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Publication Date

2010-03-01

University of California Transportation Center
UCTC Research Paper No. UCTC-2010-10 (renumbered as UCTC-FR-2010-10)

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March 2010

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Word Count: 5,446 + 8 *250 (3 figures and 5 tables) = 7,446 words.

ABSTRACT

The goal of this paper is to estimate the health impacts resulting from exposure to PM and NO_x emitted by train operations in the Alameda corridor, a crucial rail link that serves the Ports of Los Angeles and Long Beach, also known as the San Pedro Bay Ports (SPBP). We link a pollutant dispersion model (CalPUFF) to a health benefits assessment model (BenMAP) to discover population-based health impacts of PM and NO_x emissions from train operations (switching and line haul). After analyzing year 2005 as our baseline, we consider two scenarios that correspond to switching to Tier 2 and Tier 3 locomotives. We find that mortality from PM exposure accounts for the largest health impacts, with health costs in excess of $40 million annually. A shift to Tier 2 locomotives would save approximately half of the annual health costs but the benefits of shifting from Tier 2 to Tier 3 locomotives would be much smaller. This assessment is only partial, however, because of gaps in available health data. To our knowledge, this is the first application of BenMAP to conduct a health assessment at the county level.

INTRODUCTION

The contiguous Ports of Los Angeles and Port of Long Beach in Southern California, also known as the San Pedro Bay Ports (SPBP), are vital to the nation and to California's economy: according to a 2007 economic impact, the SPBP complex handles over 40% of the nation's containerized cargo import traffic (1); moreover, in 2007 approximately 886,000 California jobs were related to SPBP activities, which generated over $6.7 billion in state and local tax revenues (2). Although container traffic SPBP dipped below 2005 levels with the current economic crisis, the SPBP is expected to expand again once the economy recovers.

The transportation of goods to and from the SPBP, however, also creates congestion on local roads and freeways, and it generates large amounts of air pollutants, particularly particulate matter (PM) and nitrogen oxides (NO_x). Air pollution from the SPBP originates from sources along the coast (ships), within the ports (via heavy equipment that moves containers), and on land (as diesel locomotives and large diesel trucks transport containers to and from the SPBP). In particular, the SPBP is served by the Alameda corridor, a major rail-line that currently carries approximately 50 trains per day, flanked by the I-110 and I-710 freeways, which both carry thousands of trucks per day. These vital links connect the SPBP complex to railyards and freight terminals located along the corridor, near downtown Los Angeles, or in the Inland Empire.

According to the draft Emission Reduction Plan for Ports and International Goods Movement in California published by the California Air Resources Board (3), on a typical day, more than 400 tons of NO_x are emitted from ports and goods movement activities in California, which represents 10% of the state total. Diesel particulate matter (DPM) emissions are also a problem: according to the South Coast Air Quality Management District's MATES III study, diesel exhaust contributes approximately 84% of total toxic emissions in the region (4).

Although the economic benefits of SPBP activities are widespread, the resulting air pollution affects primarily people who live and work around the I-110 and I-710 freeways, and along the Alameda corridor. According to the public health literature (5, 6), these communities are at increased risk of respiratory problems, cancer, and death. Indeed, previous studies suggest that pollutant concentrations near sources are elevated (7) and one recent study finds that PM concentrations increase from 10 to 50 percent after the passage of a locomotive (8). Given the width of the Alameda Corridor and the volume of freight movement, air quality and health impacts of freight operations in the corridor could be extensive. Estimates of air pollution in the area are often quite crude, however. For trucks, pollutant emissions are typically calculated without accounting for actual traffic stop-and-go conditions as micro-simulation has not been widely adopted yet to study the environmental impacts of traffic; for trains, emission estimates typically rely on fuel use to quantify the amount of pollutants released (9). Moreover, we could not find any rigorous study of the health impacts of SPBP in the Alameda corridor, which have been a source of controversy for years leading to the adoption by the SPBP of extensive (and expensive) measures to improve air quality.

This paper starts to bridge this gap by analyzing the health impacts of PM and NOx emissions resulting from train operations in the Alameda Corridor (see Figure 1 for a map of our study area). Building on a previous study (10), we analyze 2005 train emissions as a baseline (Scenario 1), and then estimate the benefits of switching to Tier 2 (Scenario 2) and Tier 3 (Scenario 3) locomotives for both line haul and switching operations. Our work reveals that the health impacts of PM and NOx emissions from train operations in the Alameda corridor are substantial but a number of data gaps need to be addressed before a complete picture can be obtained. Although we focus on the SPBP, our methodology is widely applicable.

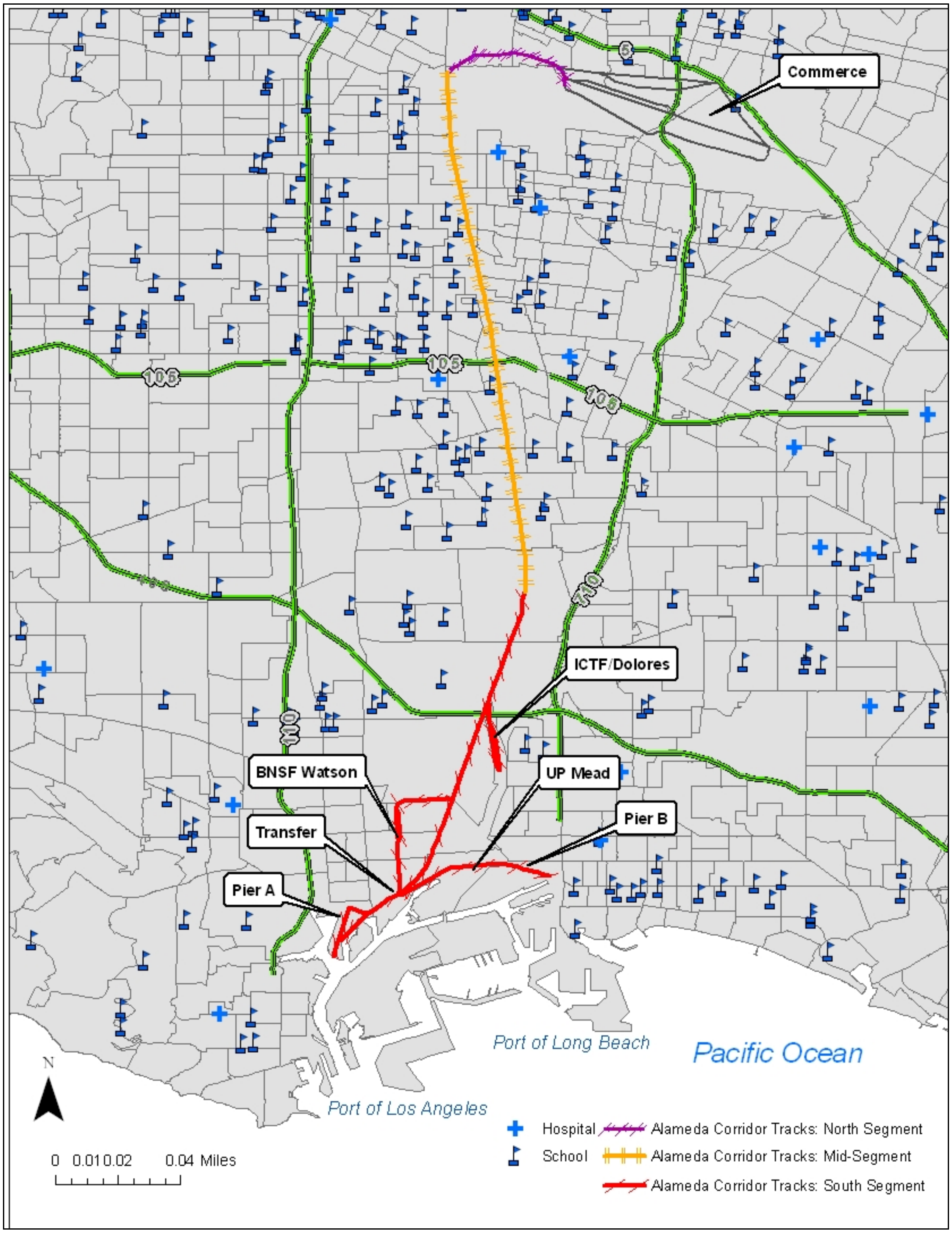

Figure 1. Study Area

BACKGROUND

The SPBP Ports are served by three railroads: Burlington Northern and Santa Fe (BNSF); Union Pacific (UP); and Pacific Harbor Line (PHL). The first two are Class 1 railroads that provide line haul service (movement of cargo over long distances) to the Port.[1] By contrast, PHL is a much smaller, Class 3, railroad that focuses on switching operations (the assembly and disassembly of trains) in and around the Ports. It was created in 1998 to take over the Harbor Belt Line (HBL), as the Alameda Corridor was nearing completion.

Almost all locomotives in the U.S. come from two manufacturers: General Electric Transportation Systems and Electro-Motive Diesel (EMD). Their lifetime can reach 40 to 50 years but they are remanufactured periodically to retrofit their engines. Locomotives used around the SPBP are diesel-electric: they use a diesel engine to power electric motors that drive the wheels, so the speed of the diesel engine is not related to the speed of the locomotive. Instead, diesel engines in locomotives operate at a series of steady-state points, known as notches; typically, there are eight notches for power settings, one or two idle settings, and one or two settings for dynamic braking. Emission measurements from locomotives are made at each notch setting in terms of an emissions rate (e.g., grams per hour), and average emissions for a locomotive are computed from a duty cycle assumed to represent normal field operations. The average emission rate from a locomotive can then be computed based on the relative time spent in each notch setting, either on a brake-specific basis (in terms of an emission rate per unit power output), or on a fuel specific basis (as an emission rate per unit of fuel consumed).

Emissions regulations

Locomotives emissions were regulated only recently, either by the U.S. Environmental Protection Agency (U.S. EPA) or by the California Air Resources Board (ARB).

The first emission regulations [63 FR 18997-19084] were adopted on December 17, 1997 and became effective in 2000 (11). These regulations require that locomotives first built after 1973 meet specific emissions standards when they are remanufactured; this is referred to as Tier 0. There are two other standards for newly manufactured locomotives: Tier 1 applies to locomotives manufactured between 2002 and 2004, and Tier 2, applies to locomotives and locomotive engines manufactured in 2005 and later. Tier 0-2 standards are met by changing engine design, without using exhaust gas after-treatment

Increasing concerns about the pollution impacts of locomotives led to more regulatory activity recently. In May 2004, the U.S. EPA introduced new requirements for off-road diesel fuel that should decrease by 99 % allowable sulfur levels in locomotive fuel. Then, in June 2005, the Air Resources Board (ARB) entered into an agreement with UP and BNSF to cut by 20% locomotive diesel PM emissions near railyards (12).

More recently, a U.S. EPA regulation signed on 14 March 2008 introduced more stringent requirements [73 FR 88 25098-25352]. First, it created more stringent emission standards for remanufactured Tier 0-2 locomotives. Second, it provisioned for clean switching locomotives, and introduced requirements for idle reduction for all locomotives. Finally, it created two new tiers: Tier 3 emission standards for new locomotives starting in December 2011, and Tier 4 standards in 2015 for newly-built engines based on the application of high-efficiency after-treatment technology (13). When fully implemented, it should reduce locomotive PM and NOx emissions by as much as 90% and 80% respectively (14).

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[1] Class 1 railroads have operating revenues over $346.8 million (2006); Class 3 railroads have annual operating revenues under $40 million and less than 350 mi of tracks (www.ibisworld.com/industry/default.aspx?indid=1133).

EMISSION ESTIMATION

Line haul emissions

For modeling emissions from line haul activities, we divided the Alameda Corridor into three segments (north, mid-corridor, and south segment) characterized by different speed limits and lengths (see Table 1).

Table 1. Estimated line haul emissions in the study area

Segment	Segment Length (mi)	Speed Limit (mph)	Assumed Notch	PM Emission Factor (g/hr)	PM Emissions (metric ton/year)	NO_x Emission Factor (g/hr)	NO_x Emissions (metric ton/year)
South segment	8	25	3	427	9.6	7267	163.0
Mid segment	10	40	5	348	6.1	25584	448.2
North segment	2	25	3	427	2.4	7267	40.7
Total	20	NA	NA	NA	18.1	NA	651.9

Notes: Each train is assumed to consist of four Tier1 locomotives and travels at the speed limit for each section. Moreover, we assume two trains per hour around the clock, every day of the year.

Table 2. Estimated railyard emissions in the study area

Railyard (Railroad)	Area (acres)	PM (metric tons/year) Trains only	PM (metric tons/year) All activities	NO_x (metric tons/year) Trains only	NO_x (metric tons/year) All activities
Combined Commerce [Commerce (UP) + Hobart (BNSF) + Eastern (BNSF) + Sheila (BNSF)]	530	13.0	41.2	113.9	797.3
ICTF/Dolores (UP)	233	1.2	8.1	50.1	351.0
Wilmington-Watson (BNSF)	17	0.4	1.3	3.6	25.2
Transfer (PHL)	6	0.1	0.3	1.2	8.4
UP Mead (PHL)	10	0.3	1.0	2.2	15.4
Pier A (PHL)	23	0.6	1.9	5.0	35.0
Pier B (PHL)	14	0.3	1.0	3.1	21.7

Notes:

- PM emissions for the combined Commerce railyards and for ICTF/Dolores come respectively from (15) and (16). PM emissions for other yards were assumed to have the same rate of emissions per unit area and per unit time as Commerce Eastern. Railyard areas were measured using Google Earth.
- NO_x emissions for ICTF/Dolores are from (15). Other yards were assumed to have the same rate of NO_x emissions per unit area and per unit of time as ICTF/Dolores.
- "All activities" includes all locomotive emissions, as well as emissions from drayage trucks, cargo handling equipment, as well as heavy equipment and transport refrigeration units (17).

Based on conversations with representatives from PHL and from the Ports, we assumed that line haul is primarily done by Tier 1 locomotives, which are in notch five on the mid-corridor segment, and in notch three on the other two segments. We then obtained the corresponding representative emission factors from (18), which is used in the State Implementation Plan to prepare locomotive emission inventories. After that, we calculated PM and NO_x emissions based on four locomotives per train. To find total annual emissions of these pollutants, we assumed two trains per hour around the clock. This is a slight overestimate for 2005 since the Alameda Corridor Authority recorded an average of 47 trains per day that year (19). A summary of line haul emissions is presented in Table 1.

Railyard emissions

As shown on Figure 1, seven railyards are associated with freight transportation from the SPBP, but two of them (the Commerce railyards, which includes UP Commerce, BNSF Hobart, BNSF Mechanical Sheila and BNSF Commerce Eastern, and the combined ICTF/Dolores railyard) are much larger than the others. Our starting point for estimating emissions is a series of recent health risk assessments of major California railyards conducted for the EPA (17). These studies only covered PM and NO_x emissions from the two main railyards in our study area, however. Therefore, to estimate emissions from the five smaller railyards in our study (Watson, Transfer, Mead, Pier A, and Pier B), we assumed their emissions to be proportional to those of the Commerce railyard, based on their area measured using Google earth. A summary of railyard emissions is presented in Table 2. Note, however, that our dispersion analysis is restricted to “train only” emissions.

AIR DISPERSION MODELING

To estimate the dispersion of various air pollutants, we relied on the CALPUFF model, which is a generalized non-steady-state air quality modeling system initially designed by Sigma Research Corporation for the California Air Resources Board (CARB). This model has been improved over time to meet the needs of various federal agencies. In 1998, the U.S. EPA recommended CALPUFF for estimating air quality impacts for the National Ambient Air Quality Standards (NAAQS) and prevention of significant deterioration (PSD) increments. This non-steady-state puff dispersion model simulates the effects of time- and space-varying meteorological conditions on pollution transport, transformation, and removal. CALPUFF can be applied for long-range transport and complex terrain.

More specifically, we relied on CALPUFF View 5.8, which includes an improved user interface. This software has three main components: CALMET, CALPUFF, and CALPOST. CALMET is a meteorological model that creates hourly temperature and wind fields on a three-dimensional grid. CALPUFF is a transport and dispersion model that advects “puffs” of pollutant from specific sources while simulating dispersion and transformations. Finally, CALPOST processes output files from CALPUFF to generate final results. In addition, CALPUFF View provides a variety of pre-processing programs that interface with 2005 MM5 datasets, which integrate terrain, land use, meteorological data. The MM5 (National Center for Atmospheric research/Penn State Mesoscale Model) is a regional weather model used for creating weather forecasts and climate projections (20).

Pollutants considered

We focus here on two criteria pollutants associated with train operations: PM (particulate matter) and NO_x (Nitrogen oxides).

Indeed, according to CARB studies (17), diesel PM accounts for approximately 80% of the potential ambient air toxic cancer risks in California and South Coast Air Basin residents are exposed to higher risks than average. Exposure to diesel PM is hazardous, particularly to children (their lungs are still developing) and to the elderly. A key concern is that approximately 92% by mass of diesel PM particles have a diameter under 2.5 microns (21), so they can penetrate deep into the lungs and carry toxics into the bloodstream. A number of population-based studies around the world have demonstrated a strong link between elevated PM levels and premature deaths (22, 23, 24), increased hospitalizations for respiratory and cardiovascular causes, asthma and other lower respiratory symptoms, as well as acute bronchitis (25).

According to the CARB (9), NO_x causes a wide variety of health and environmental impacts as it reacts with different compounds to create harmful derivatives. First, NO_x can react with volatile organic compounds (VOCs) in the presence of sunlight to create ground level ozone, which can be transported by winds far from its sources. This compound can damage lung tissue and reduce lung function in children, people with lung diseases, and people who work or exercise outside. In addition, ozone can damage vegetation and reduced crop yields. Second, NO_x can react with sulfur dioxide and other airborne substances to form acids which may be deposited as rain, fog, snow or dry particles. This phenomenon can cause pollution hundreds of miles away. It can damage cars, buildings, and causes lakes and streams to become acidic and unsuitable for many fish. Third, NO_x can react with ammonia, moisture, and other compounds to form nitric acid, which can damage the respiratory system and even cause premature death. Finally, nitrate particles and nitrogen dioxide can reduce visibility in urban areas.

In the following section, we do not distinguish between NO_x and NO_2 because almost all NO_x at concentrations below 80 $\mu g/m^3$ turns to NO_2 (26).

ESTIMATING HEALTH IMPACTS

To estimate the human health effects and economic impacts associated with changes in ambient air pollution, we relied on BenMAP, which was originally designed by the U.S. EPA to analyze large-scale air quality regulations such as the National Ambient Air Quality Standards for Particulate Matter (2006) and the Locomotive Marine Engine Rule (2008).

To estimate human health effects, BenMAP requires an estimate of change in ambient air pollution generated by an air quality model. It then estimates specific health effects (health points) resulting from changes in pollution concentration using a health impact function, also called concentration-response (C-R) function in epidemiology studies. Finally, BenMAP applies these specific health effects to the exposed population. Conceptually, this process can be summarized by the relationship (27):

$$\text{Health Effect} = \Delta(\text{Air Quality}) \times \text{Health Effect} \times \text{Exposed Population} \times \text{Health Baseline Incidence}, \qquad \textbf{\textit{(1)}}$$

where:

- Δ(Air Quality) is the difference between the baseline air pollution level and a change in air pollution level caused by a policy.
- The health effect estimates the percentage change in an adverse health effect due to a one unit change in ambient air pollution, based on epidemiological studies.
- The exposed population is the number of people affected by the air pollution reduction.
- The health baseline incidence rate estimates the average number of people who die in a given population over a given period of time.

To calculate the economic value of human health effects, BenMAP multiplies the change in health effects by an estimate of the economic value per case. The latter can be estimated by different methods. For example, the value of an avoided premature death is generally calculated using the Value of Statistical Life (VSL), which is the dollar amount people are willing to pay to reduce the risk of premature death by one unit. For other health effects, medical costs can be used, for example.

Air Pollution Monitoring and Modeling

The air pollutant monitoring data for 2005 is based on a database of ambient air pollution data collected from nine EPA standard monitors located in Los Angeles County. The concentrations of $PM_{2.5}$ and NO_x are reported as a 24-hour average. To proceed from point-based monitoring data to estimates of pollutant concentrations in the study area, BenMAP relies on interpolation. The default method, which we rely on for our results, is the “closest monitor” method, which simply assigns to a point the value of the closest monitor.

Baseline Incidence and Concentration-Response Functions

BenMAP provides an extensive list of concentration-response functions (C-R function) for various health end points, such as mortality or asthma. A C-R function measures the change in a health end point of interest resulting from a change in the concentration of a given pollutant. It can be written:

$$f(\Delta Q, I, P) = (1 - \exp(-\beta \bullet \Delta Q)) \bullet I \bullet P, \tag{2}$$

where:

- ΔQ is the estimated change in pollutants concentration;
- *I* is the incidence, i.e., the baseline mortality incidence rate from the EPA database;
- β is the parameter of the exponential distribution defined by

  $$\beta = \frac{\ln(RR)}{\Delta Q} \tag{3}$$

  In that equation, RR is the relative risk (or risk ratio) of the health end point considered. RR for an event can be defined as the ratio of the probability of an event occurring in the exposed group versus a non-exposed group.
- P is the potentially affected population. To estimate P, we used the 2005 Census block-level data and the PopGrids software to construct specific population grids matching the appropriate age-specific population from the overall population database for Los Angeles County.

In this study, we selected endpoints based on likely severity but also on data availability. Some C-R functions are based on studies for other cities and others were estimated over time periods that do not include 2005. For example, no asthma exacerbation function was provided either for Los Angeles County for 2005 so we used asthma exacerbation functions from a 2008 multi-city study (28).

$PM_{2.5}$ exposure endpoints

For $PM_{2.5}$, we selected mortality and chronic bronchitis as our endpoints.

For premature mortality, we considered several C-R functions. The first applies to adults aged 30 to 65; it is based on a 2005 Los Angeles study (29); its relative risk (RR) is 1.17 for a 10 $\mu g/m^3$ change in average annual $PM_{2.5}$ exposure. To capture $PM_{2.5}$ mortality impacts on older adults, we also used a pooled C-R function that applies to people aged 30 and more; its RR is 1.11 for people aged 30 years and more for a 10 $\mu g/m^3$ change in average annual $PM_{2.5}$ exposure (30). In addition, a 2006 study by Woodruff et al. examined mortality associated with $PM_{2.5}$ for infants aged between one and 12 months (31); they report a RR of 1.07 for a 10 μg/m^3 change in average annual $PM_{2.5}$ exposure.

Another health outcome we considered is chronic bronchitis, a progressive chronic lung disease characterized by mucus in the lungs, which causes persistent wet coughing and disrupts oxygen exchange between air and blood in the lungs (27). It is derived from the only available chronic bronchitis study that examines directly the impact of $PM_{2.5}$ (30); its RR is 1.14 for a 10 μg/m^3 change in average annual $PM_{2.5}$ concentration.

NO_x exposure endpoints

For NO_x, hospitalization information from different endpoints, such as asthma or chronic lung disease, was obtained from the National Center for Health Statistics' (NCHS) National Hospital Discharge Survey (NHDS) (28). The survey collects data on short-stay (less than 30 days) hospitals, patient characteristics, diagnoses, and medical procedures. C-R functions for asthma-related and chronic lung disease-related hospital admission are already included in BenMAP's health impact database; they rely on various studies (27).

The asthma exacerbation health impact functions are based on acute respiratory health effects of air pollution on children with asthma in US inner cities (28). The study analyzed data from 861 children age 5-12 years old with asthma in several US inner-city communities (but not Los Angeles). The endpoints we selected are: missed school day, night time asthma, slow play and more than one symptom. These functions are already included in BenMAP's health impact database.

Health Valuation Functions

Health valuation functions available in BenMAP give a cost value for each case of a specific health effect.

For $PM_{2.5}$ mortality, we used the value of a statistical life, which is a summary measure for the value of a marginal change in mortality risk. The mean value of avoiding one statistical death is approximately $ 5.5 million in year 2000 dollars; this value was converted to year 2005 dollars using the Consumer Price Index (CPI) for medical care (32).

For chronic bronchitis caused by $PM_{2.5}$ exposure, we relied on cost of illness (COI) functions derived from estimates of annual medical costs and lost earnings (33); they do not include the cost of pain and suffering in the valuation estimation. As chronic bronchitis is expected to last a lifetime, its COI is the present value of a medical expenditures and lost income discounted with a 3 percent rate.

Let us now consider health valuation functions for NO_x. The COI for hospital admission from asthma and chronic lung disease related to NO_x are available in the BenMAP valuation database. It includes hospital charges and opportunity cost of time spent in the hospital represented by lost daily wage. For asthma exacerbation endpoints, we use the same valuation function: it relies on a recent study (34).

SCENARIOS

In this study, we compare the health impacts of three scenarios. The baseline scenario assumes that all locomotives that operate in the Alameda Corridor belong to Tier 1. Scenario 2 consists in shifting from Tier 1 to Tier 2 locomotives and scenario 3 replaces all Tier 1 with Tier 3 locomotives, all for both switching and line haul.

For the maximum of the seasonal average pollution, Table 1 also provides the percentage change from the baseline to Scenario 2 and from the baseline to Scenario 3. We note that Scenario 2 cuts PM emissions by over 50%, but NO_x emissions by only approximately 26%; by contrast, Scenario 3 achieves a relatively larger reduction of NOx emissions compared to PM emissions. These percentage changes in emissions are derived from 2008 EPA emission standard for locomotives.

We also note that the summer has the highest worst day maximum for both NO_x and $PM_{2.5}$, (74.97 and1.96 ug/m^3 respectively), while the fall has the highest seasonal average maximum (7.99 and 0.88 ug/m^3 respectively); these differences are entirely due to meteorological conditions as train activity is assumed constant throughout the year.

RESULTS

Table 3 reports the worst day maximum and the maximum of the seasonal average pollution concentrations for both PM and NO_x for the baseline and the two scenarios considered. We use the seasonal average concentration for estimating health impacts because we are interested in health impacts from long-term exposure to typical daily conditions.

Table 3. Seasonal Maximum and Average 24hr average Concentrations (from CalPUFF)

		Worst day maximum		Seasonal average maximum		% change from baseline (seasonal average max)	
		NO_x (µg/m^3)	$PM_{2.5}$ (µg/m^3)	NO_x (µg/m^3)	$PM_{2.5}$ (µg/m^3)	NO_x	$PM_{2.5}$
Winter	Baseline	73.23	1.68	6.48	0.70		
	Scenario2	54.44	0.84	4.78	0.35	-26.3%	-50.3%
	Scenario3	54.43	0.65	3.13	0.27	-51.8%	-61.1%
Spring	Baseline	42.99	1.60	4.35	0.46		
	Scenario2	31.94	0.80	3.21	0.23	-26.3%	-50.3%
	Scenario3	31.94	0.62	2.60	0.18	-40.3%	-61.1%
Summer	Baseline	74.97	1.96	4.63	0.49		
	Scenario2	55.70	0.98	3.41	0.24	-26.4%	-50.3%
	Scenario3	55.69	0.76	2.74	0.19	-40.9%	-61.2%
Fall	Baseline	27.67	1.86	7.99	0.88		
	Scenario2	20.56	0.93	5.88	0.44	-26.3%	-50.3%
	Scenario3	20.43	0.72	3.85	0.34	-51.8%	-61.1%

Let us first start with results for NO_x. For this pollutant, we considered six different health outcomes, based on the health impact functions available in BenMAP and in the literature. Two of these health impact functions were estimated at the Los Angeles County level: hospital admissions from asthma and chronic lung diseases. At the level of pollutants considered, however, they yielded only low damages compared to the other health impacts (under 5 cases and $60,000 in costs for all scenarios considered) so details of their estimation is omitted.

Table 4. Some seasonal health impacts from NO_x exposure

Period	Scenario	Missed School Days	Nighttime Asthma	One or more Symptoms	Slow Play	Total Value ($2005)
Winter	Baseline	$0.24	$0.45	$0.65	$0.66	**$2.00**
		(1,229)	(2,339)	(3,375)	(3,389)	**(10,332)**
	Scenario2	$0.18	$0.34	$0.48	$0.49	**$1.48**
		(913)	(1,735)	(2,504)	(2,515)	**(7,666)**
	Scenario3	$0.17	$0.32	$0.46	$0.46	**$1.40**
		(861)	(1,637)	(2,362)	(2,372)	**(7,233)**
Spring	Baseline	$0.17	$0.32	$0.46	$0.46	**$1.40**
		(861)	(1,637)	(2,361)	(2,372)	**(7,231)**
	Scenario2	$0.12	$0.24	$0.34	$0.34	**$1.04**
		(639)	(1,214)	(1,751)	(1,759)	**(5,362)**
	Scenario3	$0.12	$0.22	$0.32	$0.32	**$0.98**
		(604)	(1,148)	(1,655)	(1,663)	**(5,070)**
Summer	Baseline	$0.19	$0.36	$0.52	$0.52	**$1.59**
		(976)	(1,856)	(2,678)	(2,689)	**(8,199)**
	Scenario2	$0.14	$0.27	$0.38	$0.39	**$1.18**
		(725)	(1,377)	(1,987)	(1,996)	**(6,085)**
	Scenario3	$0.13	$0.25	$0.36	$0.37	**$1.11**
		(685)	(1,301)	(1,876)	(1,885)	**(5,747)**
Fall	Baseline	$0.30	$0.58	$0.83	$0.84	**$2.55**
		(1,568)	(2,986)	(4,310)	(4,326)	**(13,189)**
	Scenario2	$0.23	$0.43	$0.62	$0.62	**$1.90**
		(1,165)	(2,216)	(3,198)	(3,211)	**(9,790)**
	Scenario3	$0.21	$0.40	$0.58	$0.59	**$1.79**
		(1,098)	(2,088)	(3,013)	(3,026)	**(9,225)**
Year 2005	Baseline	$0.90	$1.71	$2.46	$2.47	**$7.54**
		(4,634)	(8,817)	(12,725)	(12,776)	**(38,952)**
	Scenario2	$0.67	$1.27	$1.83	$1.84	**$5.60**
		(3,441)	(6,543)	(9,439)	(9,481)	**(28,903)**
	Scenario3	$0.63	$1.20	$1.72	$1.73	**$5.28**
		(3,248)	(6,174)	(8,907)	(8,947)	**(27,275)**

Notes. These health impacts are for children aged 5 to 12; they are based on multi-city studies. All dollar amounts are in million of 2005 dollars. A number in parentheses underneath a dollar amount is the corresponding number of cases. Although they are incomplete, the health results for NO_x emitted by train operation suggest that its impacts are substantial but limited. Total values may appear slightly off because the table shows only two significant digits.

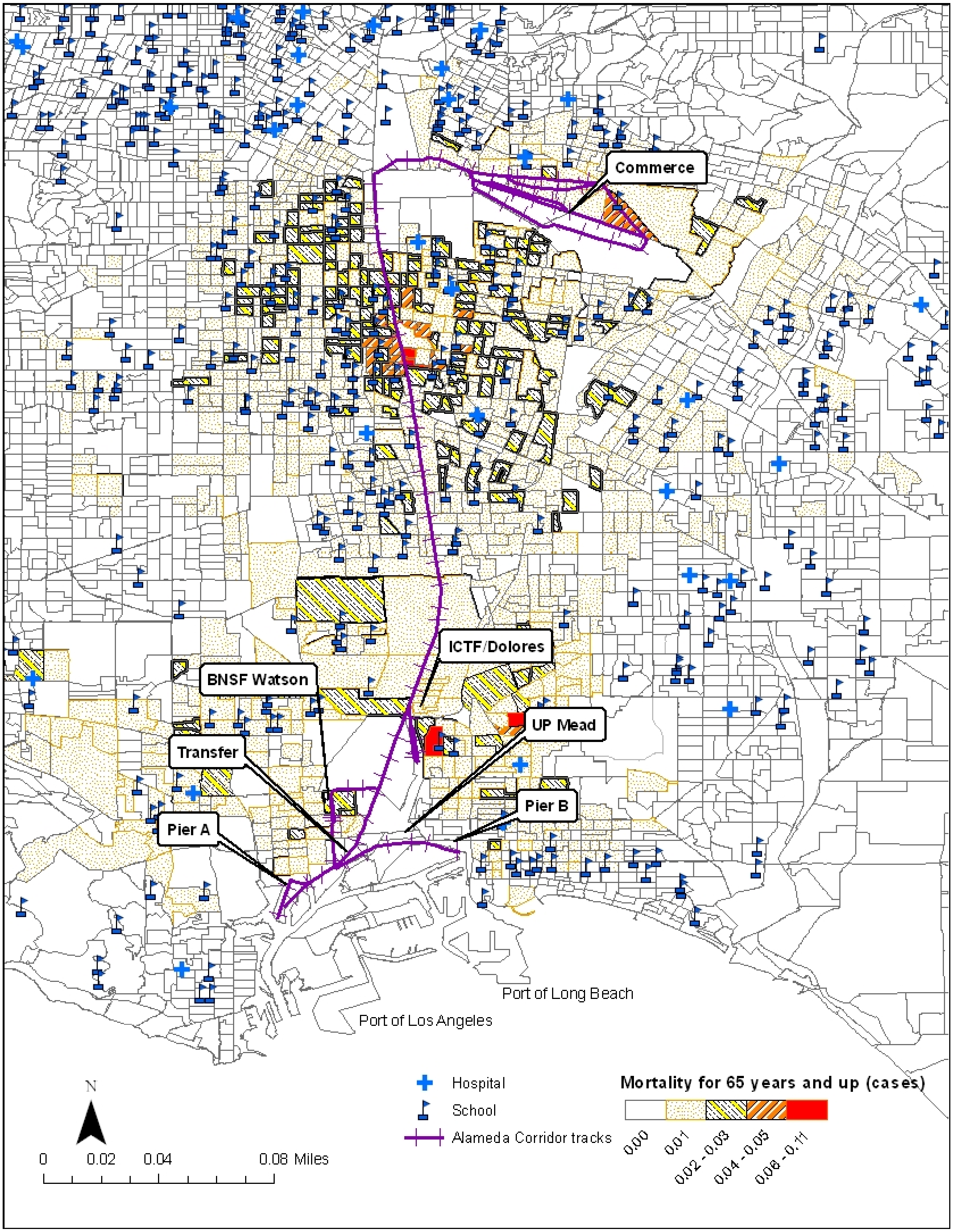

Figure 2. Number of statistical lives lost every year because of PM$_{2.5}$ exposure from trains.

Note: results were obtained at the block-group level**.**

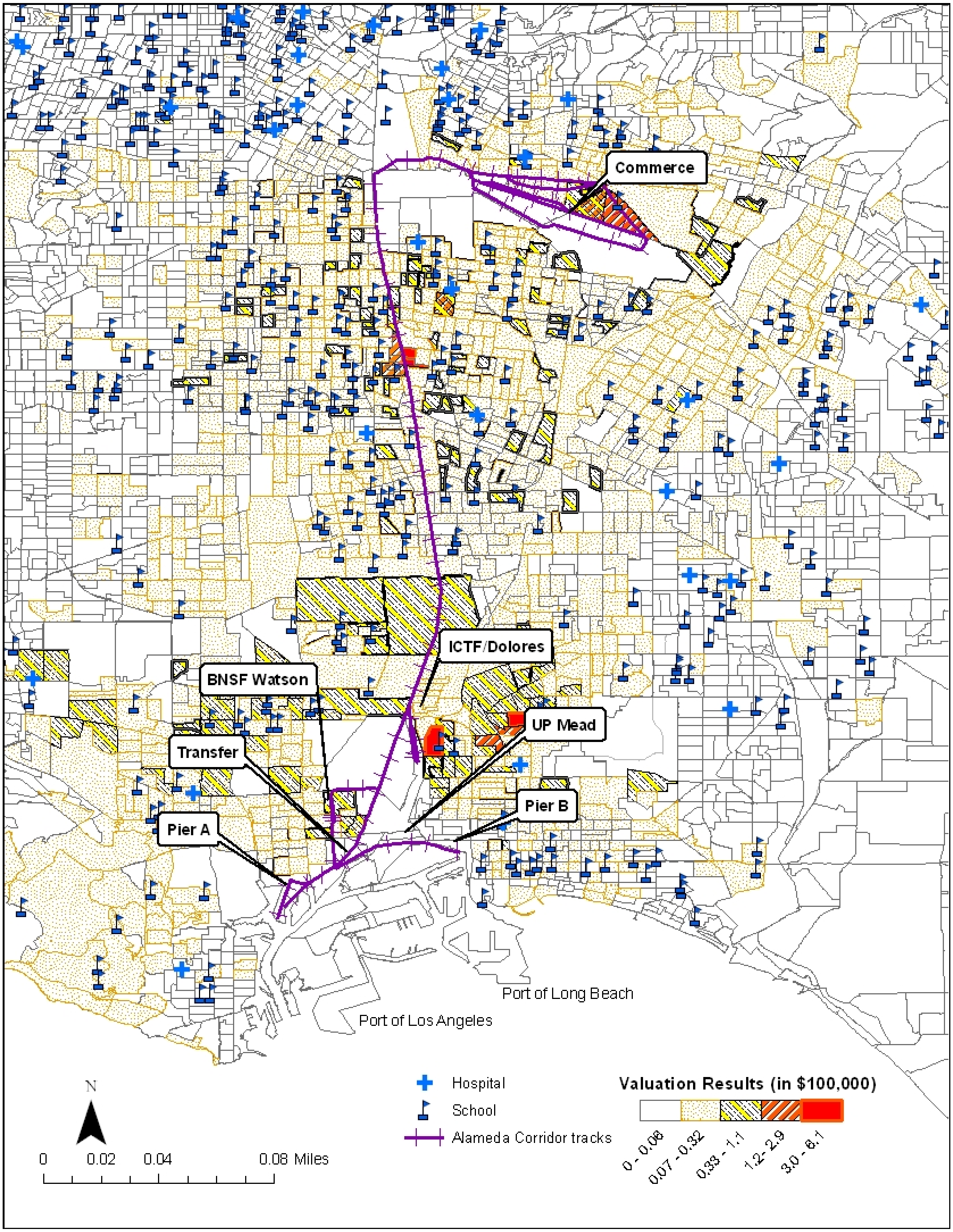

Figure 3. Value of statistical lives lost every year because of $PM_{2.5}$ exposure from trains.

Note: results were obtained at the block-group level**.**

Table 5. Some seasonal health impacts of PM$_{2.5}$ from PM exposure

Period	**Scenario**	**PM$_{2.5}$ Mortality Age: 30-65**	**PM$_{2.5}$ Mortality Age: 65 and over**	**Chronic Bronchitis**	**Total Value ($2005)**
Winter	Baseline	$4.47	$5.15	$0.22	**$9.84**
		(0.66)	(0.76)	(0.68)	
	Scenario2	$2.19	$2.51	$0.11	**$4.81**
		(0.32)	(0.37)	(0.34)	
	Scenario3	$1.98	$2.25	$0.10	**$4.34**
		(0.29)	(0.33)	(0.30)	
Spring	Baseline	$3.45	$4.07	$0.17	**$7.69**
		(0.51)	(0.60)	(0.53)	
	Scenario2	$1.67	$1.93	$0.08	**$3.68**
		(0.24)	(0.28)	(0.26)	
	Scenario3	$1.50	$1.73	$0.08	**$3.31**
		(0.22)	(0.25)	(0.23)	
Summer	Baseline	$4.21	$5.16	$0.21	**$9.59**
		(0.62)	(0.76)	(0.64)	
	Scenario2	$2.10	$2.51	$0.11	**$4.72**
		(0.31)	(0.37)	(0.32)	
	Scenario3	$1.88	$2.25	$0.09	**$4.22**
		(0.28)	(0.33)	(0.29)	
Fall	Baseline	$6.40	$7.43	$0.32	**$14.14**
		(0.94)	(1.09)	(0.97)	
	Scenario2	$3.17	$3.65	$0.16	**$6.98**
		(0.47)	(0.54)	(0.48)	
	Scenario3	$2.82	$3.22	$0.14	**$6.18**
		(0.41)	(0.47)	(0.43)	
Year 2005	Baseline	$18.52	$21.80	$0.93	**$41.25**
		(2.72)	(3.20)	(2.83)	
	Scenario2	$9.12	$10.60	$0.46	**$20.18**
		(1.34)	(1.56)	(1.39)	
	Scenario3	$8.18	$9.46	$0.41	**$18.05**
		(1.20)	(1.39)	(1.25)	

Notes. These health impacts are based on multi-city studies. All dollar amounts are in million of 2005 dollars. A number in parentheses underneath a dollar amount is the corresponding number of cases. Total values may appear slightly off because the table shows only two significant digits.

Results for the other four health outcomes were estimated based on data developed in studies that covered Boston, Chicago, Dallas, New York, Seattle, and Tucson. They focus on asthma exacerbation in children aged 5 to 12 years old; four conditions are considered: missed school days, nighttime asthma, slow play, and one or more symptoms. For simplicity, we assume that these symptoms were experienced by different children. As shown in Table 4, the number of cases and their associated social costs ranged from \$5.3 to \$7.5 million. Although the number of people affected is large, going from Tier 1 (the baseline) to Tier 2 locomotives (Scenario 2) would save \$1.94 million per year, while switching from Tier 2 (Scenario 2) to Tier 3 (Scenario 3) locomotives would save only an additional \$320,000 (=\$5.6-\$5.28) annually.

Results for PM are summarized in Table 5 and illustrated on Figures 2-3, which show the annual number of statistical lives lost and the corresponding costs at the block group level because of $PM_{2.5}$ exposure from trains. The health outcomes considered include mortality from all causes related to PM exposure and chronic bronchitis. Not all age groups are represented because of the availability of health impact functions. We also analyzed mortality for infants (children younger than 1 year) but the number of cases and the corresponding dollar amount were low so they are not reported here. As for NO_x, we observe strong seasonal variations, which are entirely due to climatic conditions. Fall is the worst season in terms of health impacts, followed by summer and winter (which are fairly similar); by contrast, spring has the lowest health impacts not only for mortality but also for chronic bronchitis linked to PM exposure.

Mostly as expected from our emission estimates, Figures 2 and 3 show that the mortality cases resulting from PM exposure are located around the two major railyards (Commerce and ICTF/Dolores), but also in one area of the Alameda corridor where land use and prevailing wind patterns tend to concentrate pollution.

A comparison of Tables 4 and 5 shows that the main health income is mortality from PM: it results in approximately 6 cases per year with a corresponding cost in excess of \$40 million; elderly people (65 years old and over) are primarily affected with 3.20 cases per year. Shifting from Tier 1 (Baseline) to Tier 2 (Scenario 2) locomotives would cut health costs in half, whereas upgrading from Tier 2 (Scenario 2) to Tier 3 (Scenario 3) would only save only a small additional fraction (\$2.1 million).

CONCLUDING REMARKS

To our knowledge, this is the first attempt at estimating the emission, the dispersion, and the health impacts of PM and NO_x train emissions in a major transportation corridor. According to our U.S. EPA contacts, this is also the first application of BenMAP at the county level, which impacted our work slightly because only a limited set of health functions were available for our analyses. We find seasonal effects and complex spatial dispersion patterns in the dispersion of both PM and NO_x, which result from land use and changing wind directions. Based on available health functions, health impacts from PM are significantly larger than those of NOx. Although estimated PM concentrations from train operations in 2005 are well below international health standards, they result in annual damages that exceed \$40 million from mortality cases alone. This is five times larger than estimated NOx health impacts, but note that these include only four health outcomes for a small subset of the population (kids aged 5 to 12). Our analyses also show that switching from Tier 1 (our baseline) to Tier 2 locomotives would cut health impacts in half. Switching from Tier 2 (Scenario 2) to Tier 3 (Scenario 3) locomotives would only produce approximately one tenth additional health benefits. More generally, our work shows that it is essential to understand the dispersion and the health impact of air pollutants for policy analysis; just knowing the amount of pollution released is insufficient.

In a companion paper (35), we extend our analysis to drayage trucks operating in the study area. Future work could assess the health impacts of shifting freight transportation from trucks to trains. Our analysis could also be extended to other health outcomes and more subsets of the population provided the necessary health impact functions are available.

ACKNOWLEDGEMENTS

Support from the University of California Transportation Center (Award 65A016-SA5882) is gratefully acknowledged. We thank Eric Shen, Shashank Patil, and Carlo Luzzi from the Port of Long Beach for answering our questions. Finally, we are very grateful to Neal Fann, from U.S. EPA's Air Benefit and Cost Group in Triangle Park, South Carolina; his help was invaluable in learning BenMAP but neither he nor EPA are responsible for any error in our analyses.

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