symptoms of bronchitis, chronic inflammation of lung tissue and reduced resistance to infection (SCAQMD 2000; EPA 2002a).

A number of researchers have suggested that diesel exhaust may contribute to allergic responses and asthma (Wade and Newman, 1993; Mauderly 2000; Pandya et al. 2002; EPA 2002a). Incidence of asthma has more than doubled from the 1978 to 1998 time period, affecting over 17 million people and highlighting the concern about possible associations between asthma and combustion related products such as diesel exhaust (EPA 2002c). A recent study of asthma rates in New England, which are consistently higher than the rest of the country, indicated 475,000 New England children (14%) and 1.62 million New England adults (15%) have been diagnosed with asthma in their lifetimes (Asthma Regional Council 2006). Asthma rates for New England children in the lowest income group were almost twice as high as asthma rates for New England children in the highest income group, and rates across all groups have been increasing (ARC 2006). There are a number of hypotheses for these increasing rates, including the impact of air pollution in urban areas. Diesel particulate matter may promote immunologic responses associated with asthma, which may help explain why some epidemiologic studies show an increased risk between children living near trucking routes and asthma (Pandya et al. 2000). EPA (2002a) has noted that children, the elderly, and people with existing heart and lung diseases like asthma are especially susceptible to the effects of whole diesel exhaust exposure.

The carcinogenic potential of whole diesel exhaust presents a major occupational and environmental health challenge. Although mutagenic and carcinogenic species have been identified in the organic carbon part of diesel particulate matter, there remains significant controversy regarding the strength of the association between environmental or ambient diesel exhaust exposures and lung cancer risk for the general public. Occupational exposures to diesel exhaust seem to indicate elevated lung cancer risk. The reported relative risks of long-term diesel emissions exposure in occupational settings range from 1.2 to 1.5, which indicates a 20 to 50% increased risk of developing lung cancer (HEI 1995). There have been at least forty epidemiological studies looking at lung cancer risk from diesel exposure (Mauderly 2000). However, though many of these epidemiological studies seemed to support a connection between lung cancer and human exposure, there has been such variety in methodological approaches – such as how smoking among study participants was addressed or whether exposures were directly quantified or instead estimated – that there continues to be a lack of scientific consensus regarding interpretation of the results and controversy regarding the findings (HEI 1995; EPA 2002a). In the next sections, I will review the health effects for each of the major components of diesel measured in this study.

1.1.3 Individual Hazardous Components: Health Effects

1.1.3.a Fine Particulate Matter (PM2.5)

Diesel exhaust is an important source of fine particulate matter (PM), or particulate matter less than 2.5 micron in mean aerodynamic diameter. As 80 to 95% of DPM mass is less than 1.0 micron in diameter (with a mean particle diameter of 0.2 micron), almost all DPM is less than 2.5 micron in diameter (EPA 2002a). Fine particulate matter's main hazard is its ability to penetrate into the deep lung during inhalation. Particulate matter at this size is associated with numerous negative health effects including but not limited to increased mortality, direct lung injury (i.e., increased inflammation), cardiovascular effects (i.e., increased risk of arrhythmia in people with heart disease) and other organ effects (Lippmann et al. 2003).

Fine particulate matter exposure is especially problematic for certain groups within the national population. Health researchers have shown an association between the incidence of cardiovascular death and disease among postmenopausal women and long term exposure to PM2.5. Miller et al. (2007) studied over 65,000 postmenopausal women without history of heart disease in 36 U.S. urban areas with an estimated mean exposure to PM2.5 of 13.5 ug/m3. These researchers determined (with a 6 year median followup) that each increase in 10 ug/m3 was associated with a 24% increase in the risk of a cardiovascular event, and a 76% increase in the risk of death from cardiovascular disease (Miller et al. 2007).

Sensitive subpopulations, such as older adults, children, and those with preexisting heart or lung disease are at increased risk from particle exposure and their associated health impacts

(EPA 2003b; Pope 2000). Although the elderly, infants, and people with chronic diseases like asthma are more likely to experience death or serious illness from acute elevated fine PM exposures, the larger population is susceptible to the cumulative effects of chronic low level exposures, resulting in a predicted reduced life expectancy in areas with high particulate matter pollution (Pope 2000). More recently, particulate matter from all sources including diesel exhaust has been linked to reproductive problems and diabetes (Weinhold 2002). These and other studies support that PM2.5 exposures are an occupational and environmental health policy problem.

1.1.3.b Elemental Carbon (EC) and Organic Carbon (OC)

Elemental carbon (EC) or the solid carbon core portion of diesel particulate matter is considered an especially potent component of the diesel exhaust mixture. These carbon particles can cause lung irritation and inhibit lung clearance mechanisms in animals, similar to other dusts like talc or silica (HEI 1995). As mentioned, EC makes up from 50-90% of DPM. The small size of the EC particle (typically less than 1.0 micron) also means it is reasonable to associate the health effects of PM2.5 described in the previous section with DPM or EC (EPA 2002a). However, another important health concern for EC is related to its high specific surface area. The combination of small EC diameter size and high surface area means that EC is an effective carrier of adsorbed chemicals that can reach the deepest portions of the respiratory tract (EPA 2002a). EC is also strongly correlated with combustion of diesel fuel rather than other combustion sources. While EC is not 'one-to-one' measure of DPM, at this time EC is considered the best available "diesel signature" (HEI 2002).

The organic carbon content of DPM can range from 19 to 43% (EPA 2002a). Organic carbon is mostly unburned fuel and lubricating oil but also may contain PAH's and nitro- PAH's of key health concern. Many of the PAH's and nitro-PAH's identified in the organic carbon or soluble organic fraction of DPM are considered mutagenic or carcinogenic (EPA 2002a; HEI 2002). These mutagenic and carcinogenic organic compounds adsorb or condense on the elemental carbon core. The EC acts as a velcro-like platform, the OC sticks to the EC, and the combination becomes an advanced inhalation delivery system of toxics to the lungs.

1.1.3.c Nitrogen Oxides (NOx)

Diesel engines also contribute large amounts of vapor phase NOx to regional airsheds.

NOx is both a health concern from direct health effects such as lung irritation and an environmental concern due to the role of NOx in ground level ozone formation. The main oxides of nitrogen include nitric oxide and nitrogen dioxide. Nitrogen dioxide was measured in this study and will be reviewed here.

Nitrogen dioxide is a severe respiratory irritant, with changes in pulmonary function noted at levels of 2 to 3 ppm, progressing to symptoms such as painful breathing as levels increase and leading to fatal lung injury at levels in excess of 50 ppm (OSHA 1991).Nitrogen dioxide symptoms can be delayed up to 12 hours after exposure (OSHA 1991).

Nitric oxide and nitrogen dioxide exposures tend to exist concurrently since NO is rapidly oxidized to nitrogen dioxide, with interconversion between species. While NOx can come from natural sources such as volcanic activity and lightning, manmade production of NOx comes mostly from combustion of fossil fuels, mainly in the form of NO from internal combustion engines (Manahan 2000). NIOSH has experimentally approximated a ratio of 35% NO2 /65% NO in industrial settings where diesel exhaust is a primary source of exposure (NIOSH 1976). Although NOx from diesel engines is primarily emitted in the form of NO, nitrogen dioxide is more harmful to human health at lower levels, and as such is a criteria air pollutant under the Clean Air Act.

Nitrogen dioxide's potential to photodissociate (or split into NO and O) in sunlight means it plays a critical role in ground level ozone formation with associated serious environmental and health impacts. Both nitric oxide and nitrogen dioxide contribute to smog formation by increasing ground level ozone, a respiratory irritant and major contributor to poor visibility or environmental haze. Ozone can cause lung and throat irritation, make breathing more difficult, and aggravate asthma (EPA 2003a). When nitric oxide emitted from diesel engines is converted to nitrogen dioxide, the subsequent photodissociation in sunlight starts a series of chain reactions contributing to ground level ozone and smog. Smog increases susceptibility to adverse health effects such as lung tissue damage, decrease in lung function, asthma, and negatively impacts crop yields/vegetation (EPA 2008b). NOx emissions cause other problems such as acid rain, water quality deterioration, the formation of toxic chemicals in our atmosphere, and decreased visibility (EPA 2008b). Thus any source of NOx, including those from diesel engines is an environmental and human health concern.

1.1.3.d The Particulate Matter/Nitrogen Oxide Tradeoff

EPA has regulated NOx emissions from heavy duty diesel engines since 1985, with allowable emissions decreasing since that time. However, a further technical and policy complication is the PM/NOx tradeoff in diesel engines: high combustion temperatures are needed to combust PM fully, yet these same high temperatures will lead to increased NOx formation in the exhaust (HEI 1995). Lower temperatures or poor air/fuel mixing – indicators of poor combustion – will lead to lower NOx emissions but higher PM emissions. The inverse relationship of NOx/PM is the main barrier to lowering diesel emissions (Yanowitz et al. 2000). Since both PM and NOx are undesired emissions, engine designers attempt to balance the undesired outputs against engine performance. The PM/NOx tradeoff is also a challenge for alternative fuel considerations because oxygenated fuels like biodiesel may decrease PM but increase NOx.

1.1.4 Environmental and Occupational Health Concerns of Diesel Exhaust

As defined by the World Health Organization (1993), environmental health "refers to the theory and practice of assessing, controlling, and preventing those factors in the environment that can potentially affect adversely the health of present and future generations." Occupational health is defined as the "multidisciplinary approach to the recognition, diagnosis, treatment, and prevention and control of work-related diseases, injuries, and other conditions" (Levy and Wegman 2000). With respect to chemical exposures, occupational health examines the relationship between disease and workplace exposure, and environmental health examines the relationship between disease and a human populations' exposure to risk factors in the environment. Environmental health typically looks at disease/exposure relationships at a regional or global scale compared to a facility or organizational scale for occupational health.

Diesel exhaust exposures present both an environmental health and occupational health problem. As shown in the previous sections, the scale and volume of diesel exhaust emissions such as the contribution of diesel emissions to ambient background levels of PM2.5 and NO2 is significant. PM2.5 impacts are of special environmental health concern, as numerous studies have consistently shown elevated fine particulate matter levels are correlated with increased hospital admissions and emergency room visits (EPA 2007c).

These environmental health impacts may also be disproportionate depending on socioeconomic status. Concerned about rising asthma rates in Harlem neighborhoods, a community based research study determined that DPM exposures in urban Harlem neighborhoods were elevated near diesel sources like bus depots (Kinney et al. 2000). DPM has been identified as having a key role in enhancing inflammatory and allergic responses in the lung (Diaz-Sanchez 1997; EPA 2002a). Environmental justice advocates maintain that incidence of asthma – and the link to diesel sources - disproportionately occurs in poorer neighborhoods (Kinney et al. 2000; Corburn 2005).

Diesel exhaust also poses an occupational health concern, as NIOSH (1988) has estimated over 1 million people are occupationally exposed to diesel emissions.

Occupational exposures pose numerous noncancer health risks like lung inflammation, bronchitis, and asthma. A spectrum of epidemiological studies has indicated an increased risk of lung cancer associated with diesel exposure. For example, a detailed cohort study of railroad workers with occupational exposure to diesel exhaust indicated elevated lung cancer mortality (Garshick et al. 2004). However, EPA's (2002a) meta-review of the epidemiological literature of occupational exposure to diesel exhaust in various jobs (such as trucking, mining, construction, and railroad workers) indicated a moderately increased relative risk of lung cancer but numerous methodological problems. Main points of controversy were correction (or lack thereof) for the impact of smoking on lung cancer cases, lack of a clearly identifiable diesel signature or singular marker for diesel exposure, and the use of surrogates for exposure (such as job title) due to the lack of measured, quantitative exposure data (EPA 2002a). These issues of scientific uncertainty have prevented development of a definitive dose-response curve for human exposure.

Diesel exhaust exposures remain a health concern for workers because occupational diseases like lung cancer may take decades to manifest, and external variables (such as high ambient background air pollution) make causality difficult to prove. In addition, certain work scenarios can result in combined environmental and occupational health impacts.

Emissions from construction equipment can create unique microenvironments of elevated diesel exhaust levels, posing an increased health risk for equipment operators. Long term construction projects can create hazards for not only workers but nearby residents as the construction site becomes a semi-permanent source of air pollution in the local community. A recent exposure assessment performed for Northeast States Coordinated Air Use Management (NESCAUM) measured construction and industrial worker PM2.5 exposures ranging from 1 to 16 times greater than background levels (Treadwell et al. 2003). The report estimated that as many as 200,000 workers may be exposed to harmful levels of diesel exhaust from nonroad equipment in the northeast (Treadwell et al. 2003).

In summary, in both the environmental and occupational health context, diesel exhaust poses a daunting challenge. In the next section, I will discuss the current regulatory approaches to manage risk from diesel exhaust exposure in the environment and workplace.

1.1.5 Current Regulatory Approaches for Managing Diesel Exhaust Exposures

1.1.5.a The Environmental Protection Agency's Regulatory Approach

EPA's main regulatory approach to manage diesel exhaust exposures has been two fold: requiring enhanced engine technology in new engines to reduce emissions, and reduction in sulfur content of highway diesel fuel from 500 ppm to 15 ppm. This ultralow sulfur diesel (ULSD) has been phased in since 2006, and as of 2007, new model heavy duty on road engines are required to meet stringent tailpipe emissions requirements that will significantly reduce PM and NOx by as much as 90%. The emissions standards are based on new catalytic emissions control devices or other technology improvements, and are expected to reduce annual emissions of NOx and PM by 2.6 million tons and 109,000 tons, respectively, by the year 2030 (EPA 2000). When fully implemented by 2030, the emissions reductions are expected to prevent over 8000 premature deaths, 9500 hospitalizations, and 1.5 million lost work days an annual basis (EPA 2000). Similar regulatory schemes will apply to nonroad engines, although emissions controls will not be required until 2014, and smaller engines do not have to meet the stringent emissions requirements of larger ones (EPA 2004). Nonroad diesel fuel sulfur content will be reduced to 500 ppm by 2007 and to 15 ppm by 2010.

EPA has also initiated a number of voluntary programs to encourage the replacement of existing engines with cleaner ones or place new retrofit emissions control technologies (such as oxidation catalysts) onto existing tailpipes. EPA provides technical and financial assistance through its voluntary National Clean Diesel Campaign for those eligible fleets that work towards reducing emissions. The Clean School Bus USA program encourages a number of strategies such as particulate filters, cleaner fuels (such as biodiesel) and anti- idling programs.

States have also tried to implement different policies and in some cases laws to reduce diesel exhaust pollution. In the Northeast, Connecticut, Massachusetts, and New Hampshire have anti-idling regulations (EPA 2008a). For example, New Hampshire has codified at Env-A 1101.5 that diesel engines may not idle for more than 5 minutes when the outdoor temperature is above freezing.

Finally, EPA has established a reference concentration (RfC) of 5 µg/m3 as an acceptable diesel exhausts exposure. This value is averaged over a 24 hour period, everyday for a lifetime, and is based on noncancer health effects only. The reference concentration of 5µg/m3 is considered sufficiently protective of the general population for a lifetime of exposure without experiencing adverse respiratory effects like lung inflammation. However, the reference concentration mainly provides policy guidance for determining if air quality is acceptable from a health standpoint; there is no compliance or action-forcing provision if RfC is exceeded.

In contrast, although not specific to diesel exhaust, EPA does have other health-based regulatory programs in place to control exposure to the components of diesel, such as the Clean Air Act's National Ambient Air Quality Standards for PM2.5 and NO2 levels. In 2006, in response to the growing body of knowledge of public health impacts from particulate matter, EPA lowered the National Ambient Air Quality Standard, commonly thought of as the "safe level" of exposure, from 65 to 35 µg/m3 for a 24 hour average (EPA 2007c). The NAAQS for nitrogen dioxide has remained at 100 µg/m3 average for an annual period.

Under the Clean Air Act, states are required to submit State Implementation Plans to reduce air pollution and monitor air quality to ensure pollution is controlled. If air quality exceeds the NAAQS, the state could face sanctions from the federal government. States try to control sources of air pollution within their borders via permits and programs in order to ensure ambient air quality stays in attainment of NAAQS.

1.1.5.b The Occupational Safety and Health Administration's Regulatory Approach

The Occupational Health and Safety Administration does not regulate whole diesel exhaust exposure in the workplace. There is no Occupational Safety and Health Association permissible exposure limit (PEL) for diesel exhaust or diesel particulate matter. Although not legally binding, a DPM level of 150 µg/m3 was proposed by the ACGIH (American Council of Governmental Industrial Hygienists) in 1995-1996. The proposed DPM exposure level was reduced to 50 µg/m3 until the ACGIH withdrew the DPM listing in 2003. There is no legally binding standard other than in mines where MSHA limits average workday DPM exposure to 160 µg/m3. Outside of mines, any reductions to diesel exposures in the workplace such as ventilation controls or "no idling" policies result from voluntary actions by employers.

With respect to the components of diesel exhaust, under the broader category of particulate matter exposure (which includes non-diesel sources of particles such as dusts), OSHA's permissible exposure limit is 5000 µg/m3 compared to EPA's level of 35 µg/m3. The OSHA PEL is an 8 hour time weighted average, as opposed to EPA's 24 hour time weighted average exposure limit. OSHA considers a PEL to be the allowable exposure for a worker that will not result in adverse health impacts if that worker were exposed 8 hours a day, 40 hours a week, over an entire career. OSHA's PEL for nitrogen dioxide is a 9000µg/m3 ceiling limit that cannot be exceeded during a workshift compared to 100 µg/m3 averaged over a year. While OSHA does have diesel exhaust listed on its website as a safety and health topic, the information and links are mainly educational and point out the individual component PEL's and regulatory actions taken by EPA to manage the risk of diesel exhaust.

1.1.5.c The Insufficiency of Current Regulatory Approaches

There are a number of reasons why current regulatory approaches are insufficient. Ironically, one need not go any further than EPA's own National Clean Diesel Campaign (2007b) website to find justification for the need for faster action to reduce diesel exhaust exposures:

Even with more stringent heavy-duty highway engine standards set to take effect over the next decade, over the next twenty years millions of diesel engines already in use will continue to emit large amounts of nitrogen oxides and particulate matter, both of which contribute to serious public health problems. These problems are manifested by thousands of instances of premature mortality, hundreds of thousands of asthma attacks, millions of lost work days, and numerous other health impacts.

In short, due to the durability and longevity of onroad and nonroad diesel engines and vehicles, EPA's main regulatory approach will not fully produce human health dividends until 10 to 20 years from now. New engines will very slowly replace existing diesel engines in current fleet inventories. Another generation of children, the elderly, workers and the general public will continue to be exposed to harmful levels of diesel exhaust. The public health concern is more critical in urban areas, such as in Los Angeles, Boston and New York City. Data from a community air quality study in Harlem, New York City (Kinney et al. 2000) indicated that locations with high diesel vehicle counts exceeded the 5 µg/m3 reference concentration set by EPA to protect against lung impacts.

The current regulatory approach focuses mainly on PM and NOx, not on the carcinogenic potential of diesel exhaust. Due to the scientific uncertainty regarding the association of diesel exhaust exposure with carcinogenic effects like lung or bladder cancer, it is unlikely stronger or faster regulatory action will occur. EPA's (2002a) weight of evidence approach in the Health Assessment Document concluded that diesel exhaust could only be classified as a B1 probable human carcinogen by inhalation at lower level environmental exposures due to numerous uncertainties. The uncertainties cited by EPA included a lack of understanding of diesel exhaust's cancer causing mechanism in humans, lack of scientific consensus regarding the relationship between occupational exposures and lung cancer, and expected changes in future engine and fuel technologies which would change future diesel exhaust exposures (EPA 2002a).

However, due to the identification of mutagens and carcinogens in diesel exhaust, and belief that no safe exposure threshold for mutagens and carcinogens exists, many scientists and advocates remain concerned that EPA's B1 assessment of diesel exhaust does not adequately protect public health. Typically EPA will advance regulatory options when the risk of cancer is at a 1 in 1,000,000 level (one excess cancer case per million people exposed). Although risk estimates from diesel exposure were not listed in the Health Assessment Document, other EPA policy documents put the risk estimate at 1 in 1,000 to 1 in 100,000 (Weinhold 2002). Although not enough to change its overall risk assessment, EPA allowed that evidence of mutagenic potential meant "a cancer hazard is presumed possible" at lower or environmental exposure levels (EPA 2002a).

Although EPA followed the steps to risk assessment outlined by the National Research Council (1983) report in developing its Health Assessment Document, there were major departures from typical EPA policy. Usually, the end product of a risk assessment is a quantitative estimate of excess unit cancer risk, sometimes also called the slope factor or potency estimate. Many researchers felt the mechanism that appeared to cause cancer in rats (via "lung overload") was not specific to diesel exhaust exposure and not expected to occur in humans (EPA 2002a). Due to scientific uncertainty EPA (2002a) did not develop a definitive dose-response curve or slope factor for diesel exhaust.

The practical impact of not having a slope factor or cancer unit risk estimate is limited federal action to reduce diesel exposures via health protective emissions controls (Treadwell 2005). In other words, EPA completed a quantitative risk assessment, without ever finalizing an actual quantitative level of risk from exposure to diesel exhaust. Without an estimated level of risk, it is difficult to implement a cohesive regulatory approach to reduce diesel exposures to levels protective of human health. In contrast, maximum achievable control technology is required for carcinogenic air toxics emissions from industrial sources. Without a potency estimate, diesel exhaust exposures continue because they are not considered urgent enough for immediate and stringent control. It is also worth noting that the scientific discussion and review necessary to complete the EPA Health Assessment Document took over 10 years to finalize, due to ongoing debate between stakeholders and regulators, including the desire to review the latest science at each meeting (Treadwell 2005). It took over 10 years of debate in scientific and policy making circles to issue a nonbinding reference concentration value. With this background context, attempting to reconcile significant scientific uncertainty for more rapid implementation of emissions controls seems highly improbable.

Due to their proximity to sources of diesel emissions, workers as a subpopulation experience even higher exposures and have little to no regulatory protection. Occupational exposures to diesel exhaust tend to be much higher than environmental or ambient air exposures, posing increased risk to workers such as mechanics, miners and railroad employees (Cantrell and Watts 1997). In their seminal research study, Zaebst et al. (1991) found diesel mechanics and diesel forklift operators had diesel exposures significantly higher than background levels. A more recent diesel exposure assessment determined elevated levels of PM2.5 and EC at sites that use nonroad equipment such as construction, farming, and a rural lumber yard (Treadwell et al. 2003). Treadwell and colleagues (2003) found workers at construction or similar sites were exposed to near field and in-cabin levels of PM2.5 ranging from 2 to 660 µg/m3 , levels that were 1 to 16 times higher than background ambient levels.

The main way OSHA protects workers from chemical exposure risk is through enforceable permissible exposure limits (PEL's). As mentioned, there is no PEL for diesel exhaust, even though EPA (2002a) concluded "available human evidence shows a lung cancer hazard at occupational exposure levels" and NIOSH (1988) – the research arm of OSHA – concluded that diesel exhaust was a probable occupational carcinogen.

Additionally, although there are existing PEL's for diesel exhaust components such as particulate matter, these "safe" levels are orders of magnitude higher than EPA "safe" limits for the same chemical (5000 µg/m3 [OSHA] vs. 35 µg/m3 [EPA]). Treadwell (2005) points out even when the different averaging times are considered in the calculations (OSHA averages the exposure over an 8 hour workshift versus EPA's 24 hour day), workers can be exposed to daily particulate matter levels below occupational health limits but far above acceptable environmental health limits. Due to the discrepancies in EPA/OSHA health protective values, assuming a 5 µg/m3 background PM2.5 exposure, workers could theoretically experience the dose equivalent of about 48 EPA "unhealthy air" days in a single workshift. In a relatively short time, workers could experience a lifetime equivalent exposure in scenarios that would be considered completely unacceptable for a resident just outside the facility fence.

Diesel exhaust is an example of a chemical exposure risk vigorously debated in the environmental health sphere but not considered a priority risk in the workplace. "Acceptable" chemical exposure levels vary depending on whether one is standing inside or outside the facility fence. Some scholars consider the difference between the higher chemical exposure levels allowed by OSHA compared to EPA a manifestation of a hidden "ideological hazard" that considers worker health protection differently from the general public (Kasperson and Kasperson 1991). A "double standard" exists as a result of an ideological view that emphasizes the power of private business in the United States, and underscores the general reluctance of government to interfere with business operations. This lack of a health protective PEL also raises questions of environmental justice. Workers are more at-risk than the public due to higher exposure levels yet there is no workplace regulation. In summary, the case of diesel exhaust illustrates a disconnect between environmental and occupational health with respect to management of chemical exposures. Some of the possible reasons for the discrepancies will be discussed in the next section.

1.2 How the Problem of Diesel Exhaust Highlights a Disconnect Between Environmental and Occupational Health Risk Management

As mentioned, NIOSH (1988) identified diesel exhaust 20 years ago as a potential occupational carcinogen, estimating at the time that over 1,000,000 workers were exposed to diesel exhaust. The EPA Health Assessment Document noted the occupational data were "strongly supportive" of a diesel exposure–lung cancer link but did not regulate as a carcinogen and instead issued a reference concentration of 5 µg/m3 to protect the public from noncancer health effects (EPA 2002a). No OSHA PEL exists for diesel exhaust. The PEL's that do exist for components of diesel – such as nitrogen dioxide and particulate matter – are 10 to 40 plus times higher than allowable EPA recommended limits. Why do such discrepancies between protection of environmental/public health and protection of occupational health persist? Though referring to other workplace hazards and not specifically to diesel exhaust, Shrader-Frechette (2002) argues the increased risk many workers face in the U.S. today is a clear example of environmental injustice. According to Shrader-Frechette (2002), if environmental justice is concerned with equalizing the burden of pollution across all segments of society, then environmental injustice occurs when one group bears a disproportionate risk, has less opportunity to participate in decision-making or has less access to environmental goods. Workers exposed to diesel exhaust appear to experience a disproportionate risk of exposure to diesel exhaust and also appear to have less opportunity to participate in decision making.

Both Shrader-Frechette (2002) and Kasperson and Kasperson (1991) suggest that the OSHA and EPA discrepancies in chemical exposure standards exist due to embedded societal beliefs including the following: job selection is considered a voluntary, individual choice, workers are both well compensated and well informed of the risks, and workers' compensation programs exist to pay for work-related injuries and illnesses. Shrader- Frechette's (2002) detailed analysis debunks many of these societal beliefs, showing for example, that workers in high hazard industries often do not earn better pay, nor are they well informed of the risks. Her arguments are compelling and outline important societal and ethical questions as to the fairness of different 'safe' exposure limits between agencies.

However, there are also a number of other, arguably more structural barriers that impede progress toward an integrated chemical risk management approach protective of both environmental and occupational health. In the following sections, these barriers will be reviewed.

1.2.1 EPA vs. OSHA: Mandates

There are several explanations for why the discrepancy between EPA and OSHA safe exposure limits exists. Embedded within the broader environmental justice argument are a number of regulatory and institutional barriers that foster a separation between environmental and occupational health practice. Ironically, early research in the risk analysis field identified the workplace as a key source of present and future environmental risks and suggested that the workplace was an ideal hazard monitoring system, because exposures could be easily identified, monitored and effects on employees documented (Fischhoff et al. 1981). This viewpoint saw the workplace as the proverbial canary in the coal mine for environmental health risks and also that workplaces were clearly situated in the outside environment creating environmental health risks. Yet the swift passage of numerous environmental laws in the 1970's led to the emergence and evolution of dramatically different legislative mandates and agency cultures that helped create an artificial divide between the workplace and outside environment.

The divergent agency mandates of EPA and OSHA lead to significant regulatory barriers. EPA has responsibility to develop and enforce regulations for over 30 environmental laws, such as the Clean Air Act and Clean Water Act, while OSHA has responsibility for only one law, the Occupational Health and Safety Act (OSH Act).

Environmental chemical hazards may be present as pesticide residues, new chemicals entering into commerce, or sources of air pollution from industrial sources. How EPA regulates chemical exposure risk depends on the environmental law as EPA is only authorized to take those actions specified within each law. Depending on the statute, EPA may or may not have to consider the economic or technological feasibility of compliance. Under the Clean Air Act (CAA), EPA does not have to consider economic or technological feasibility in developing health protective standards for the criteria pollutants (such as particulate matter), but must consider such feasibility in promulgating maximum achievable control technologies for chemicals identified as hazardous air pollutants (such as benzene). As another example, under the Toxic Substances Control Act (TSCA), EPA must balance risk to human health against the benefits of the chemical (to consumers and manufacturers) in order to make a determination of "unreasonable risk" (Cranor 1993). Per TSCA the burden of proof is on EPA to prove that a chemical is unsafe or that an extremely large number of people will be exposed in order to compel a company to perform additional toxicity testing.

These varying mandates set up a complex web of regulations that requires administration by technical experts in both the agency and the regulated industries, often setting up an adversarial relationship between experts over the finer points of regulatory interpretation and implementation. Other regulatory and institutional barriers have evolved since the 1970's. Environmental regulations are categorized by media (air, water, and soil), rely heavily on intense judicial review, focus narrowly on compliance rather than prevention, and center mainly on "end-of-pipe" controls (Fiorino 2006). In addition, environmental regulation, with its reliance on technical expertise, legal interpretation, and politically neutral managers, is also an excellent example of bureaucratic rationality (Fiorino 2006). However, there is a common thread throughout much of the environmental regulations that pertain to managing chemical exposure risk: EPA as an institution relies on quantitative risk assessment as an analytic tool to help meet statutory requirements and justify regulatory actions.

OSHA manages chemical exposure risk mainly through adoption and enforcement of permissible exposure limits. OSHA can initiate a new standard on its own or on petition from any other interested party, usually with input from an advisory committee (Ashford 2000). OSHA must also consider the economic and technological feasibility of the proposed standard. As such, setting health protective chemical exposure standards has been difficult for OSHA to implement in practice. OSHA has not updated the vast majority of its PEL's since the initial adoption in 1971 and most of these PEL's consider only noncancer health effects. The reasons why are related to OSHA's institutional use of risk assessment and are reviewed next.

1.2.2 EPA vs. OSHA: Institutional Culture of Risk Assessment

EPA uses quantitative risk assessment as a tool to characterize risks posed by chemical hazards much more frequently compared to OSHA. Although EPA utilized quantitative risk assessment techniques since its inception, in the 1980's returning EPA Administrator William Ruckelshaus more fully embraced the National Research Council's (1983) risk assessment/risk management paradigm (Graham 1995). Ruckelshaus emphasized that much of the language in environmental laws contained "pious hope" that could not be met in practice and more pragmatic goals of risk management were needed (Ruckelshaus 1985). Under Ruckelshaus, EPA increasingly relied on risk assessment to meet evidentiary requirements within environmental statutes, especially to help determine acceptable risk levels for carcinogenic chemical exposures. Quantitative risk assessment provided a defensible basis for agency decision-making, or what Jasanoff (1991) refers to as "a lifeline to legitimacy."

Depending on the statute, EPA typically begins risk management policy deliberations at a risk level of 1 in 1,000,000 (one excess cancer case per 1,000,000 people exposed). Risk is typically defined in technocratic terms, as the probability of a hazardous injury/illness occurring. Simplify a very complex process, inhalation cancer risk is ultimately calculated by the equation: risk = exposure X toxicity, where exposure is the concentration of the chemical in air and toxicity is represented by the slope factor or unit cancer risk value.

Exposures are then regulated via risk management policy decisions to ensure these risk levels are not exceeded. Since its inception, the benefits to EPA of risk assessment as an analytical tool soon became clear: allowable pollutant emissions levels could be standardized, clean-up standards at contaminated sites could be specified, acceptable levels of exposure could be determined, and enforcement mechanisms could be developed in a straightforward manner (Ginsburg 1997).

In summary, EPA's use of risk assessment increased dramatically during the 1980's as the scientific underpinning of regulatory decisions. Per the NRC (1983) paradigm, the more scientific risk assessment process was kept separate from - but fed information into - agency risk management or risk decision-making functions. The NRC (1983) paradigm is still prominent today, as exemplified by the recent diesel health assessment document.

In contrast, regulation of occupational chemical hazards is generally limited to the smaller universe of those chemicals common in the workplace. Unlike EPA, OSHA did not formally use risk assessment in the 1970's. At the time, OSHA did not consider risk assessment to be a necessary step in setting health standards under the OSH Act (Jasanoff 1986). OSHA viewed risk assessment as a potential tool to prioritize among risks but not to determine regulatory exposure levels (Cranor 1993). OSHA relied more on it's expertise and it's authority under the OSH Act in making decisions. In 1971, OSHA adopted as consensus standards the 1968 ACGIH (American Conference of Governmental Industrial Hygienists) threshold limit values (TLV's) for 450 chemicals, renaming them permissible exposure limits (PEL's). The PEL's are the centerpiece of OSHA's approach to chemical health risk management – employers are expected to keep workplace exposures below these limits, with penalties for non-compliance. PEL's are mainly protective against noncancer effects, and are based on a threshold concept, or that a threshold of exposure exists for most people below which adverse health effects are not expected to occur.

While many toxicologists do support the concept of a threshold for noncancer effects, many do not believe the threshold concept applies to carcinogens (Graham 1995). Many scientists believe there is theoretically no safe exposure threshold for a carcinogen because any exposure is associated with an increased cancer risk. OSHA was so concerned about exposure to workplace carcinogens that it proposed a generic carcinogen standard in 1977 that would regulate exposures to the lowest feasible levels (Graham 1995). Risk assessment wasn't needed by OSHA to establish a safe level or "acceptable" exposure level for carcinogens, as the goal was best practicable control to the lowest possible exposure level. With the proposed generic carcinogen standard, OSHA tried to avoid case-by-case, individual chemical risk assessments. Individual risk assessments can take 5 or more years to complete and are resource intensive (Cranor 1993).

However, industrial interests argued that risk assessment should be used to determine if the size of the carcinogenic risk was significant and to estimate health benefits in a cost benefit analysis of regulatory alternatives (Graham 1995). Some industrial legal challenges went all the way to the Supreme Court. In 1980, the "Benzene" case (Industrial Union Department AFL-CIO vs. American Petroleum Institute [448 U.S. 607]) became one of the most influential cases regarding OSHA's authority to issue health standards. OSHA had proposed to reduce the existing permissible exposure limit (PEL) of benzene, a known human carcinogen, from 10 ppm to 1 ppm, which was considered a feasible level. A majority of the Court ruled that OSHA did not provide substantial evidence that there was a "significant health risk" to workers at the present exposure level. OSHA was directed by the Court to use appropriate quantitative methods such as risk assessment to show workers were at significant risk at the present exposure level and that that risk would be reduced by the proposed standard (Jasanoff 1986). In short, agency expertise was not considered sufficient, and OSHA was directed to use quantitative techniques to evaluate risk.

After the "Benzene" decision, OSHA began conducting quantitative risk assessments for carcinogens and suspended the generic carcinogen standard (Jasanoff 1986). In addition, OSHA selected the lower range of the Court's suggested risk spectrum, and considered those occupational exposures that posed an excess cancer risk greater than 1 in 1000 as a starting point for further regulatory attention. Going forward, the 1 in 1000 value became OSHA's "bright line" decision rule for unacceptable risk. But there was a large universe of chemicals beyond carcinogens that posed health risks to workers. In 1989, OSHA proposed updating the bulk of the 1971 PEL's list to add new chemicals and to reflect more recent scientific information on existing chemicals. These updated PEL's were vacated in 1992 by the Eleventh Circuit Court of Appeals (AFL-CIO v. OSHA, 965 F.2d 962 [11th Cir. 1992]), which indicated that OSHA needed to determine significant risk existed for each substance, as required by the "Benzene" decision (Ashford 2000). In other words, OSHA needed to perform individual risk assessments on over 400 chemicals. These legal interpretations of OSHA's authority have severely constrained OSHA's ability to issue exposure limits to protect worker health.

OSHA has also adhered to a 1 in 1000 acceptable risk level compared to EPA's 1 in 1,000,000 acceptable risk level to trigger regulatory action. Most PEL's today still reflect the 1968 ACGIH values. These crucial court cases and policy decision rules meant the practice of risk assessment to protect human health and the environmental had now been opened to public and judicial critique. The science that informs the practice of risk assessment was also often critiqued, in what Fischer (2000) describes as an emerging politics of expertise and counterexpertise. In the next sections, I will more fully discuss the traditional risk assessment/ risk management paradigm outlined by the NRC (1983), and

1.2.3 Risk Assessment vs. Risk Management: The 1983 Red Book Approach

An emerging theme from the above analysis is the prominence of quantitative risk assessment in agency decision making and the role of science in the risk assessment process. According to Jasanoff (1986), after the "Benzene" decision and publication of the NRC (1983) report, agencies like OSHA and EPA almost immediately incorporated the NRC's (1983) recommendations into their rule-making practices. In the NRC's (1983) risk assessment/risk management paradigm, risk assessment consists of four steps: hazard identification, dose-response assessment, exposure assessment and risk characterization. The output of the risk characterization is typically a quantitative estimate of risk, such as the excess risk of cancer that may result from inhaling a chemical at a specified concentration.

Various scientific methodologies can be used to develop a risk assessment, including but not limited to epidemiology, toxicology, environmental science, statistics, industrial hygiene, and environmental engineering. While risk assessment may determine a quantitative estimate of risk, it does not determine whether that risk level is acceptable. Acceptability is considered the domain of risk management. Risk management refers to the evaluation of regulatory options to control risk, which includes the identification of associated public health, economic, social, and political consequences (NRC 1983).

Experts in risk assessment often relied on "uniform guidelines" to standardize "judgments", ultimately communicating risk estimates to the agency risk manager, who would develop and evaluate regulatory options.

During the risk management phase, values associated with various options would be considered. Options would be deliberated by experts, with public participation where required by law. Although the NRC (1983) did recommend communication between assessment and management functions, in practice risk assessment and risk management became essentially divided.

Risk assessment came to be seen as embodying more of the "science or facts" and risk management came to be seen more as the "policy or values" part of the decision-making process. Although the traditional paradigm frames risk assessment as a scientific process and risk management as the policy-oriented dimension of decision-making, in practice the two are very much intertwined. Risk assessment over the past thirty years has become institutionalized in EPA and OSHA. In numerous cases – such as the proposed ban on urea formaldehyde foam insulation - judicial review has emphasized the need for risk assessment and even critiqued agency risk assessment results (Graham 1995). The tangled relationship between risk assessment and risk management has resulted in multiple controversies and public erosion of trust in agency decision making.

Jasanoff (1986) describes the controversy over EPA's risk assessment of formaldehyde as prototypical of problems created by the facts vs. values dichotomy. In the early 1980's, an industry sponsored study showing a connection between formaldehyde exposure and increased risk of nasal cancer in rats prompted EPA to recommend a priority review under the Toxic Substance Control Act. While the rat data was considered reliable, and the doses used in the study comparable to human exposures, the available epidemiological evidence in humans was considered less certain, due to a lack of nasal cancer cases noted in human populations (although other cancers were noted). Industry scientists argued that the nasal cancer results observed in the rat study were specific only to rats, and not expected to occur in humans. The technical arguments and counterarguments revolving around EPA's risk assessment of formaldehyde ultimately led to the agency's reversal of a decision to more stringently evaluate formaldehyde's toxicity and prevalence of human exposure (Jasanoff 1991). Scientific uncertainty was exploited in a competing fashion by different experts to influence policy – pro-regulation scientists supported the rat studies as sufficiently conclusive to regulate, and pro-industry scientists argued regulation was premature as the data was too uncertain.

When viewed through the above lens, the diesel exhaust controversy shares many similarities with the formaldehyde case. While the animal studies indicated high concentrations of diesel exhaust can cause lung tumors in rats, EPA (2002b) pointed out the lung overload response observed in rats was not expected to occur in humans at environmental or occupational exposure levels. Similar to the formaldehyde risk assessment process, the diesel exhaust epidemiological studies were considered weaker and less reliable, due to issues of uncertainty. There have also been other technical issues: the Health Effects Institute's (2002) comprehensive report on risk from diesel exhaust expressed concern with both the methodological uncertainty associated with existing and proposed exposure assessments and the lack of an identifiable, specific diesel signature. While many scientists have argued for more regulation to reduce the health risk from diesel exhaust (Decker et al. 2003; Wargo et. al. 2001; Treadwell 2005), ultimately the regulatory approach has been cautious and incremental. For both the formaldehyde and diesel exhaust cases, scientific uncertainty in risk assessment appears to be a key point of political and scientific conflict in the risk decision-making process. Depending on one's worldview, scientific uncertainty can be used as an argument to either increase or postpone regulation of chemical exposures.

1.2.4 The Epistemiological Dimension: Policy vs. Normal Science

The appropriate role of science in risk decision-making and how to handle scientific uncertainty continues to challenge policy makers, agency experts, researchers and the public. Jasanoff (1986, 1991) states many risk controversies occur in the U.S. as a result of the desire to eliminate uncertainty by further refinement of quantitative techniques. As EPA has to justify its decision to both the public and regulated entities, risk policy has evolved to emphasize risk numbers upon which to base decisions. Yet, risk assessment debates can allow new kinds of uncertainty to come to the forefront, as shown in the formaldehyde case.

In the diesel exhaust case, the desire to incorporate evolving science to reduce uncertainty led to extensive delay and limited regulatory action. Ultimately, additional science did not resolve the contentious issues in both cases, but instead just brought more or new technical issues into the deliberations. These examples lay bare the policy conundrum of wanting a scientific basis for a policy decision, but coming up against the realization that not all questions are capable of being answered by science. Even if science determines an answer, often scientific inquiry creates new, relevant questions.

Part of the debate regarding the implications of scientific uncertainty may have more to do with competing epistemological understandings of science. “Mainstream” or “normal” science adheres to a reductionist philosophy that assumes systems can be taken apart, studied, and then put back together (Ravetz 2004). This idea of science builds on Kuhn’s (1970) description of “normal” scientific research as a puzzle solving activity, intending to add to the foundation of existing scientific knowledge. Mayo (1991) asserts adherents to “normal” science believe that pure, value-free science exists as a kind of ultimate truth.

Personal values must be kept separate from the objective fact-finding process of scientific investigation. Via this epistemology, one uses science to pursue a solution to the policy problem, believing that with enough research, a “best” solution will emerge from among alternatives. In both the formaldehyde and diesel exhaust cases, “normal” science did help make progress on total understanding of the exposure risk, but this progress was incremental, slow, and resulting regulatory action considered insufficient. “Normal” science is by its nature slow and incremental – but policy science needs facts quickly because decisions are often urgent, and policy makers regularly must make decisions without the desired ideal level of understanding.

Normal science is challenged by a social constructivist view of science in which facts and values interact (Fischer 2000). This viewpoint suggests science and policy are interconnected in ways not immediately obvious, even to scientists. Examples of science/policy interaction include when scientists decide to use certain statistical tests of significance, or the process of peer review. Science does not occur in a vacuum, segregated from the problem, nor is one “true” or “best” solution emphasized. While science is acknowledged as necessary to inform the policy process, the decision-maker at some point must cut the “knot of uncertainty” and the decision may not be improved by more quantitative analysis (Jasanoff 1991). Science by itself cannot solve many policy dilemmas simply because reasonable people (including scientists) disagree how to interpret information as well as decide which information is most important in making decisions (Stern 2005).

In closing, traditional risk decision making views science via a “normal” science lens, separate from policy, or that “science = facts” and “policy = values”. The “facts” vs. “values” separation is comparable to the separation of risk assessment and risk management functions that has taken root in institutional cultures here in the U.S. (Jasanoff 1986; 1991). Attempting to separate science and policy by adhering to the “facts vs. values” dichotomy perpetuates a politics of expertise vs. counterexpertise (Fischer 2000). Yet the scientific method is itself a social process: scientific “facts” emerge often after a complex process of formal and informal peer review. Peer review, in essence, debates facts, because there is no one objective standard of “good” science. Since scientific expertise is thus interpreted, technical or expert judgment should not be the sole basis of policy decisions (Fischer 2000).

In summary, the regulatory, institutional and epistemological barriers outlined in this essay are formidable. Looking at the barriers separately invites speculation on regulatory or institutional solutions. But the cases in this chapter show that it is highly unlikely institutional or regulatory solutions will advance how scientific uncertainty is addressed in contemporary risk decision-making processes. Although not emphasized thus far, there are other uncertainties equally as challenging to risk decision-making as scientific uncertainty. For example, competing stakeholder and public values will also impact the risk decision- making process. Additionally, there are uncertainties in the level of trust stakeholders and citizens may have in regulatory institutions. Rayner and Cantor (1987) suggest that the conflict surrounding many risk management decisions has more to do with the lack of attention paid to issues of equity, trust and liability than issues of certainty of the estimates of probability of harm. Novel approaches to risk decision-making are needed to address these multiple dimensions.

1.3How Risk Decision-Making has Changed: Moving from the NRC (1983) to the NRC (1996) Report

By the 1990’s, it became clear new approaches to risk decision making were needed.

Many scientists and environmental advocates had become frustrated with quantitative risk assessment’s role in risk decision making. Some even considered risk assessment “ethically repugnant” and anti-democratic as it allows people to be exposed to toxic substances against their will, and legitimizes premeditated murder via chemical exposure (O’Brien 1997). Various calls for risk reform were made. Some critics of risk policy-making argued more broadly implemented cost/benefit analysis techniques could best guide regulatory agencies (Sunstein 2002). Others suggested a focus on democratic rather than technocratic improvements by expanding citizen participation in environmental decision making (Fischer 2000; Renn et al. 1995).

One view of policy-making is that policy emerges from shared understandings or knowledge. The critiques identified above may highlight the frustration with quantitative risk assessment (QRA), but it is arguably how risk assessment is used in decision-making that is at the root of the frustration. Ozonoff (1998:49) summarizes this view clearly: What gets environmentalists riled up about QRA has little to do with its use as an assessment device, but its use as a decision justification device. The agency/industry/policy maker has shot the arrow, and the risk assessment obligingly paints the target around it, preferably with sophisticated paint using an abundance of integral signs and capital sigmas to make it look infallible.

Fischer (2000) has recommended approaches to policy-making that incorporate a constructivist understanding of knowledge with a deliberative framework that reflects both scientific inquiry and local knowledge in an “evolving conversation.” Facts and values should not be kept artificially separate, and citizens and technical experts should work together. Improving risk decision-making in general - and integrating environmental and occupational health risk management more specifically - requires increased attention to the initial problem formulation stages, as well as ways to incorporate changes in understanding. One promising model that may lead to more informed risk decision-making is the NRC (1996) analytic-deliberative (A-D) model, which will be reviewed next.

1.3.1 Detailed Description of the A-D Framework

In the 1980’s and through the 1990’s, quantitative risk assessment had become the predominant frame for U.S. regulatory policy-making managing chemical exposure risk in the workplace and environment. However, the NRC (1996) acknowledged a fundamental deficit in the final risk characterization step in the QRA process: its emphasis on accurate translation of risk numbers for policy makers at the expense of missing the broader decision context and public concerns. The risk characterization step's focus on numbers and risk communication efforts to educate the public led to agency decisions – such as those regarding cleanup actions at contaminated hazardous waste sites – that resulted in controversy, public outrage, litigation, and overall increased public mistrust of agency decision-making processes. Yet the NRC committee realized during its work that the core issue was not improving QRA as a tool but how to best inform risk decision-making in a way that reflected the multidimensional nature of risk (Stern 1998). The scope of the problem was broader than deficiencies in one analytic tool.

Recognizing risk characterization as a complex nexus of science and judgment, the National Research Council (1996) undertook a broader look at this step and recommended that risk characterization be reconceptualized as decision-driven activity oriented towards solving problems. Risk characterization is performed via an iterative process of analysis and deliberation. Analysis refers to the use of "rigorous, replicable" methods from a wide variety of disciplines such as the physical sciences, law and mathematics to "arrive at answers to factual questions" (NRC 1996 p. 3 - 4). Deliberation refers to "formal or informal" communication processes where participants "discuss, ponder, exchange observations and views, reflect upon information and judgments…and attempt to persuade each other" as typical in consideration of issues of collective interest (NRC 1996, p.4). The NRC (1996) is careful to point out that the concept of "deliberation" is broader than "public participation" as it focuses on improving the understanding of a risk situation, especially in its initial stages preceding agency action.

In the NRC (1996) conceptualization, there is no separation of assessment and management functions, analytic-deliberative processes may vary at each step, and participation in any step may include scientists, public officials, and interested and affected parties. The benefits of this new approach are the anticipated improved quality and acceptability of the final decision.

The attention given to the problem formulation stage is significant: comprehensive diagnostic questions are suggested to survey the risk decision landscape to ensure the knowledge base is as complete as possible and issues (like legislative mandates that may constrain agency decision-making in practice) are identified early. A key point of the NRC (1996) report is that interested and affected parties as well as experts should also be part of deliberative processes that occur in the early problem definition stage, when the risk problem is being defined or diagnosed, to help direct performance of necessary analysis. This focus on the problem formulation stage – the stage where risk is defined and knowledge gaps identified – and the recursive nature of the interaction between analysis and deliberation appear especially well suited to the goal of defining occupational and environmental health risks concurrently. This made the A-D model attractive for application to this study.

The next step is process design, or the identification of interested and affected parties and how participation will occur. Deliberative processes should be broadly based, involving not only decision-makers or experts but also interested and affected parties. In arguing for inclusion of interested and affected parties in analysis and deliberation, the NRC (1996) refers to Fiorino's (1990) three rationales justifying broadly based public participation in risk decision-making: normative, substantive, and instrumental. The normative rationale refers to the rights of citizens in a democratic society to participate in governmental decisions that may affect them. The substantive rationale explains that experts do not have exclusive domain over knowledge relating to a risk decision. The instrumental rationale for participation emphasizes the potential to legitimize agency regulatory decisions. Ideally, increasing the legitimacy of decisions would reduce conflict and controversy.

Since a wide literature already existed on analytic techniques, the NRC (1996) report focused on drawing out the role of deliberation. But understanding how to "do" deliberation, and do it well, remains a key challenge today. There is limited knowledge about how best to integrate analysis and deliberation. How to deliberate, who to involve, and what should be deliberated remain critical questions. While the attributes of various deliberative processes, such as citizen advisory boards and public hearings are discussed in the report, the NRC (1996) does not specify which types of risk problems should be matched with which deliberative processes. Instead, the NRC (1996) suggests an analytic-deliberative framework should meet the following objectives: getting the science right, getting the right science, getting the right participation, getting the participation right, and developing an accurate, balanced, and informative synthesis of the risk scenario. These criteria are meant to guide the analytic-deliberation processes that inform the overall risk decision making process.

While helpful to point policy-makers in the right direction, these criteria are relatively vague and may not be especially helpful for any given risk decision. From a practical standpoint, regulatory agencies and participating organizations need "how to" guidance to be able to increase the quantity and quality of deliberative processes.

For deliberative processes may hold promise to improve risk decision making, but there are also numerous challenges. First, opening the decision-making process up to interested and affected parties in early stages requires a commitment of time and resources that can significantly delay a decision. Second, making participation more "open" does not necessarily mean an equal playing field between participants, especially when there is a discrepancy in technical expertise. As Fischer (2000) makes clear, whenever discussions take place on experts' "intellectual turf", citizens are disadvantaged in the debate. Unequal power dynamics can add fuel to the fire of a controversial decision situation. Third, there are important ethical considerations that become apparent in expanding deliberations. U.S. society is made up of numerous value systems and worldviews, challenging risk managers in how to determine whose values to select as legitimate (Renn 1999). While acknowledging citizens can bring important knowledge to bear on a risk decision, technical expertise is still a necessary component in the evaluation of hazards. Finally, recommendations resulting from deliberation may still be rejected by the ultimate decision-maker, consensus may not be attainable via deliberative processes, and legal mandates may prescribe certain agency actions regardless of the views of interested and affected parties (NRC 1996). In short, broadly based deliberation can be expensive, time intensive, ethically charged, and offers no guarantee of success. In fact, success in itself can be a difficult variable to define.

While there is no cookbook formula to match deliberative processes to specific types of risk decisions, there is a body of literature that can be reviewed to help guide those interested in implementing participatory processes. Chess (2000)'s review of recent case studies guides environmental health professionals in how to "get the participation right" when involving the public in environmental decision-making. Successful participation can be defined by participants in different ways: consensus, reaching a desired decision outcome (i.e., accept or reject an agency proposal), improvement in environmental quality, an evaluation of the participatory process itself, or some combination thereof (Chess 2000).

Similar to the NRC (1996) report, Chess (2000) emphasizes that evaluation and feedback of the process are important, and participation processes may need to be adapted in response to this feedback. Additional critical process design considerations include transparency, giving participants ownership of the process, creating a "safe" setting for dialogue, and creating a process where people feel like they can make a difference (Webler and Tuler 1999).

Deliberation is also critical in the next step in the A-D model: selection of options and outcomes. Webler and Tuler (1999) explain that selecting management outcomes and options gets at a number of key questions in the decision-making process: what do people care about, what should people care about, and what are good indicators for characterizing and ranking problems, options and outcomes? Deliberation about these criteria may identify the need for more analysis. In suggesting how this can happen in watershed management planning, Webler and Tuler (1999) explain that selection of a management option like tax breaks to prevent extensive shoreline development may trigger the need for an economic feasibility analysis. Analysis and deliberation feed into each other, directing future steps and action.

The development of options and outcomes requires the need to gather and interpret information. This is the next step in the A-D model, the place where analysis as conceptualized under a "normal science" paradigm is often located. In order to assess the viability of options and outcomes, data are needed. For example, in trying to establish the health risk from a chemical exposure at a hazardous waste site, health effects data from animal toxicology or epidemiological studies are traditionally reviewed. Yet, other types of analytical data may also be useful: other techniques to gather health effects data include worker health surveys or focus groups of affected community members. Affected parties may feel it is critical to gather their own health data as the local context may be unique or poorly researched. Corburn (2005) cites an example of an EPA health risk assessment in Brooklyn that overlooks the impact of subsistence fishing from polluted waters on a typical urban diet.

These types of research projects on health and exposure risk have traditionally been the domain of technical experts. Experts feed research results into deliberation processes regarding which options and outcomes are appropriate or if new ones are needed. Participation mechanisms like citizen advisory councils or other ad-hoc panels rely heavily on outside presentations of scientific data to inform their decision. Some researchers have critiqued the privileged role of technical expertise in gathering information to inform deliberative processes. A focus on deliberation of data primarily provided by scientific experts results in limited opportunities for the public to participate in activities that influence the analytic process (Judd et al. 2005). Fischer (2000) also critiques the NRC's (1996) focus on deliberation as leaving science squarely in the domain of experts, diminishing nonexpert participation in analysis. Since the NRC (1996)'s report adheres to a positivist (or "normal") conception of science, Fischer (2000) argues that scientific evidence remains the preferred type of evidence in environmental decision-making, and current institutional structures limit citizen involvement mainly to deliberation, not analysis.

I highlight these critiques at this point because this study had a community participation focus that attempted to expand and extend the idea of analysis beyond normal science. Other researchers have also recently begun using an expanded A-D framework to solve environmental problems. While most cases in the literature have focused on citizen participation in environmental decision-making, there are a small but growing number of cases where citizens have worked more actively within analysis as well as deliberation. Judd et al. (2005) applied the A-D model to increase community deliberations to frame scientific analysis in three cases. In each case, health risks related to chemically contaminated seafood were a major concern to the local community. Prior to the research, the typical way the risk of contaminated seafood was managed in the community was the issuance of fish advisories – a one way risk communication process. Many questioned the effectiveness of fish advisories due to language barriers. Another critique was that this process did not provide any feedback for safe management of contaminated fisheries. Researchers and community organizations worked together to come up with ways to better understand local consumption patterns of contaminated seafood, both from community markets and subsistence fishing, and helped set up local monitoring capability. While each case had a unique context, researcher and community collaboration led to similar benefits: enhanced research that met the needs of the community, community performance of the analysis and interpretation of data, better understanding of exposure risk, and building capacity among tribal groups to do their own risk management (Judd et al. 2005). A key result in each case was that community framing and participation in scientific activities led to better characterizations of risk from contaminated seafood (Judd et al 2005). The data collected was more easily integrated and synthesized into local decision-making process as well as associated educational processes due to the enhanced legitimacy that resulted from community participation.

Synthesis of information is the last step in the NRC (1996) A-D framework. The gathering of information step and the synthesis of information are closely related. This synthesis can take many forms: quantitative or qualitative, policy recommendation or management plan, recommendation for regulation or educational programs. As in the other steps, analysis and deliberation interact and the synthesis of information to address an initial problem may naturally lead to new problem formulations. For example, a watershed management plan would be the synthesis product from a watershed management process, but this process - and the associated plan - will likely evolve over time as conditions change.

Webler and Tuler (1999) recommend that the final synthesis documents the uncertainties, assumptions, and information in a way accessible to interested and affected parties.

The previous explication of the A-D framework shows how the thinking regarding risk decision making has progressed since the 1970's and 1980's. Compared to the NRC (1983) risk assessment/risk management paradigm, the NRC (1996) report represents a more flexible and collaborative approach to risk decision-making. The A-D approach is detailed enough to provide guidance yet open and adaptive enough to be suitable to a number of environmental applications at the federal, regional, and local level. At a theoretical level, the NRC (1996) report is important and noteworthy because it provides a way to replace the traditional facts/values and science/policy dichotomy with a framework that is more consistent with how people actually make decisions (Webler 1998). Scientists and policy-makers each do analysis and deliberation naturally but just might not do it reflectively. For example, the scientific research process emphasizes objectivity in the discovery and analysis of facts, but the process also requires deliberation: scientists analyze facts, but often deliberate these facts at conferences and in other forums like peer reviewed articles. Another key contribution of the NRC (1996) report is highlighting how analysis includes more than traditional quantitative risk assessment or scientific hypothesis testing and deliberation includes more than traditional public participation mechanisms (Webler 1998). This broader conceptualization of analysis and deliberation is especially important when local knowledge may offer significant insight into environmental problem solving. The NRC (1996) report acknowledges that different ways of knowing should be respected and integrated to best inform decision making.

1.3.2 How the A-D Framework Can Be a Good Fit for the Problem of Diesel Exhaust

The above cases and review of the A-D model formed a rationale or basis for selection and application in this study. The problem of diesel exhaust is significant, and at a federal level, agency action to reduce exposures and associated health risk is limited or moving forward glacially at best. There is no federal action to prevent workplace exposures to whole diesel exhaust. The regulatory examination and evaluation of diesel exhaust risk (EPA 2002a) has mainly followed the NRC (1983) traditional paradigm. The Health Assessment Document followed this 4 step risk assessment process. EPA's regulatory approach with its emphasis on risk assessment vs. risk management has become relatively stuck on the point of scientific uncertainty regarding animal and human health studies. One could argue enough science has been done and the regulatory decisions have been motivated by politics and not existing scientific evidence.

However, the NRC (1983) risk assessment/risk management process is not well suited to the complexity of the diesel exhaust problem such as the evolving technology, widespread use, and variability in application of diesel engines. The multidimensionality of the problem of diesel exhaust exposes the weaknesses of the traditional paradigm. There are also multiple scales of exposure that overlap: workplace, community, regional and national. While public concern is somewhat limited, many environmental/occupational health scientists, and EPA itself on its website, recognize the significant contribution of diesel exhaust to ambient levels of air pollution and local elevated levels in the workplace. The known negative health effects of components of diesel exhaust – such as fine particulate matter - are substantial. Emerging knowledge supports that other components have their own unique health hazards. A new approach to the problem of diesel exhaust outside the traditional paradigm is needed.

The A-D framework presents one possible approach to understanding risk and one suitable to the unique local context of this study. This study applied the analytic-deliberative (A-D) model to a collaborative exposure assessment research project that evaluated the impact of biodiesel fuel – as a risk reduction alternative to petroleum diesel – on environmental and occupational exposures. Biodiesel use is growing in popularity in the U.S. for a number of reasons which will be discussed below. My research interest was the potential of biodiesel as a risk reduction intervention to reduce exposures to petroleum diesel emissions such as particulate matter, EC/OC, and nitrogen dioxide in both the workplace and local environment. Instead of following a more traditional risk assessment approach to inform development of a biodiesel potency estimate, I was interested in performing a real world, comparative study to assess the concurrent impact of switching to a 20% biodiesel blend (B20) on both occupational and environmental exposures. My initial research questions were inspired and informed by observations from the community and informal conversations with both City of Keene and Keene State College employees that indicated dramatic improvements in workplace air occurred soon after biodiesel was introduced in local fleets. I worked with these community members, technical experts and students from KSC to develop and implement a collaborative exposure assessment, an analytic process that measures levels of air contaminants in workplace and local ambient air. To connect analysis with deliberation I also organized and set up a local Biodiesel Working Group as a deliberative forum for dialogue, information exchange, and a place for analysis and deliberation to interact. More detail on the specific research questions and application of the A-D model to this study will be reviewed in Section 1.6.3. First I will discuss the basics of biodiesel and why it is considered a green alternative to diesel. In the next sections, I provide a brief background on biodiesel, its potential as an alternative to diesel fuel, and review the literature on biodiesel emissions, exposures and associated health impacts.

1.4 Introducing Biodiesel

1.4.1 Biodiesel: What Is It? How's It Made? Who's Using It?

Biodiesel is an alternative fuel made from vegetable oil, animal fat, or waste grease. While relatively recent in the U.S., biodiesel has been widely available and used in western European countries such as Germany for at least the last 10-15 years (Pahl 2005). In contrast to the US close to half of the European passenger vehicle fleet utilizes diesel engines. Over 1,900 public filling stations in Germany currently offer biodiesel, and officials there believe national biodiesel production capacity could displace almost 12% of that country's petroleum diesel by the end of 2008 (Bockey 2005). In the U.S., there are about 800 retail pumps nationwide, and 11 in New Hampshire (NBB

2008).

While rapeseed is the primary feedstock for German-made biodiesel, the most popular feedstock in the U.S. is soybean oil (Pahl 2005). Since the soybeans that make up this virgin oil feedstock are grown domestically, biodiesel is often referred to as a sustainable or renewable fuel. Researchers in the U.S. are examining other feedstocks such as mustard seed, rapeseed and even algae to increase oil yield and opportunity for farmers and other oil producers to enter into the biodiesel economy (Pahl 2005). Biodiesel is not the chemical equivalent to pure vegetable oil or grease; rather it is the mono-alkyl esters that remain after oil or grease undergoes a transesterification reaction.

Most biodiesel in the U.S. is made via base catalyzed transesterfication (Pahl 2005). In this chemical process, oil or grease is reacted with methanol (or ethanol) in the presence of a sodium hydroxide (or potassium hydroxide) catalyst to make mono alkyl esters (biodiesel) and glycerine as a by-product. When 100 pounds of oil are mixed with 10 pounds of methanol (plus necessary catalyst) approximately 100 pounds of biodiesel and 10 pounds of glycerine are produced (DOE 2004). Although this process is the most common in the U.S., there are other methods of biodiesel production, such as acid catalyzed transesterification, and research continues into new, more efficient methods to manufacture biodiesel from various feedstocks.

In terms of physical characteristics of the fuels, biodiesel and diesel fuel differ in many respects. Biodiesel has a higher cetane number than petroleum diesel fuel. The cetane number is a measure of a fuel's ability to autoignite. A higher cetane value is preferred in compression-ignition engines as this indicates the fuel will ignite more quickly. Other key differences: biodiesel has a higher boiling point and flash point than diesel, which means it is safer to transport as it is even less likely to combust than diesel. However, B100 has significant cold weather problems due to its high cloud point (or the temperature at which the fuel begins to cloud or crystals appear). B100 will start to cloud at around 36 ºF and will begin to gel at 28 ºF (DOE 2004). This limits B100's suitability in colder areas of the U.S. As a result, in the U.S. marketplace, diesel is often added to biodiesel. B20 blends have cloud and gel points almost identical to 100% petroleum diesel blends for similar performance in winter climates. Most biodiesel in the US is sold as B20 or a 20% soybased biodiesel and 80% petroleum diesel blend (DOE 2002). BXX is used to refer to the percentage of biodiesel in the blend; B10 would equal 10% biodiesel and 90% petroleum diesel.

Many U.S. organizations interested in a renewable and domestic source of energy are considering switching from 100% petroleum diesel to biodiesel/petroleum diesel blends for transportation and heavy-duty equipment use. According to the National Biodiesel Board, over 800 fleets in the United States are using biodiesel blends (NBB 2008). These fleets include municipal and government fleets located across the country, such as public works vehicles in the city of San Franscisco, CA and the city of Keene, NH. School buses from Medford, NJ to Clark County, NV run on B20 (NBB 2008).

The volume of biodiesel consumed nationwide is steadily increasing. Approximately 200 million gallons of biodiesel blended fuel were sold in 2006, and one blue-sky scenario predicts 1.5 billion gallons production capacity for 2007 (Schmidt 2007). Although the U.S. consumed more than 40 billion gallons of petroleum diesel in 2005 alone, some experts believe biodiesel could someday displace up to 25% of the current volumes of diesel fuel used in the U.S. (Schmidt 2007). The use of biodiesel is expected to continue to rise.

Cost is another key area where diesel and biodiesel differ. Petroleum markets continue to be widely volatile, making price comparisons between B20 and 100% petroleum diesel difficult. There are also tax subsidies supporting biodiesel at the federal and state levels which may or may not be reflected in the final price at the pump. Howerever, B20 blends are typically more expensive than petro-diesel, varying between 5 to 20 cents more per gallon. At the end of 2005, B20 blends averaged 10 cents more per gallon, and B100 blends averaged 59 cents more per gallon (Methanol Institute/International Fuel Quality Center 2006). This differential cost may be a key deterrent in market expansion of pure biodiesel. The lower cost differential and similar cold weather properties of B20 to diesel may help explain why B20 is the most popular blend in the U.S.

1.4.2 Advantages of Biodiesel

1.4.2.a Biodiesel as an Alternative to Petroleum

A key benefit of biodiesel is that no major engine modifications are necessary to existing diesel engines prior to use. The only recommended adjustment is replacement of rubber seals with synthetic materials in pre-1993 fuel systems if B100 is used as B100 has solvent properties that can degrade pure rubber (DOE 2002). Biodiesel, especially B20 blends, can be immediately introduced into existing distribution infrastructures and diesel engine applications. There are numerous case histories (such as from the municipal fleet in Keene, NH) testifying to smooth and beneficial integration into existing fleets. Although some documentation indicates biodiesel use will result in lower miles per gallon (DOE 2002), others report B20 use resulted in increased mileage efficiency. Wayne Hettler, Head Mechanic of St. Johns Public Schools, St. Johns, Michigan reports:

We have experienced very positive results with B20…We now extend our oil services another 10 percent. Our buses don't have the exhaust soot on the back that needs to be scrubbed off. The fleet average fuel mileage has increased from 8.1 to 8.8 miles per gallon. When all of these things are added up, we are seeing about $7500 savings per year. When we take out the cost difference in the price of the B20, we still see about $3000 per year savings (USDA, undated publication).

Biodiesel offers a number of political, economic, and operational benefits. A fuel that can be domestically sourced is politically attractive. The growth of the biodiesel industry has resulted in new jobs and new revenues for soybean farmers, who for many years had a glut of surplus soybean oil (Pahl 2005). Biodiesel fuel is also biodegradable, low toxicity, and has high lubricity characteristics which may help extend engine life (DOE 2004). Biodiesel also has key industry support: most diesel engine manufacturers will not void warranties for burning up to a B20 blend as long as the fuel is ASTM (American Society for Testing and Materials) certified (Pahl 2005). Biodiesel has a slight solvent effect, cleaning out engine deposits – but this may help improve engine performance. At the same time, biodiesel increases lubricity in the engine compared to diesel fuel. This can have enormous benefit as sulfur content, the traditional lubricant in petroleum diesel, has been recently reduced in EPA mandated ultra low sulfur diesel fuel. The combination of cleaning and lubricity benefits can extend engine life. Adding just low levels or 1 to 2% biodiesel to ULSD is expected to improve overall lubricity (DOE 2004).

Biodiesel has a number of environmental benefits in addition to low toxicity that make it an attractive alternative to petroleum diesel. Compared to petroleum diesel use, biodiesel is more energy efficient, and reduces net carbon dioxide emissions. A joint study performed by the United States Department of Agriculture and the United States Department of Energy determined that over its life cycle of production and use, biodiesel yields 3.2 units of fuel product energy for every unit of fossil fuel energy that goes into making it (Sheehan et al. 1998). By contrast, petroleum diesel has a ratio of 0.83 units of fuel product energy yield per unit of fossil fuel energy consumed, or a net loss of energy over its entire life cycle. Another way of understanding this relationship is that, on a per gallon basis, soy based biodiesel provides 69% more energy than the fossil fuel energy that went into making it. The same study also found that use of soybean-based 100% biodiesel in an urban bus reduced net carbon dioxide emissions by 78% and B20 reduced CO2 by almost 16% (Sheehan, et al. 1998). Hill et al. (2006) performed a more recent life cycle accounting and determined that soy based biodiesel provides 93% more energy than the fossil fuel energy invested in its production, and reduces greenhouse gases by 41% compared to diesel (Hill et al. 2006).

Additional benefits of biodiesel relate to human health and the environment. Burning biodiesel vs. petroleum diesel results in reduced tailpipe emissions of carbon monoxide, particulate matter, and hydrocarbons (EPA 2002b). These reductions are shown in Table 1.1 below. B20 use results in an average 10% reduction in particulate matter (less than 10 micron diameter) but a corresponding average 2 percent increase in NOx (EPA 2002b). In the next sections I will review the environmental benefits as reported by two fleets and review the scientific literature on biodiesel emissions studies.

1.4.2.bIs Biodiesel a Carbon-Neutral or Carbon-Reduced Fuel? Stories from the Field

An examination of the biodiesel policy discourse identifies a number of political, economic, and health (both human health and environmental health) arguments driving increased biodiesel use. The political argument focuses on the domestic production of biodiesel as a way to lessen U.S. dependence on foreign petroleum imports. The economic argument states an increase in domestic production of biodiesel fuel would lead to an increase in U.S. jobs and a stronger economy. The human health-based argument points to existing scientific evidence indicating burning biodiesel fuel may present less risk to the environment and human health. Finally there is an argument for the environmental benefits suggested by widespread use of biodiesel as a renewable, plant based fuel. These benefits include reducing carbon in the form of carbon dioxide released into the atmosphere. Since biodiesel is made from plant sources, these plants can capture carbon dioxide during the cycle where feedstock plants are grown. Use of waste grease for making biodiesel fuel is even more beneficial, as the feedstock is a waste, but the pure oil used in cooking was initially made from plant materials.

For these reasons and others, many cities are adopting biodiesel as a way to improve environmental quality and reduce their overall carbon footprint. In the paragraphs that follow, I will discuss two city's stories: San Francisco, CA and Keene, NH.

In 2006, Mayor Gavin Newsom of San Francisco issued an executive directive that all municipal diesel vehicles use B20 by the end of 2007 as part of a city wide effort to reduce petroleum consumption, improve air quality, and reduce greenhouse gases (Newson 2006). This directive also initiated a Biodiesel Task Force to streamline regulations and encourage private sector biodiesel use. At the end of 2007, all of the City's 1500 diesel vehicles were powered by B20, making it one of the nation's largest green fleets (Marshall 2007). This equates to a displacement of approximately 1.2 million gallons of diesel fuel per year. In addition to use of biodiesel, San Francisco's Public Utilities Commission is setting up a program to collect waste grease from restaurants for free and sell this material for processing to local biodiesel manufacturers. City officials believe this could be a win-win for the restaurants and the City, because dumping of waste grease is a problem in local sewers, and costs the City $3.5 million a year to clear grease blockages in sewer lines (Cohen 2007). Since the City of San Francisco also uses B20 in its fleets, the hope is to move from using soy-based B20 to waste grease-based B20.

In the City of Keene, NH, the story behind the use of biodiesel is similar yet unique. Since the City of Keene's relationship with biodiesel provides important background for this study, I will present the local biodiesel story in more detail. Keene is a small city of approximately 22,000 people located in southwestern New Hampshire. With respect to environmental awareness, Keene could be considered a community more concerned about protection of the environment than most. In 2000, Keene signed the Cities for Climate Protection Campaign, administered by the International Council for Local Environmental Initiatives (City of Keene 2007). The Cities for Climate Protection (CCP) Campaign focuses on local solutions to global warming, primarily by reducing emissions of greenhouse gases at the municipal level. Keene has signed on to reduce emissions of carbon dioxide and methane by 10% of 1995 levels by 2015, but the City municipal departments have committed to a 20% goal. To meet this goal, a number of environmental projects have been initiated, such as installing a methane recovery system at the local landfill, and implementing energy conservation measures in municipal buildings. Although biodiesel use is listed on the City's 2004 Local Action Plan (City of Keene 2007), the decision to use biodiesel happened concurrently and outside the formal CCP process, at least initially (Russell 2006).

The initial decision to use biodiesel in the City of Keene fleet originated with Department of Public Works Fleet Manager Steve Russell. Others interviewed as part of this study all point to Russell as being the critical component of the decision to use B20 in Keene. As Duncan Watson, Assistant Director of Public Works, and currently Russell's supervisor, puts it, "Steve Russell really took the initiative to get biodiesel into the fleet. Steve was the primary driver on this." (Watson 2006). Russell himself has acknowledged becoming a kind of biodiesel expert in the area, "I guess I'm the biodiesel king" (Cleary 2005). The city has been using B20 in its fleet since 2002.

However, there were a number of key steps in the decision that happened before B20 was finally implemented. In 2001, Russell attended a Granite State Clean Cities meeting at Antioch New England Graduate School (now Antioch University) where the question of biodiesel use came up. At the meeting, he offered to try the alternative fuel in his municipal fleet, but stated his budget could not allow for the extra 35 cents per gallon cost for B20. The next day he received a call from the New Hampshire Governor's Office of Energy offering a small $2500 grant to offset the cost differential to purchase B20. At that point, Russell recalls, "I started doing my homework" (Russell 2006). He developed a list of biodiesel's positives and negatives, particularly warranty issues. At the time, some engine manufacturers were taking a negative stance towards biodiesel, stating that use of the fuel could void the warranty. This meant that any problems with an engine subsequent to trying the fuel could be challenged. However, Russell researched the language in the engine warranties in his fleet and determined that engine warranties specifically cover workmanship of parts. If he used a quality certified biodiesel fuel the engine manufacturers had to stand by their commitment to correct any engine defects.

Yet, instead of immediately placing the order for a B20 delivery, Russell spent the next six months meeting with department heads across the City's organization in a long process of education and advocacy to address concerns and build support to try the fuel. When the $2500 from the initial grant ran out, Russell kept using B20 in the fleet, wondering if this would result in problems for him later:

I kept it going for a while, and then I thought when my budget goes over, and they start asking questions, I am going to be in trouble. I said, I'll take the chance. I noticed it was doing good things for the fleet. I noticed the air was cleaner, the mechanics noticed it. There were a lot of positives (Russell 2006).

B20 is distributed to most of the Keene municipal fleet from the city's central underground storage tank system. B20 is used in fire engines, dump trucks and diesel trucks. Fleet nonroad vehicles at remote locations (that can't access the UST) do not use B20 due to lack of availability and higher cost for special delivery. As of 2007, the City of Keene DPW has used over 200,000 gallons of B20 in their centralized fleet.

1.5 Is Biodiesel a Promising Technical Solution to the Problem of Diesel Exhaust Exposure? A Review of the Air Quality Impacts & Associated Health Risks

A review of existing scientific evidence on biodiesel tailpipe emissions suggests biodiesel may indeed provide an attractive alternative to petroleum diesel with respect to air quality. For example, numerous studies have shown burning biodiesel reduces harmful particulate matter from tailpipe exhaust (EPA 2002b; Graboski and McCormick 1998; Bagley et al. 1998; Durbin et al. 2000; Wang et al. 2000). This scientific evidence indicates biodiesel fuel may hold promise as a technical solution to the problem of diesel exhaust with respect to its impact on particulate matter emissions.

However, while much about biodiesel is known, there is also much that is unknown.

There are multiple dimensions to the study of biodiesel tailpipe emissions that have implications for risk decision-making. Most of the studies in the literature have focused on laboratory based tailpipe emissions from heavy duty on road diesel engines. There is limited data from nonroad engines on biodiesel tailpipe emissions (EPA 2002b). There is also limited data on 'real world' (compared to laboratory-based) biodiesel tailpipe emissions.

There is almost no data on biodiesel exposures in the workplace, with only one regulatory study identified at the time of this writing. The next sections identify what is currently known about biodiesel, identifies data gaps in the literature, and discusses the challenges in the use of biodiesel as an alternative to petroleum.

1.5.1 EPA's Regulatory Review of Biodiesel and the EPA (2002b) Draft Technical Report on Biodiesel Emissions

Biodiesel is the only alternative fuel that has passed the EPA Clean Air Act Tier I and II testing requirements for health effects. Unlike straight vegetable oil, biodiesel is legally registered as a fuel for sale and distribution in the U.S.; for registration, EPA's Tier I and Tier II tests are required by the 1990 Clean Air Act amendments for any fuel or fuel additive sold in the U.S.

The Tier I test is a series of tailpipe emissions tests and the Tier II test is a 90 day (or subchronic) inhalation rat study where the animals are exposed to varying levels of biodiesel exhaust. The emissions testing for the Tier I requirements followed a series of protocols (CFR Title 40 Part 79), including detailed tailpipe emissions characterizations with the fuel burning on one or more diesel engines. These engines were operated according to specific test requirements (Federal Testing Protocol CFR Title 40 Part 86 Subpart N) that span the engine's torque capabilities and operating speed (Sharp et al 2000a). The Tier I tests were performed in a lab controlled environment, characterizing regulated emissions of particulate matter, total hydrocarbons, NOx, and carbon monoxide as well as unregulated emissions of aldehydes, PAH's, and nitro-PAH's. Emissions levels are reported as grams/horsepower*hour or mass per unit of work, not in units of concentration such as g/m3. The Tier I test results found B100 and B20 emissions of PM, total hydrocarbon, and carbon monoxide were reduced when compared to petroleum diesel, although NOx levels increased (Sharp et al. 2000a). B100 and B20 emissions of aldehydes, PAH's and n-PAH's also were reduced relative to diesel emissions (Sharp et al. 2000b). For both regulated and unregulated emissions, the B100 emissions profiles showed more dramatic reductions of measured emissions vs. diesel than B20, except for NOx, where B100 use resulted in higher emissions than B20.

In the Tier II animal study, rats were exposed to 100% soy-based biodiesel exhaust (at three levels represented by exhaust concentrations diluted to 5, 25, or 50 ppm NOx. After the 90 day test period, Finch et al. (2002) determined only modest adverse effects at the highest exposure level. The inhalation exposures for the rats resulted in a dose-related increase in particle-containing alveolar macrophages; however, this observation was similar to that seen in similar petroleum diesel exhaust rat exposure studies (Finch et al. 2002).

In addition to the regulatory Tier I and Tier II requirements EPA also completed a draft technical report studying biodiesel emissions. EPA's study (2002b) analyzed and consolidated data from numerous published studies and concluded that B20 would reduce particulate matter (PM) by approximately 10%. The report also found B100 could reduce PM by as much as 50% compared to petroleum diesel. Most of the EPA (2002b) reviewed studies found increased NOx levels in biodiesel exhaust compared to diesel exhaust (2% increase in NOx for a B20 blend); however, the impact of biodiesel on NOx has been controversial and will be discussed in the next section.

The EPA (2002b) reported biodiesel use resulted in reductions in total hydrocarbon (vapor phase) and carbon monoxide as summarized in Table 1.1. The EPA (2002b) report recommended additional research was needed to fill in a number of data gaps including: more data from nonroad engines, from newer heavy duty engine models, from light duty diesel engines, and more air toxics data, especially on toxics of public health concern such as benzene and 1,3-butadiene.

1.5.2 Additional Literature on Biodiesel Tailpipe Emissions

1.5.2a Particulate Matter and Nitrogen Oxides

Most of the research literature on biodiesel tailpipe emissions indicates particulate matter (usually 10 micron diameter and lower) levels are reduced by burning pure biodiesel or biodiesel blends (EPA 2002b; Graboski and McCormick 1998; Bagley et al. 1998; Durbin et al. 2000; Wang et al. 2000; Sharp 2000a; McCormick et al. 2001). A more recent study that employed both urban and freeway driving cycles to compare petroleum diesel/ B20 tailpipe emissions for heavy duty engines reported average PM reductions of 16% from B20 use (McCormick et al. 2006). Most research in the U.S. has indicated biodiesel use lowers PM emissions compared to petroleum diesel, with B100 use resulting in greater PM reductions than B20 use. However, due to the PM/ NOx tradeoff, lower PM levels are expected to result in higher NOx levels.

There have been conflicting research results regarding the impact of biodiesel on NOx tailpipe levels, with some studies indicating an increase, and others no significant change.

The contradictory evidence regarding biodiesel's impact on NOx levels has prompted some states like Texas to consider – though not yet implement - a ban on biodiesel (Schmidt 2007). EPA's (2002b) report indicated use of B20 would result in a 2% increase in NOx emissions, with increasing levels of NOx associated with each percentage increase in the biodiesel/ petroleum diesel blend ratio. However, researchers from the National Renewable Energy Laboratory (NREL) team recently challenged these findings. McCormick et al. (2006) examined NOx emissions from eight heavy duty diesel vehicles and concluded that while NOx levels were highly variable, there was no statistically significant difference in NOx emissions between B20 or petroleum diesel use. When they expanded the review to include other engine and vehicle studies they found the net average overall NOx effect from B20 was ± 0.5 % (McCormick et al. 2006). McCormick et al. (2006) point out almost half of the NOx data in EPA's (2002a) draft technical report came from engines from a single engine manufacturer, potentially biasing the NOx predictions when considering the engine variety in the national fleet. Since NOx contributes to ground level ozone, and many areas in the country exceed National Ambient Air Quality Standards for ozone, these types of scientific inconsistencies have left local state air regulators and other policy makers unsure about how to regulate biodiesel as the market expands.

In other relevant literature on diesel vs. biodiesel PM comparisons, Shi et al. (2005) showed B20 use reduced particulate matter emissions 17 to 34% compared to pure diesel. Chen and Wu (2002) found that burning B100 reduced the total number concentration of ultrafine particles (less than 1.0 micron in diameter) by 24 to 42% and the total mass concentration by 40 to 49%. Ultrafine particles have been identified as a potential health concern since they are smaller than fine particulate matter, and may penetrate into even deeper regions of the lung (HEI 2002). Jung et al. (2006) found burning B100 resulted in decreased particle size (80 nanometer to 62 nanometer diameter), number (by 38%), and volume (by 82%). Although the decreased number and volume of particles are beneficial, the smaller particle diameter appears to indicate the biodiesel particle may be morphologically different than diesel, which can be associated with negative health effects.

1.5.2.b Elemental Carbon/Organic Carbon

There is little data characterizing elemental and organic carbon levels in biodiesel emissions. Organic carbon levels for both B100 and B20 blends were higher when compared to a California diesel and synthetic diesel blend; alternately, elemental carbon levels were lower for B100 in the same study (Durbin et al. 2000). More typically, SOF or soluble organic fraction is measured. Here the database is limited but research is beginning to provide a clearer picture of biodiesel emission profiles. The level of soluble organic fraction (SOF) of particulate matter has been found to be higher in biodiesel exhaust compared to diesel exhaust (Graboski and McCormick 1998). However, polycyclic aromatic hydrocarbons (PAH's), which are organic species of primary human health concern due to their potential mutagenicity and carcinogenicity are generally reduced when biodiesel emissions are compared against petroleum diesel (Bagley et al. 1998; Durbin et al. 2002; Sharp et al. 2000b; Correa and Arbilla 2006). Bagley et al. (1998) found that both particle phase and vapor phase PAH's were lower with B100 compared to diesel fuel exhaust from nonroad equipment used in mines. Correa and Arbilla (2006) determined in their study of heavy-duty bus engines that reductions in PAH levels correlated with the percentage of biodiesel in the blend, with an average reduction of 2.7% for B2, 6.3% for B5 and 17.2% for B20.

1.5.2.c Air Toxics and Other Research Needs

Also relatively unstudied are the levels of air toxics (such as formaldehyde and acrolein) in biodiesel exhaust and the size distribution of particulate matter (fine particles vs. ultrafine particles). While Sharp et al (2000b) showed biodiesel reduced formaldehyde and other carbonyl levels, Turrio-Baldassarri et al. (2004) determined significantly higher formaldehyde emissions in B20 exhaust compared to diesel exhaust. More research is needed to better understand the composition of toxic gases in biodiesel exhaust as well as the impact of biodiesel on particle size distribution (McCormick 2007).

1.5.2.d Other Literature: Biodiesel Emissions Health Effects Testing

While the literature on biodiesel emissions characterizations is growing, there has been limited research examining biodiesel emissions' impact on human health via in vivo or in vitro tests. Epidemiological studies are not available, likely due to the relative newness of biodiesel in the U.S. The primary biodiesel exhaust animal study (in vivo) was the rat inhalation study by Finch et al. (2002) described previously, which indicated no major adverse health effects associated with subchronic exposure. In an in vitro study, Bagley et al. (1998) determined no vapor phase mutagenicity with soy based B100, and suggested that use of biodiesel is not expected to increase toxic health effects (associated with particle bound PAH's) compared to diesel emissions. Bunger et al. (2000a) found that particles from both rapeseed and soy based biodiesel exhaust contained lower levels of black carbon and total PAH's than diesel fuel, with less mutagenic potential. Kado and Kuzmicky (2003) found higher mutagenicity rates for canola based biodiesel exhaust compared to soy, but both were lower than mutagenicity rates associated with petroleum diesel exhaust. Researchers who studied both mutagenic and cytotoxic effects of diesel and rapeseed based biodiesel determined lower mutagenic potency for the biodiesel but higher cytotoxic effects on mouse fibroblasts (Bunger et al 2000b).

Contradictory results for biodiesel's impact on health effects have been reported in the literature. Mutagenicity tests performed in a more recent study on biodiesel (B20) and diesel exhaust from a heavy duty bus engine indicated no statistical difference between both fuels (Turrio-Baldassarri et al. 2004). While Kado and Kuzmicky (2003) reported lower total mutagenicity emission rates from biodiesel exhaust due to the lower particle mass emission rate, they found higher mutagenic activity per particle mass for biodiesel fuels. Other researchers point out long term human health effects from biodiesel emissions have not been given "due diligence" especially as biodiesel appears to increase the soluble organic fraction of particulate matter (Swanson et al. 2007). Swanson et al. (2007) recommend study of the potential for increased oxidative stress from biodiesel exhaust due to its higher soluble organic fraction. Composition of the soluble organic fraction remains relatively uncharacterized as most tailpipe studies have focused on regulated pollutants such as total particulate matter and NOx, and not the speciation of the soluble organic fraction (SOF).

Finally, the rat inhalation study of Finch et al. (2002) used subchronic (i.e., less than 90 days) animal testing protocols. Long term health effects may be missed and exposure data are needed from multiple and varied end-uses of biodiesel to ensure humans exposures are similar to the doses used in animal health effects testing (Swanson et al. 2007). These research gaps emphasize the need for multiple biodiesel exposure assessment studies from "real world" applications.

1.5.3 Emissions vs. Exposure

The literature above briefly summarizes the tailpipe emissions characterizations for biodiesel, as well as the emerging health effects literature. The tailpipe emissions literature is growing rapidly as economic and political forces expand the biodiesel market. However, as others have noted (Swanson et al. 2007; McCormick 2007; EPA 2002b), more research on biodiesel emissions and health effects are needed to fill in the following gaps: understanding changes in tailpipe emissions profiles from different types of engines (such as potential changes in particle size, organic composition and organic fraction), characterizing air toxics in biodiesel exhaust, quantifying exposures from different applications, and evaluating potential long term health effects.

In addition to tailpipe emissions testing, a critical need exists for the characterization of exposure profiles in real world applications. While tailpipe emissions data inform environmental decision-making regarding the composition of exhaust emissions and aggregate mobile source contributions to air shed inventories, exposure data are necessary to inform decision-making regarding the impact of emissions on human health and the environment. Exposure - or human contact with the components of tailpipe emissions - is a key link in the chain between pollutant sources and ultimate health effects. Exposure is much closer to what people are actually breathing.

According to Ott's (2007) risk conceptual model, pollutants first originate from sources and then undergo fate and transport processes as they move through the atmosphere. When either diesel or biodiesel exhaust exits a tailpipe, there are a number of physical and chemical atmospheric processes that may occur prior to entering the breathing zone of a worker or community member. Physical processes include wet or dry deposition, and chemical processes include oxidation or nitration. Diesel particulate matter less 1.0 micron in diameter may have a residence time of days before settling out via dry deposition (Winer and Busby 1995). Physical and chemical processes may modify the exposure — either increasing or decreasing the toxicity of the associated health effect. For example, PAH's released in the vapor phase of diesel exhaust may be chemically transformed in the atmosphere by the addition of nitrogen to become more potent mutagenic species like 1- nitropyrene (HEI 1995). Alternatively, physical and chemical processes may reduce exposure or reduce the toxic health effect. Rain events can remove particulate matter from the atmosphere, effectively scrubbing them out of the air, thereby reducing human exposures.

Ultimately, the measured exposure (and estimated potential dose) determines the human health effect. Therefore, in any inhalation risk characterization of a chemical or pollutant, exposure data is necessary in addition to source data for fully understanding the impact of air contaminants on human health and the environment.

An additional benefit of collecting exposure data is that exposure data is determined "in the field" or during real-world ongoing activities or processes. In contrast, most tailpipe emission profiles reported in the diesel and biodiesel literature are not "in the field" concentrations but are determined by testing tailpipe exhaust in a laboratory via the Federal Testing Protocol (FTP). The FTP involves sequential steps where the vehicle is in a controlled environment and the engine is operated at different speeds for set time periods.

These steps are not expected to be the same as real-world engine operation, but provide a way to model emissions output at different speeds.

Tailpipe emission testing has advantages compared to exposure monitoring. In a lab setting, the researcher can control environmental variables like temperature and humidity.

There is also no wind so there is neither dispersion of pollutants nor interference from another upwind pollution source. While the control of confounding variables clearly helps understand speciation of components generated during the combustion process, the data may not necessarily reflect emissions from actual stop and go urban driving conditions or on- highway moderate or heavy traffics.

It is because of the real-world variability in weather and driving/operating conditions that make it difficult to predict occupational or community exposures based on tailpipe emission datasets. Lab based tailpipe studies may not reflect typical engine types, engine use patterns or emissions profiles from "real use" scenarios. When Shah et al. (2004) used a mobile laboratory to measure petroleum diesel tailpipe emissions in real time from heavy duty trucks, the researchers found that PM, EC, and OC levels were highly variable and strongly dependent on the mode of vehicle operation. Higher emissions were determined from trucks in congested traffic conditions compared to highway cruise conditions (Shah et al. 2004). Other researchers found the organic carbon/elemental carbon ratio from a diesel engine tailpipe varies depending on operating conditions and vehicle load. Heavier load cycles increased elemental carbon levels and lighter load/idling conditions increased organic carbon levels (Shi et al. 2000).

A final gap in the biodiesel tailpipe emissions and exposure database is that nonroad engines are underrepresented in emissions characterizations. Yet, nonroad engines are more common in workplace scenarios such construction sites or industrial warehouses making them more relevant to understanding workplace or community exposures. These types of nonroad applications or scenarios are favorable for quantifying exposures, as activities may be consistent throughout a workshift, the population exposed is easily identifiable, and exposures tend to be higher and provide worst case scenarios for health impacts. The relationship between nonroad engines and typical workplace uses and the lack of current biodiesel exposure data is discussed in the next section.

1.5.4 Lack of Biodiesel Exposure Data

Nonroad engines are used in a number of work settings such as farming, construction, and industrial operations. With respect to existing diesel engine technology, and assuming the use of 100% petroleum diesel fuel, nonroad engines generate higher levels of NOx and PM compared to onroad engines. As previously discussed, workplace exposures to diesel exhaust tend to be much higher than community exposures, raising important questions about the environmental injustice occurring inside compared to outside the facility fence. Nonroad engine applications that persist over long time periods in a community, such as a multi-year construction site, may impact both environmental and occupational health concurrently. For these reasons, nonroad diesel engine exposure data are particularly relevant and examination of biodiesel as an alternative to petroleum diesel especially compelling.

Biodiesel emissions data indicate pure biodiesel and biodiesel blends reduce particulate matter compared to petroleum diesel. Although this data has been collected mainly from onroad engines, the limited nonroad tailpipe tests also indicate PM is reduced by burning biodiesel. There is a large scientific database supporting the connection between fine particulate matter exposure and significant negative health effects such as lung injury, respiratory illness, asthma exacerbation, irregular heartbeat and heart attacks. Reducing fine particulate matter in both the workplace and local environment would have enormous health benefit. In fact, EPA quantitatively estimated public health benefits in the range of 9 to as much as 75 billion dollars by the year 2020 from reducing the fine particulate matter standard from 65 to 35 µg/m3 (EPA 2006).

Biodiesel blends may offer an effective risk intervention that can reduce some of the key, harmful components like fine particulate matter associated with diesel exhaust in high exposure scenarios like the workplace. Because of the operational benefits to the diesel engine such as increased lubricity, biodiesel blends also appear to be an intervention that can be implemented immediately.

To fully understand the impact of biodiesel on human health and the environment, exposure data is needed. Yet, there is a critical lack of biodiesel exposure data in the scientific literature. At the time of this writing, a literature review found only one biodiesel exposure assessment - an internal Mine Safety and Health Association report that measured biodiesel work area exposures in different areas in a mine in Maysville, Kentucky. B20 use generally reduced PM & EC, and increased OC (Shultz 2003). There was no research identified that examined the effects switching to biodiesel may have on both occupational and environmental exposures concurrently.

This lack of integrated research is a symptom of the regulatory and institutional barriers described earlier that impede looking at ways to reduce both environmental and occupational chemical exposure risk. This study addresses that disconnect by evaluating biodiesel's impact on environmental and occupational exposures concurrently. Biodiesel may offer an important health risk reduction alternative to petroleum diesel exhaust.

However, biodiesel's impact on NOx is still unclear. The data gaps in the literature on biodiesel emissions and exposures, if not examined, may ultimately present new risk challenges, especially as biodiesel production capacity and distribution increases in the U.S. There has also been increasing concern among scientists and environmentalists that biodiesel use may result in unintended environmental and social harm. These points are discussed next.

1.5.5 Food vs. Fuel: A Challenge?

A big political push for biodiesel has been the need to identify renewable sources of energy that can replace liquid petroleum fuels. Decreasing domestic oil reserves, reliance on oil from the volatile Middle East, diminishing worldwide oil supply, global warming concern and other extrinsic drivers are driving the growth of the renewable energy industry (Klass 2003). Yet, in spite of the potential political benefits, biodiesel does have some detractors who point out what they perceive as significant problems with the alternative fuel.

Biodiesel is more expensive than petroleum diesel, and the cost varies depending on the feedstock used to make the biofuel portion. Pure biodiesel has an EEL (energy equivalent liter) cost of 82 cents per liter versus 53 cents per liter for diesel (Manuel 2007). An energy equivalent liter cost attempts to normalize the costs of the different types of fuel by accounting for both the energy that goes into making the fuel as well as the energy output of the fuel. B20 prices at the retail pump tend to be only slightly higher than pure petroleum diesel due to tax credits. In 2005, biodiesel could not compete economically with petroleum diesel without federal subsidy (Hill et al. 2006). This subsidy has been in the form of a tax credit for distributors at a penny per percent point of biodiesel blended into petroleum diesel, with the savings passed to consumers (Pahl 2005). Even with the subsidy, biodiesel is more expensive for consumers than diesel, but this difference has narrowed to a less than 5 cent difference per gallon for B20 in some regions of the country.

Coupled with biodiesel's higher cost have been feedstock availability issues. Current agricultural feedstocks such as soy cannot come close to meeting existing petroleum diesel demand. Even if all the soy grown in the U.S. today was converted to biodiesel fuel, the amount would only meet 6% of petroleum diesel needs (Hill et al. 2006). In addition, critics point out that soy may be an overall poor choice of feedstock with respect to an energy balance over the fuel's life cycle. With its low yield of soy oil per kg of soybeans (18%), Pimentel and Patzek (2005) contend soybean crops are poor producers of biomass energy.
Per their calculations, production of 1000 kg of biodiesel with an energy output value of 9 million kcal requires an energy input of 11.9 million kcal, resulting in a net overall loss of energy of 32% (Pimentel and Patzek 2005). Other researchers also question the long term viability of a soy based fuel. Via their life cycle analysis that evaluates fertilizer impacts, Hill et al. (2006) found that cultivation of soy requires huge inputs of fertilizer (derived from fossil fuels) and releases nitrogen and pesticides from agricultural activities. In accounting for fertilizer impact, converting all soy to biodiesel would reduce biodiesel's net energy gain from displacing a maximum of 6% of petroleum diesel to displacing just 2.9% of diesel consumption (Hill et al. 2006). Conversely, Pimentel and Patzek (2005) found soy based biodiesel had little nitrogen impact and suggested biodiesel's limited nitrogen impacts were a benefit.

There is also concern among policy-makers that if biodiesel becomes more popular that the competition for soybean oil can set up a food vs. fuel war. Hill et al. (2006) believe that the potential for soy based biodiesel will be constrained by the important role that soy plays in human food supplies. While some biodiesel advocates believe this concern has been overemphasized (Pahl 2005), others argue that soy-based biodiesel is just a first generation biofuel. Biodiesel is considered by some to be a transition fuel with the critical next step developing biofuels from non-food based materials (Manuel 2007).1.5.5Food vs. Fuel: A Challenge?

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2. Methods

2.1 Analytic-Deliberative (A-D) Framework as Organizing Conceptual Approach to the Study: Overall Research Approach

The research design for this study is best described as multiple iterations of analysis and deliberation. Each A-D iteration revolved around a unique central research question.

Each central research question was linked into the study's operative research question. In this section, I present the specific research questions and how I collected data in support of each question. I will review how I applied the NRC's (1996) analytic-deliberative framework to ongoing biodiesel research activities between Keene State College and the City of Keene. I will review the operative (or "linking") research question, the three central research questions, and the relationship between the operative question and central research questions.

2.1.1 Overall Design Framework and Operative Research Question: Does Applying an Analytic-Deliberative Approach to Understanding B20 Exposures Lead to Improved Decision-Making?

The main application was the integration of a collaborative exposure assessment (CEA) (the "main analysis" in this study) with a Biodiesel Working Group (BWG) forum for deliberation. The collaborative exposure assessment (CEA) was performed at the City of Keene Recycling Center (KRC), a municipal resource recovery facility that utilizes non-road, construction-type equipment. The KRC is a relatively isolated, stable and long term source of diesel exhaust emissions in the local environment, which made it an excellent site to evaluate the relationship between occupational and environmental exposures.

The collaborative exposure assessment compared the impact of a 20% soy-based biodiesel/80% petroleum blend (known as B20) against 100% petroleum diesel on occupational and environmental exposures. The field work was performed by Keene State College (KSC) researchers, KSC students, and City of Keene employees. The CEA team measured in-cabin, work area, and local environmental concentrations of particulate matter, elemental carbon, organic carbon and nitrogen dioxide. The Biodiesel Working Group (BWG) was the deliberative forum for discussion of the collaborative exposure assessment strategies, activities, results, and potential future decisions related to the use of biodiesel by the City of Keene Department of Public Works (DPW). BWG members included participants in the collaborative exposure assessment, local decision-makers, and other interested and affected parties. The interconnected phases of analysis and deliberation informed each other throughout the dissertation research and after the dissertation data collection phase ended.

The CEA/BWG connection is the heart of this study. The linking, operative research question was: does applying an analytic-deliberative approach to understanding B20 exposures lead to improved decision-making?

However, I must stress that the Biodiesel Working Group's initial envisioned purpose was to help improve the collaborative exposure assessment research process as described above and subsequently communicate the exposure assessment results locally in educational outreach initiatives. The primary aim in June 2006 at the first BWG meeting was that CEA/BWG participants would discuss exposure assessment strategies and uncertainties, any concerns relating to exposure assessment activities, and review where and how to communicate the results. No other structured goals were in place when the first BWG meeting was held; in this sense, this study was an application of the A-D model, not a test of it to predict specific outcomes. In fact, Central Research Questions #2 and #3 emerged from participatory aspects of the process. These questions were not predicted, but I studied them as they were a direct result of application of the A-D model. At the start of this study - the connection of the BWG to the collaborative exposure assessment - Central Research Question #1 was: Does use of B20 reduce exposures of PM2.5, EC/OC and NO2?

2.1.2 Central Research Question #1: Does use of B20 reduce exposures of PM2.5, EC/OC and NO2?

Russell and the City engaged researchers at Keene State College in 2004 to try to help their organization answer the initial question: is biodiesel healthier? Researchers and undergraduate students from Keene State College had collaborated with City of Keene employees to examine the impact of biodiesel fuel on occupational and environmental exposures in a 2004 pilot study. The City wanted to more fully understand what they perceived to be real, undocumented benefits of biodiesel – the cleaner workplace air - in order to increase biodiesel awareness locally and regionally. Russell in particular was frustrated at being consistently asked during his local and regional presentations for "facts" to support his claim that biodiesel had made his workplace air cleaner (Russell 2006).

There are multiple ways to approach the question: "is biodiesel healthier?" For example, worker health surveys or animal toxicology studies are other potential research strategies. Based on the KSC and City of Keene team's interests, collective expertise and available resources, we decided on a comparative exposure assessment strategy. We took the original question, "is biodiesel healthier?" and refined it to the testable hypothesis "does use of B20 compared to petroleum diesel result in differences in PM2.5, EC/OC and NO2 levels in the workplace ("occupational exposures") and local environment ("environmental exposures")?" These pollutants were selected because of their policy relevance, since there is a wide literature connecting PM2.5 exposure to health effects, EC is widely accepted as a surrogate for diesel, and NO2 is of key interest in regulatory circles for its connection to smog. When the 2004 pilot indicated significant reductions in particulate matter, both groups agreed to do an expanded exposure assessment study. Prior to the expanded exposure assessment field work, I organized and started the deliberative Biodiesel Working Group. For the first question, this will include review of the strategy of the collaborative exposure assessment, the strategy of the Biodiesel Working Group, and the quantitative and qualitative data collection methods employed in each phase. I will also more fully describe the roles of the participants in the research.

However, like a gear turning other gears in a watch, the initial integration of the exposure assessment with the BWG led to new, subsequent central research questions that continued the analytic-deliberative interactions among KSC researchers and interested and affected parties. As a real-world application of the A-D model, there was no guarantee that the BWG process would ever gain traction or much less lead to any tangible outcomes or decisions. However, participants desired to "do more" with the exposure assessment results, and this led to the development of subsequent Central Research Questions #2 and #3. The A-D framework was then applied to each of these questions.2.1.2Central Research Question #1: Does use of B20 reduce exposures of PM2.5, EC/OC and NO2?

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2.1.3 Central Research Question #2: How Can Local Supply of B20 Be Increased?

The results of the collaborative exposure assessment performed in July and August of 2006 led to a decision by the BWG to explore increasing use of B20 in Keene. Various ideas such as using biodiesel for heat were discussed, but almost immediately the lack of local biodiesel supply was identified as a critical structural barrier. Thus the second Central Research Question #2 in this process became: how can local supply of B20 be increased?

While the main deliberative activities continued to be meetings of the BWG, new analytic activities included interviewing local fuel oil and diesel duel distributors. The time frame of Central Research Question #2 activities spanned from January 2007 to approximately March 2007.

2.1.4 Central Question #3: How Can an Innovative Public/Private/College Collaboration Manufacture Biodiesel in the Local Community?

Further analysis and expanded deliberations (and an expanded-yet-again BWG) led to the final question, Central Research Question #3: How can local stakeholders collaborate to build a local biodiesel production facility? Information gathered during A-D activities for Central Research Question #2 indicated a number of external barriers impeding the expansion of biodiesel supply in rural areas like southwestern New Hampshire. The BWG membership had expanded yet again, to include a private engineering firm interested in collaborating with KSC in the production of biodiesel. This led to the final research question of this study, and numerous associated analytic and deliberative activities. Leadership of the BWG transferred from me to the KSC administration, and the BWG substantially expanded its membership. The decision-making process by this point had literally taken on a life of its own. These analytic and deliberative activities are still on-going as of the publication date of this dissertation, but I stopped collecting field data in June 2007.

2.1.5 Rationale for Linkage

The overall research design framework or organizing conceptual schema for this study is the integration of analysis and deliberation as recommended by the NRC (1996). This integration of analysis and deliberation was implemented as illustrated in Figures 2.2, 2.3, and 2.4. The 3 central research questions converge in support of the operative question: does applying an analytic-deliberative approach to understanding B20 exposures lead to improved decision-making? The NRC (1996) states application of the A-D model can lead to better risk decision-making by ensuring that decision-relevant knowledge level is as a complete as possible, uncertainties are addressed as comprehensively as possible, and concerns are acknowledged as fairly as possible. In this case, application of the A-D model was expected to better fuse local and expert knowledge on biodiesel and link any new knowledge that emerged from the CEA/BWG research process to the ongoing biodiesel policy discourse at the local, regional and potentially national policy level. I expected that accomplishing these aims would lead to an enhanced understanding of B20 exposures which could lead to overall improved decision-making as suggested by the NRC (1996). In short, I hoped purposely connecting analysis and deliberation would enhance the CEA process itself (design and data collection) as well as increase the policy relevance of the results.

From a broader, more theoretical perspective, I applied the A-D model to move beyond the existing risk assessment vs. risk management divide that artificially segregates science and policy, as well as segregating technical and other forms of expertise. Instead of keeping technical analysis and deliberations separate, as is common in scientific research performed in regulatory contexts (such as the assessment of diesel exhaust emissions and exposures), I hoped combining the two would increase collaboration among participants and help move beyond regulatory and institutional barriers to better inform understanding of B20 exposures.

Additionally, since Biodiesel Working Group membership consisted of diverse people involved in both analytic and deliberative activities, who represented various viewpoints and values systems, process concerns could be identified early and any decisions made had the potential to be considered more legitimate. And finally, the A-D model helped structure research and discussion of the concurrent impact of B20 on occupational and environmental exposures, to help move beyond regulatory and institutional barriers that tend to segregate the workplace from its environmental context.

In most cases from the environmental decision-making/ public participation literature, citizens and stakeholders take information from technical experts as a "given" input to the decision-making process. Technical analysis activities are often kept separate from deliberation. The NRC (1996) report argues that this separation contributes to risk decisions that miss important relevant knowledge, do not address citizen/stakeholder concerns, are seen as illegitimate, waste regulatory agency resources over the long term and decrease citizen/stakeholder trust in regulatory processes. While citizen participation via town hall meetings, advisory panels and other mechanisms has become commonplace in environmental policy-making over the past 30 years, citizen involvement in the science that informs the policy is relatively recent (Lynn 2000).

While mainly using the A-D framework and associated literature referenced in the NRC (1996) report, I was also influenced by similar ideas from the literature on community based participatory research (O' Fallon and Dreary 2002; Judd et al. 2005; Sclove et al. 1998), in trying to increase participation in analytic activities. For example, three principles of community based participatory research relevant to this study were promoting active collaboration at every research stage, fostering of co-learning, and disseminating research results in useful terms (O' Fallon and Dearry 2002). While not explicitly identified as such by its advocates, community based participatory research (CBPR) may be considered philosophically similar to participatory action research, although the action in CBPR is guided more by the sponsoring research organization, not necessarily the participants (Corburn 2005). In addition to CBPR principles, I was influenced by Fischer's (2000) critique of the NRC's (1996) focus on deliberation as leaving science squarely in the domain of experts, diminishing nonexpert participation in analysis. The community based aspects were especially pertinent in involving KSC undergraduate students in the performance of much of the day-to-day field work, working alongside KRC employees at a location often frequented by community members.

One final point about the overall study design: since both natural and social science phenomena were studied, this research employed both quantitative and qualitative methods to collect data. The research design (or application of the A-D model) was clearly unique and specific, and as such the overall methodological approach was hybridized. I found case study design principles provided a helpful methodological lens. Focusing on the KSC/City B20 research collaboration as a case unit of analysis helped coordinate the use of and clarify the purpose of different quantitative and qualitative research strategies and data collection techniques. According to Yin (1984), case study is an appropriate strategy for "how" or "why" questions for contemporary events over which the research has little or no control.

My participation as both natural and social scientist meant this case could be considered revelatory per Yin (1984), as my role gave me insider status to phenomenon of risk-decision making not typically pursued or available to most natural scientists. Typically, scientists present and explain data to policy-makers under the traditional risk decision-making model that emphasizes a facts vs. values dichotomy. Finally, case studies use a variety of evidence in data collection to triangulate data analysis, an approach I followed for this study.

The need for quantitative strategies and data collection methods is relatively intuitive for studying natural phenomenon: to measure levels of air contaminants in the workplace and local environment, quantitative measurements were required. The Biodiesel Working Group and associated deliberations embodied the social phenomenon of this research. Social phenomena are better suited to qualitative inquiry. Creswell (1998, p. 15) defines qualitative research as follows:

Qualitative research is an inquiry process of understanding based on distinct methodological traditions of inquiry that explores a social or human problem. The researcher builds a complex, holistic picture, analyzes words, reports detailed views of informants, and conducts the study in a natural setting.

Creswell (1998) further clarifies that complex and holistic refer to a narrative examining the "multiple dimensions of a problem or issue". Since there are multiple dimensions to this study, qualitative methods provided a deeper understanding of the holistic and interactive relationship between the exposure assessment analysis and associated deliberations. Staying only within a quantitative realm would overlook the larger, more complex picture of how the collaborative aspects of the research emerged and evolved. Without a qualitative component, we would lose insight into the interactive nature of the process of scientific analysis and how connecting deliberation to analysis can better inform risk decision-making. Creswell (1998) emphasizes that qualitative inquiry is appropriate when such a detailed view of a topic is desired.

2.2 How the Concepts from the A-D Model Were Applied: Central Question #1

A summary of the how each of the A-D model steps were applied to each Central Research Question is shown in Appendix D. For the remainder of this section, I will explain in detail how these concepts were applied. First, I must note that while the central research questions and A-D model steps are listed sequentially, this does not imply the research activities actually occurred in a straightforward linear fashion, or that analysis or deliberation "neatly" interacted in a prescribed fashion. In fact, one of the main challenges in discussing the research methods (and later, presenting results) has been how to best capture the overlap and interactive relationships between analytic and deliberative activities, while clearly explaining what I did and the results that were observed in an accessible manner for the reader. While the A-D framework as illustrated and applied in this case in Appendix D, is shown as an ideal progression of steps, the NRC (1996) emphasized that a "common misunderstanding" is that analysis and deliberation in decision-making will proceed in a prescribed sequence. The research activities in this study certainly did not proceed in a linear fashion or followed the steps in exact order as outlined in Appendix D. In fact, the research progressed more like the saying - three steps forward, two steps back. But even within the significant overlap or "messiness" of analytic-deliberative activities, there was an overall forward progression of decision-making. Therefore, I have attempted to organize these activities to be accessible, with as much clarity as possible. This study also does not fit in a neat methodological taxonomy, but rather borrows from a quantitative and qualitative methodological toolbox unique to the operative and central research questions. In this way, this dissertation was truly interdisciplinary.

2.2.1 Problem Formulation

Central Research Question #1 originated from the local observations made by City of Keene employees about B20 use in the Department of Public Works (DPW) fleet. As summarized by Russell, "You pull a truck into my shop now and you don't even know it's diesel" (Cohen 2003). Similar observations were shared with me during informal conversations with the City of Keene and Keene State College employees regarding their B20 and B100 use. Bud Winsor, Assistant Director of Physical Plant and Grounds at Keene State College noted, "Equipment operators report fewer headaches at the end of the day, the fumes don't smell bad; it was a great move" (Cohen 2003). These informal discussions framed the initial question, "is biodiesel (B20) healthier than petroleum diesel?" The dramatic impact of B20 in the workplace is best summarized by Russell (2006):

I noticed it myself. My office in the old building was adjacent to the shop…every time they would drive a diesel engine into the shop… we had no air quality equipment in that shop. Those diesel fumes would stay there for a period of time and I found myself with a lot of headaches. I would go open the window, try and get rid of the headaches so fast forward to using biodiesel… the same equipment goes into the shop, same environment, same everything and I'm not getting any headaches. It was very strange and I'm trying to rack my brain, why aren't I getting headaches now. Then I realized it was the B20. It was the biodiesel.

Russell and I approached Dr. Melinda Treadwell at Keene State College to collaborate on a research strategy to attempt to quantify this observation. Dr. Treadwell had specific expertise in lung toxicology, and she had pervious experience in performing diesel exposure assessments. She agreed the City of Keene observations supported exploring B20 as a risk reduction intervention to diesel exhaust exposure. Dr. Treadwell and I collaborated to refine the initial question of "is biodiesel healthier" to the testable hypothesis "does B20 compared to petroleum diesel use result in differences in occupational and environmental exposures of PM2.5, EC/OC, and NO2?" How to test this hypothesis became the initial problem formulation. Dr. Treadwell provided the funding, equipment, and student resources for the 2004 pilot exposure assessment and 2006 expanded exposure assessment. In summary, the genesis of Central Research Question #1 started the way many scientific studies begin, by developing a hypothesis for an observation made over time. In this case, the observation initially came from nonscientists. Further detail on roles and responsibilities in performing the research is discussed in the section 2.2.4.a.

2.2.2 Process Design

2.2.2.a Site Selection

The City of Keene Recycling Center (KRC) was chosen after internal deliberations as the best site for the exposure assessment due to a number of characteristics: remote location, consistent operations on a week to week basis, use of nonroad diesel equipment by workers, a stable source of diesel emissions in both the workplace and local environment, and generalizability of findings to other sites. The site is one of the largest municipal owned material recovery facilities in New Hampshire, but comparable to a number of privately owned facilities with respect to tons of material processed per year. Operations at the recycling center used non road or construction type equipment such as front end loaders to move cardboard, paper, plastic containers, glass and aluminum cans throughout the site.

There was also a segregated trash transfer area on the far end of the KRC building where local refuse was dropped off, consolidated, and then picked up via a large track excavator and placed into open box trailers for off site transport to landfills. There were 3 main pieces of equipment used: a large front end loader (John Deere Model 624H - 160 HP), a small front end loader (JCB Model 409 – 67 HP), and a large track excavator. Due to a building fire during the petroleum diesel use time period, B20 data was not collected in the large track excavator area; therefore, this equipment and the work area will not be discussed further.

The area of the fire was segregated from the other KRC recycling area and did not impact the data collection process for the other perimeters in this study.

The KRC consists of a single large building with one large bay door on the lower level/main floor area and 5 other side bay doors on the upper level of the building. Trucks from other towns and local trash hauling companies drive into the lower level area to dump cardboard and paper waste on the main floor. Town residents or other trucks drop off newspapers, aluminum cans or plastic containers at one of the side bays. Employees stand alongside a conveyor belt system to separate non-recyclables from the process stream. The conveyor belt and employee break room are located on a second level inside the facility. The small front loader works on the main floor area moving cardboard inside the building to another conveyor belt leading to a bailer machine located on a sub level in the building. The large front loader typically works on the metals pile in another outdoor location on the property, but also works on the main floor area inside the building to move paper into an open trailer for transport to another facility. Air monitoring was performed in areas designated Perimeter #1, #2, #3, and #4 during days when equipment operated on petroleum diesel and then on a B20 blend. Perimeter #5 was the large track excavator area; due to a fire in this area in early August 2006, B20 data was not collected for comparison purposes.

2.2.2.b Quasi Experimental Strategy for Exposure Assessment

The exposure assessment estimated diesel vs. biodiesel environmental and occupational exposures in "real world" scenarios at a rural recycling center. Exposure to a chemical is defined as the contact with that chemical with the outer boundary (i.e., skin, nose, mouth, eyes) of a human (EPA 1992). Mathematically, exposure is a function defined as the measured concentration over a specified time period, $E = \int C(t)\, dt$, usually simplified as a time weighted average, $E = \sum C_i t_i / T$ (Ramachandran 2005). Occupational exposure assessment is the process of defining and evaluating the acceptability of exposure profiles (Mulhausen and Damiano 1998). Because the workplace consists of many microenvironments through which and within which workers move, occupational exposure assessment focuses on measuring concentrations of air contaminants within the breathing zone of the worker (Ramachandran 2005). At a theoretical level, since the breathing zone area is emphasized, occupational exposure assessment closely estimates actual exposure, and is decision driven because it will typically compare the breathing zone concentration against a "safe" regulatory exposure limit.

Environmental exposure assessment measures concentrations of air pollutants in specific, stationary locations or areas. At a theoretical level, environmental exposure assessment is more focused on local/regional levels of pollutants, and determining the relationship between exposure and biologically effective dose. Exposure and the biologically effective dose (the delivered dose that impacts the target organ's receptor sites and causes a response) are never the same due to complex pharmacokinetic [i.e., absorption, elimination] and pharmacodynamic [i.e., repair, compensation mechanism] processes (Ramachandran 2005). An EPA exposure assessment would take the measured air pollutant concentration and apply a standardized breathing rate to define an "intake rate", then a potential dose (EPA 1992).

The quasi-experimental approach was appropriate for a number of reasons. A true experiment where a site is randomly selected from a population of similar sites was not possible since the KRC was the only site to which we had access, and no other recycling center in New Hampshire was using B20 in its equipment at the time of the study.

A quasi-experimental design was used to test the central research question, "Does B20 use change levels of PM2.5, EC/OC and NO2?" Independent variables are summarized in Table 2.1. Independent variables were: fuel type, engine type, day, temperature, relative humidity, wind speed, wind direction, level of equipment activity, equipment proximity to monitor, and outside vehicle traffic. These independent variables were measured for statistical control. The dependant variables were the levels of air contaminants (PM2.5, EC/OC and NO2) at each Perimeter #1, #2, #3, and #4. We addressed threats to validity, as discussed in Section 2.2.4.h.

In the summer of 2006 we spent five weeks at the Keene Recycling Center conducting environmental air monitoring in operator work zones and in the local environment. PM2.5 and EC/OC were measured at Perimeters #1, #2, #3, and #4. NO2 data were measured at Perimeter #2 only. For ten days during the period June 27 to July 27, 2006, equipment was running on 100% petroleum diesel to 90% petroleum diesel/10% biodiesel. For eight days of the study, from the period August 7 through August 17th, equipment was running on a soy-based 20% biodiesel/80% diesel blend (B20). Nitrogen dioxide data only was collected on the days August 22 and August 23, 2006.

Each day was a replicate measurement to minimize bias. The same equipment was operating and was monitored during both fuel uses. The main equipment at the Keene Recycling Center that ran on B20 included the small front end loader (JCB Model 409 – 67 HP) and large front end loader (John Deere Model 624H - 160 HP). Integrated samples (over at least a 6 hour period) were collected. Integrated sampling is defined as the continuous collection of a sample over an extended specified time period, typically an 8 hour work shift (Bisesi 2004). A single, integrated value for the level of air contaminant for the time period was determined and is presented in the results chapter. The advantage of integrated sampling is that multiple shifts and associated integrated values can be measured and averaged into a long term average. The long term average is considered a relevant index of dose for chronic health risk (Mulhausen and Damiano 1998). Diesel exhaust is considered a chronic health risk, though acute health impacts may also be a concern for airway irritation; chronic exposure metrics were emphasized in this study.

While KRC operations varied from day to day, operations were relatively consistent on a week to week basis. Other scholars have supported a strategy of 6-10 measurements to estimate the mean of an exposure profile (or the mean of a series of daily time-weighted averages) of a similar exposure group (Mulhausen and Damiano 1998; Ramachandran 2005). Similarly, using statistical theory, six daily integrated PM 2.5 measurements are necessary to estimate the average daily exposure so that the sample mean is within +/- 5 µg/m3 of the population mean at the 95% confidence level, assuming a standard deviation of 5 (cf. Kinney et al. 2000). This level of error is adequate for the goals of this study (pilot work indicated PM2.5 results on the order of 100 to 5300 µg/m3), but may not be considered adequate for other exposure assessment goals, such as comparing the mean to an occupational exposure limit.

The rationale in selecting where to place air monitoring equipment within the KRC site itself considered the nonroad equipment as pollutant sources, and "in cabin" breathing zone measurements as "worst case" employee exposure. Each location was measured during each sampling day. Perimeters #1, #2 and #4 would be considered occupational exposures since they are located within a work area or in the equipment cabin. Perimeter #4 is also a mobile source moving in and out of the building so it makes a contribution to the outside environment. Perimeter #3 as the main outside location would be considered a near field or environmental exposure. Due to these multiple contributions and since the KRC is a stable, long term source of diesel emissions in the local environment, perimeter #1, #2, #3 and #4 measured concentrations for PM2.5 and EC/OC were pooled together to triangulate the site to determine a "total KRC site average". NO2 was measured only in perimeter #2 which as an indoor work area, and at the height of the equipment exhaust discharge, was considered to be a "worst case" location. All employees who worked in Perimeter #4 were non-smokers in consideration of the potential confounding effects of cigarette smoking found by other researchers (Zaebst et al. 1991).

Consistency in the general air monitoring protocol was critical to minimize threats to validity from systematic errors. Students were trained by faculty and staff to perform basic air monitoring functions and traffic counts. For both petroleum diesel and biodiesel sampling days, researchers and students performed equipment calibrations before and after sampling activities, positioned the equipment in the same locations, and regularly performed operational checks on the equipment while in use. Preparation and calibration of the equipment was used as an instructional activity for the students. Therefore, the sampling interval was reduced for some days from a typical eight hour period to just over six hours.

This still measured the exposures over the majority of the work shift. Field days were cancelled if rain occurred in the morning because precipitation will scrub particles from the air. Only 2 biodiesel days were cancelled due to rain, but this resulted in less biodiesel days compared to diesel days.

2.2.2.c Biodiesel Working Group

The Biodiesel Working Group was the mechanism used for formal deliberation between exposure assessment collaborators, and other interested and affected parties. Using standard definitions for participatory mechanisms in environmental decision-making, it was most like a citizen advisory committee (NRC 1996; Beierle and Cayford 2002). Advisory committees usually look at an issue in depth and provide recommendations to an organization. The Biodiesel Working Group (BWG) looked at the issue of biodiesel use in Keene and performance of the exposure assessment in depth. However, the BWG was not commissioned by any organization to give recommendations. It was also envisioned to be more actively involved in exposure assessment. The BWG provided a forum or mechanism for face to face deliberation of issues relating to the research collaborative to extend discussions beyond individual emails, phone calls, and spontaneous conversations.

Achieving consensus was not emphasized as a goal of the group since participation was voluntary, and our recommendations were not requested by any organization. Since the City DPW approached KSC for expert assistance, my initial goal was to continue that conversation in a more formal way.

Webler and Tuler (1999) recommend that selection of members for a policy planning group, such as a Watershed Community Council, be based on representativeness, political clout, ability to motivate others, and ability to provide information and judgments. A snowball process is one way that members can be identified; this was the method I used to set up the BWG. I identified people with experience with biodiesel, internal decision-making authority, or were affected by biodiesel exposures. Then I consulted with Russell on these criteria and his ideas for the initial membership. Russell was aware of who in the City was involved in supporting the B20 decision, as well as who within the City organization may have a desire to become more involved in the exposure assessment research deliberations.

Getting a BWG off the ground was the primary initial goal at this time because without it there was no application of the A-D model. In addition to the members Russell suggested, I reached out to the KRC supervisors for their participation and also to recruit KRC workers as both groups would be considered affected parties. Between June 2006 and December 2006, I also designed and distributed a Biodiesel Knowledge Survey (discussed in the next section) via email using the email "cover letter" to try and recruit new BWG members.

Motivating participation in decision-making processes in today's busy world has been recognized as a challenge (Webler and Tuler 1999). Many of the potential BWG members were already veteran "meeting-goers" and were averse to participating in another structured process. Since I had already been working with many of the BWG participants in other aspects of the pilot study collaboration, I did not document a detailed process design, such as clarifying roles, meeting procedures, decision-making procedures, or expectations of the group membership. These elements did not seem necessary in this case. Instead, I stressed openness, flexibility and transparency for the process: members could come and go as they pleased, all members were included on email exchanges, if members couldn't come to a meeting they could send feedback via email, previous meeting discussions were reviewed at the start of each meeting, and emphasis was on maintaining a safe, respectful and open place for dialogue. I told people their input was important because without them there would be no BWG. Webler and Tuler (1999) have suggested these strategies and others – such as giving participants ownership of the process - are helpful in motivating participation in environmental decision making processes.

I structured the first meeting in June 2006 toward getting feedback on the exposure assessment strategy before the start of actual air monitoring in the field, and to introduce and get feedback on the idea of starting a BWG. I sent out an email to 3 City of Keene employees suggested by Russell, and added 2 KRC supervisory staff. In the first email, I identified who I was, my dissertation research, and the idea of introducing a more formal collaborative approach to the exposure assessment. I suggested in my email three goals for the meeting: talk about the field work planned for the summer/present the proposed research sampling plan, ask for feedback (did we miss anything/should we add anything), and discuss ideas/request feedback for a BWG moving forward.

For the first BWG meeting on June 16, 2006, I asked attendees to suggest other BWG participants, to continue to build participation via a snowball selection process. I used the meeting to ensure that the KSC was getting the "right science" in the exposure assessment.

In trying to apply the analytic deliberative model, I considered the NRC's (1996) suggestion to spend time on problem formulation. But for the first meeting, there wasn't really a classic "problem" confronting the BWG, and I didn't want to suggest one. So I brought back into focus the original reason the City and KSC were collaborating on the exposure assessment: to answer the question, "Is biodiesel healthier than petroleum diesel?" via the specific question "Compared to use of petroleum diesel, does use of B20 reduce exposures of PM2.5, EC/OC, and NO2?" I asked the group to think about what they would like to do with the results of the exposure assessment. Without participants defining the BWG's purpose, it seemed unlikely that meaningful participation would occur. I used open ended surveys at the first meeting as a way to encourage brainstorming on potential BWG goals and to structure future BWG discussions.

I committed to making a number of local public presentations (communicating the exposure assessment results) as one goal of the BWG, requesting BWG support and participation. After the first meeting, I tried to facilitate discussions toward the idea of giving the BWG a decision to make. The BWG meetings were not intended to be brainstorming sessions or wide ranging discussions on everything related to biodiesel. Instead, the BWG was structured as being purposeful – the initial purpose being what to do with the exposure assessment results, if anything. I tried to be sensitive to the paradox that I wanted to encourage participation but I also needed to lead the process as there was initially limited interest. My role in these first meetings was to facilitate discussions to honor the desires of all members, but also get a conversation going about potential ideas for BWG goals. In emails confirming meeting dates and times, I requested feedback and agenda items from the potential participants.

Again this highlights the paradox of the initial BWG meetings: my influence was on the process was stronger, but without it there likely would be no process. In fact some BWG participants gave me feedback that I wasn't being strong enough of a group leader. I provide a high level of detail of the BWG meeting interactions in Chapter 3: Results to be as transparent as possible about my role and influence. It was very difficult to inspire participation between June 2006 and the second meeting in December 2006. I was surprised by how much time it took to schedule meetings and the difficulty in locking in time with an already very busy group. Meetings had to be suggested at least two to three weeks before having one. Even then, there was never one time that was best for everyone. As a strategy to encourage participation, I stressed openness and flexibility so there was no sanction for not attending.

Admittedly, much of my early strategy in the first two BWG meetings was simply "just do it". Juggling my roles in quantitative data collection, data analysis, making public presentations, recruiting BWG participation and then preparing for BWG meetings was extremely challenging. Task management issues led to part of this delay between meetings.

2.2.3 Select Options and Outcomes

These figures inspired the initial deliberations within the BWG. However, another objective of this study (as an initial BWG goal) was to directly communicate the exposure assessment results locally through a series of workshops. This objective was important to support ongoing biodiesel educational outreach and to be sensitive to the community participation aspects of this project. A number of public presentations were held in late 2006 and early 2007. To assist in creating effective presentations, and to act as a tool for recruiting potential BWG members, a Biodiesel Knowledge Survey was designed and implemented among BWG participants and other interested and affected parties. Therefore there were a number of quantitative and qualitative data collection methods applied during this step of the A-D model. I will review the Biodiesel Working Group data collection methods first, the Biodiesel Knowledge Survey next, and the Outreach presentations last.

2.2.3.a Participant Observation

To gather data on the deliberations at BWG meetings regarding potential options and outcomes, I employed the following qualitative methods: participant/observation and documenting meeting minutes. Participant/observation was the primary qualitative data collection method used for BWG meetings. As my role as research project facilitator allowed me extraordinary access to biodiesel decision-makers in Keene and other participants such as workers, quite simply there was much data to be mined via the participant observation approach. Access was relatively straightforward for this research because I was approached by Russell in 2004 as an "outside expert" to help answer local questions about health and biodiesel. Over the next 2 years, I developed working relationships with many of the participants in this study revolving around issues of biodiesel use in Keene. This ongoing collaboration helped me gain access to other participants as I started the Biodiesel Working Group. For performance of the exposure assessment at the KRC, I was able to coordinate access for the other KSC researchers and students to the site and to work with staff as needed.

During the BWG deliberations and field work phase of the collaborative exposure assessment, the study had many characteristics similar to ethnography (Hammersley and Atkinson 1995). First, I was in the field setting for a prolonged engagement. The initial biodiesel conversations through performance of a pilot exposure assessment and expanded exposure assessment through the BWG process spanned a 3 year timeframe. As an example of the type of working relationship developed, I would occasionally travel with Steve Russell during his educational and outreach presentations to various groups. I co-presented "Biodiesel: Lessons Learned" to the Sustainable Energy Resource Group in Hanover NH in September 2006. Similar to ethnography, the research activities in analysis and deliberation had an evolving nature as the story of the collaboration unfolded.

Other ethnographic elements: during the collaborative exposure assessment phase, as monitoring equipment was left in place for a 6 to 8 hour period, I had time to engage in informal conversations with workers about what they thought about the exposure assessment and biodiesel, questions they might have, or suggestions for how to communicate the exposure assessment results. I made detailed observations and took reflective notes. My data collection approach during this field work phase and throughout the BWG process was to document any relevant discussions relating to biodiesel or the City of Keene's relationship with biodiesel. In short, if the subject of biodiesel or other related environmental issues (such as sustainability, air pollution or public health) came up, I would try to flesh out the participant's meaning and write it down. But this study is not ethnography in key ways: I was not trying to describe the "workplace culture" at the KRC, or trying to understand social roles and relationships, or trying to describe the "day to day life" as common in ethnography. Instead, I kept detailed notes only of case relevant discussions or observations in a field journal. Therefore, while I kept the journal with me everyday, I did not necessarily take notes everyday but only when biodiesel or related discussions, observations, or interactions occurred. I would return to my journal notes as soon as possible after data collection but no later than 24 hours to make reflective comments and memo in the margins. I would also make analytic comments in the journal to process the data as I was collecting it, to ease the formal data analysis process completed later. I used different colored inks or specifically wrote "NOTE" to distinguish analytic comments from original journal notes.

2.2.3.b Meeting Minutes

Meeting minutes were taken by a KSC student during the first 4 BWG meetings. Students were asked to write down the names of meeting attendees, the activities of the meeting, any major comments or questions that arose, and who initiated the comments or questions. Student meeting minutes were usually more substantive on exactly what people said, and my field journal focused on my interpretation of the meeting energy, body language, tone, and important comments. Together, both the meeting minutes and participant/observation data collection provide a comprehensive record.

2.2.3.c Biodiesel Knowledge Survey

A main objective of the Biodiesel Knowledge Survey was to assess the baseline level of knowledge about biodiesel in the prospective Keene BWG member pool. For example, while many people within the City of Keene municipal organization and Keene State College staff might be aware that both organizations were using biodiesel, it was less clear what people actually knew about biodiesel. Data from the Biodiesel Knowledge Survey would then be used to identify knowledge gaps to help more effectively communicate the results of the collaborative exposure assessment. These outreach presentations were a major outcome of this step in the A-D model. A secondary objective of the survey was as a communication and recruiting tool to solicit and motivate more participation in the BWG process. I sent an internet survey via the e-survey site Surveymonkey.com to BWG members, names suggested as potential BWG members and any names from my participation/observation data (including meeting minutes) that were even peripherally associated with the decision to use B20 in Keene.

On December 1, 2006, I sent the Biodiesel Knowledge Survey to 19 people via email, including the mayor of Keene, other City department heads affected by biodiesel use, and a number of Keene State College employees who also used or supported the decision to use biodiesel in the college fleet. In March 2007, I sent the same survey to the KSC student research team. Surveymonkey.com was used to design the 12 question survey, with questions based on basic factual knowledge about diesel exhaust and biodiesel fuel characteristics derived from internet, government and media sources.

For those potential participants without email access, such as KRC site employees, paper surveys were taken to the specific work area locations for City employees to fill out. These surveys were dropped off during a workshift in January 2007 and then picked up 2-3 days later. The paper survey's were precoded with a date and location, but were not individually coded. Therefore only group categorizations and evaluation were performed. The Biodiesel Knowledge Surveys were categorized by group: "Keene DPW workers" "KRC workers", "Decision-makers", and "Students". I was unable to identify individual respondents but able to look at group averages to evaluate for inconsistencies against data I collected via other methods, such as interviews, document review and participant observation regarding knowledge levels.

3. Discussion

Compared to use of petroleum diesel, the use of B20 at the Keene Recycling Center resulted in significantly lower PM2.5 exposures, some significantly lower elemental carbon (EC) exposures, significantly higher organic carbon exposures (OC), and higher nitrogen dioxide exposures. At first glance, these mixed results may seem to indicate that there is limited promise to the use of B20 as a technical solution to the problem of diesel exhaust exposure. But a deeper analysis shows otherwise; in this section I discuss the meaning and implications of the exposure assessment results.

Fine particulate matter exposure is well associated with numerous acute negative health effects, ranging from asthma, to arrhythmia, to increased emergency room visits, to premature death (EPA 2003b; Lippmann et al. 2003). Chronic low level exposure for healthy adults to high levels of fine particulate matter (similar to those seen in urban areas) is associated with a predicted reduction in total life expectancy (Pope 2000). Fine particulate matter exposures are even more harmful to children, elderly, and those with preexisting heart or lung disease. Diesel exhaust is an important source of fine particulate matter in many parts of the country, especially urban airsheds (EPA 2002a). Due to the body of evidence connecting fine particulate matter exposure and acute/chronic health effects, any intervention that could reduce fine particulate matter exposures from diesel engines would be highly relevant to environmental, public and occupational health policy. Any reduction in fine particulate matter exposures would be expected to have tangible and immediate improved health benefits for an exposed population.

Comparing diesel vs. biodiesel exposures, use of B20 at the KRC site resulted in consistent reductions in fine particulate matter levels in both the workplace and near field locations. The total KRC site mean during B20 use was significantly less (60.4%) then during petroleum/low biodiesel blend use. When the "transition fuel" days were removed from the analysis, the total KRC site mean for PM2.5 was significantly less (72.9%) during B20 operation than during 100% petroleum diesel operation. The decrease in PM2.5 after switching to B20 at the site was observed across all perimeter locations and during both high and low equipment activity levels.

While a reduction in PM2.5 was expected from B20 based on tailpipe emissions literature, the magnitude of the reduction observed in this study was unanticipated. The literature consistently supports significant reductions in particulate matter in tailpipe emissions when biodiesel blends are compared to petroleum diesel (EPA 2002b; Wang et al. 2000; Graboski and McCormick1998; Bagley et al. 1998; Sharp et al. 2000a; Chen and Wu 2002; McCormick et al. 2006). However, our study exceeded the higher end of reported particulate matter reductions in the literature (between a 30-40% reduction) that resulted from burning a B20 blend. It is possible some of the difference in magnitude is related to the different measurement methods used in tailpipe emissions studies, which typically measure a larger diameter particulate matter (< PM10).

Yet the reduction in fine particulate matter remains an intriguing question: why was a 60% to almost 78% reduction (at P4) in PM2.5 seen from use of only a B20 blend? This was actually a common question my colleagues and I were asked during both BWG deliberations and public forum presentations. There are multiple explanations. First, the chemistry of biodiesel fuel is fundamentally different than petroleum diesel. Unlike petroleum diesel, biodiesel does not contain aromatic hydrocarbons or sulfur, but is made up of methyl esters, which have oxygen embedded within the hydrocarbon chain. Biodiesel has a higher cetane value than diesel – due to its higher oxygen content in the fuel. The increased oxygen content enhances combustion, thereby reducing soot formation, which when combined with the lack of sulfur and aromatics results in lower overall particulate matter mass. Other researchers have hypothesized that the increased oxygen content in biodiesel could result in more efficient combustion, reducing particulate matter (Wang et al. 2000). The enhanced combustion hypothesis to reduce particulate matter is also supported by this study's significant 22.4% reduction in KRC total site elemental carbon (carbon soot) levels from B20 use. Carbon soot is the core of diesel particulate matter. Use of B20 immediately translates to a 20% reduction in sulfur, which is also part of PM2.5. Reduced aromatics in the B20 fuel as well as lack of metals within the biodiesel portion of the fuel also would be expected to reduce the total mass of PM2.5 as seen in this study.

This study measured exposures from nonroad engines, typically dirtier than onroad engines. The nonroad engines in this study were often operating under load which also produces more particulates. The higher exposures can lead to the potential for more dramatic particulate reductions for an oxygen rich fuel like biodiesel. Connecting back to the local observations of Keene workers, the 60 to 78% overall reduction in PM2.5 mass seen in this study could be enough to reduce acute impacts like eye irritation and headaches, leading to the anecdotal observations made by workers in both the City and KSC organizations.

However, while the exposure assessment showed a decrease in PM2.5 and EC, there was a highly significant 370.4% increase in organic carbon levels. Using other sampling and analytical methods, the soluble organic fraction of the PM has generally been reported as higher for biodiesel (Bagley et al. 1998; Graboski and McCormick 1998). As biodiesel fuel has a higher boiling point than diesel, less biodiesel will be vaporized and it is likely more unburned fuel will condense on any particles exiting the tailpipe. While higher organic carbon levels were expected, the highly significant increases highlight the need for additional research. Are the increases in organic carbon simply unburned biodiesel fuel, which is relatively nontoxic, or other species of hydrocarbons from incomplete combustion products?

For diesel particulate matter, adsorbed organic species are of particular health concern because many of these species have been found to be mutagenic (HEI 2002). The long term implication of chronic exposure to adsorbed, potentially toxic species on particulate matter may not be immediately noticeable. This may be especially problematic in local use contexts, like the City of Keene, where participants did not associate any health risk with biodiesel exposure, even after being informed of the organic carbon results. Further research in the speciation of the organic carbon is a recommendation from this study. Other researchers have stressed that higher soluble organics in biodiesel indicate a pressing research need to conduct more long term health effects research for biodiesel.

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