THE ECONOMICS OF AIR POLLUTION HEALTH RISKS IN RUSSIA: A CASE STUDY OF VOLGOGRAD

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ABSTRACT

A combined health risk assessment, cost-effectiveness analysis, and cost-benefit analysis was undertaken for particulate emissions from stationary source polluters in the city of Volgograd, Russia. Annual particulate-related mortality risks from stationary source polluters were found to be substantial in Volgograd, with an estimate in the range of 960 to 2,667 additional deaths per year in a city of one million. The majority of these risks were created by two major facilities in the northern part of the city. The cost per life saved for several emission reduction projects was found to be quite low. The total net benefits to the city of implementing five of the six projects (leading to roughly a 25 percent reduction in mortality risk) were estimated to be at least $39.5 million in present value terms.

1. INTRODUCTION

Economic theory provides two simple principles for managing health risks. The cost-effectiveness principle suggests that any reduction in health risks should be obtained in a least-cost way. The benefit-cost principle suggests that health risks should be reduced as long as the additional benefits from risk reduction are greater than the additional costs. In practice, cost-effectiveness analysis is used to prioritize possible risk reduction options from least expensive to most expensive per change in risk, while benefit-cost analysis is needed to answer the question "how much should risk be reduced?"

Managing risks to human health from ongoing industrial pollution is a priority for environmental policy in Russia. Given the scarce resources available in Russia for environmental management, it is especially important that these resources be used effectively to reduce health risks. While the basic methods of risk assessment and economic analysis are now known in Russia, however, few empirical analyses have been carried out there that provide useful, practical guidance to environmental policy makers for managing health risks from current industrial emissions. This gap between theory and practice needs to be closed if the concepts of risk assessment, cost-effectiveness analysis, and cost-benefit analysis are to be incorporated into Russian environmental and health policymaking and implementation.

As an initial step toward filling this gap, this paper first presents the results of a health risk assessment in Russia, with risks defined as additional annual mortality risks from particulate emissions from stationary source polluters in the city of Volgograd. Second, cost-effectiveness analysis is applied to several risk reduction options that have been identified for the major particulate emitters in Volgograd. Finally, using a lower-bound estimate of the value of a statistical life in Russia, a benefit-cost analysis is carried out to determine the magnitude of the net benefits to the city of these risk reduction options.

The main results of this paper can be summarized as follows. Annual particulate-related mortality risks from stationary source polluters are substantial in the city of Volgograd, with an estimate in the range of 960 to 2,667 additional deaths per year in a city of one million. The majority of these risks are created by two major facilities in the northern part of the city. The cost per life saved for several emission reduction projects is quite low. The total net benefits to the city of implementing five of the six projects (leading to roughly a 25 percent reduction in mortality risk) are estimated to be at least $39.5 million in present value terms. Morbidity effects of particulates, and particulate damage to buildings, other structures, and ecosystems are not included in the analysis. If included, they would increase the net benefits of particulate reduction significantly.

2. ASSESSING MORTALITY RISKS FROM PARTICULATES IN VOLGOGRAD

Volgograd, previously known as Stalingrad, is an industrial city of approximately one million people on the western bank of the Volga River in southern Russia. It is a long, narrow city, with a distance by road between its northern and southern borders of about 70 km. Following the almost complete destruction of the city's infrastructure, housing, and industry during the Great Patriotic War (World War II), Volgograd was rebuilt during the next two decades. Industrial facilities are scattered throughout the city, although there are major concentrations of industry in the north and south.

Health risk assessment methods began to be used in the United States in the 1970s to estimate risks to human health from specific chemicals and levels of exposure (Andrews 1995). Since that time, a wide variety of risk assessment methods have been developed to address different kinds of risk and different reasons for conducting an assessment.[1] The health risk assessment reported here uses a generally accepted approach recommended by the U.S. Environmental Protection Agency (U.S. EPA) which involves four steps: (1) hazard identification; (2) exposure assessment; (3) dose-response assessment; and (4) risk characterization.[2]

Since the purpose of this study was to provide a practical example for environmental and health policymakers in Russia, three guiding principles were adopted for the risk assessment. First, the analysis used only existing information resources that are likely to be available in many Russian cities so that the process developed for Volgograd can easily be replicated in other parts of Russia. This principle led to a focus on health risks from stationary sources. Emissions data for stationary source air polluters are available because of Russia=s established framework of air pollution permits and charges. Stationary sources are also a primary target of current Russian air pollution laws and regulations, making the results of the risk assessment immediately useful to local and regional policymakers.

Second, the study applied the standard four-step quantitative health risk assessment methodology described above. While many Russian scientists have read about and received some training in modern risk assessment methods as understood outside the former Soviet Union, there have been few attempts at actually applying the methods to concrete problems in Russia. To enhance its value as a guide for future applied research, the Volgograd risk assessment thus followed a standard, internationally-accepted methodology, within the constraints imposed by the need to use only existing, commonly available types of data.

And third, this study makes explicit the link between health risks and the facilities that generate the risks in order to permit subsequent identification of cost-effective policy options for reducing health risks. A direct link between health risks and generators of the risks is necessary to make the results of the risk assessment relevant to the current environmental policy debate in Russian cities. Emissions inventories from stationary source air polluters provided the core pollution data for the health risk assessment, and the link between industrial facilities and health risks was made through modeling the dispersion of air emissions throughout the city. Consistent with the desire to use existing information resources likely to be available in many Russian cities, the standard Russian air dispersion model used for regulatory purposes, such as setting stationary source air pollution limits, was also used to model air dispersion in this study.

Due mainly to modeling and data constraints imposed by these principles, the risk assessment has several important features:

- inhalation is the pathway for which health risks are estimated
- only mortality risks from particulates are considered
- only emissions from large stationary sources are considered.

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[1] Risk assessment is the process of identifying the probability of some unwanted effect on human health. Percival et al. (1992) provide a concise, non-technical introduction to risk assessment and risk management.
[2] U.S. EPA guidance is provided in several documents (see, e.g., U.S. EPA 1996, 1989, 1988). See VHRAWG (1997) for a more complete discussion of the risk assessment analysis as well as the method and results for carcinogenic risks.

The methods and results of the risk assessment are presented below.

Step 1. Hazard Identification

In Russia, stationary sources are regulated under a two-level system of annual pollution limits (tons per year) and charges (rubles per ton of pollutant per year). The annual pollution limit (PDV in Russian) is related to air concentration standards (PDK in Russian) with 20-minute averaging times. Enterprises pay a base charge rate per ton for levels below the PDV level and a higher rate for tons above the PDV level (see Bluffstone and Larson, 1997 for more on such systems). In principle, a few hundred pollutants are included in this system and more than two hundred industrial facilities are regulated in Volgograd. Just 29 of these facilities, however, account for more than 90 percent of total reported stationary source air emissions in the city (in terms of total tons per year in 1995). These 29 facilities, which each reported total annual emissions (gases and solids) greater than 10 tons in their emission inventories for the years 1993, 1994, and 1995, were selected for the study. These 29 facilities are also the largest emitters of particulates in the city.

Table 1 provides a complete list of the 29 facilities included in the Volgograd analysis, along with an indication of each facility's primary production sector, general location in the city (central, northern, or southern zone), and facility code.[3] A fairly wide distribution of industrial sectors is represented by the 29 facilities, including district heating, construction materials, chemicals, ferrous and non-ferrous metals, furniture, and agricultural machinery.

Following the selection of the 29 facilities, facility-level emissions inventories for 1993, 1994, and 1995 were reviewed to determine which year's data should be included in the risk assessment.[4] The purpose was to select a year for which emissions levels are likely to approximate emissions levels in the near future, assuming no major changes to environmental regulations, and a year for which detailed emissions data exist. Emission levels in 1995 were about 40-60 percent lower than 1990 levels. Given the substantial declines in industrial production in Russia and Volgograd between 1990 and 1995, the continuing slump in the local economy, and the expectation that emissions will not fall much lower than 1995 levels, emissions data for 1995 were selected for the analysis. Estimates of risks based on 1995 emission levels can be considered a conservative (low) estimate of future risks from particulates. If output picks up in the future, risks will probably increase as well.

Particulates are the focus of this study because of the well-documented health effects of total suspended particulates (TSP), and specifically of PM_{10} (particulate matter with a diameter of 10 microns or less) (see, e.g., Schwartz, 1994; Wilson and Spengler, 1996). Particulates are a serious environmental health concern in many major industrial cities and, given limited resources, are a sensible starting point for a health risk assessment in a city like Volgograd.[5] Cancer risks, on the other hand, were estimated to be small in Volgograd and to be greatly outweighed by risks from particulates. In Volgograd, TSP encompasses a high percentage of the total noncarcinogens in enterprises' emissions inventories. Table 2 provides the TSP emissions inventory for the 29 facilities in the study.

Current Russian law does not regulate particulates as an aggregate category (such as TSP or PM_{10}). As a result, emissions inventories for Russian facilities do not include a pollutant called "total particulates" or

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[3] Due to the large area encompassed by Volgograd's city limits and its large number of enterprises, the Working Group defined the Volgograd study area boundary as the currently defined city limits.

[4] All emissions data utilized in this study are based on engineering estimates of chemical emission rates, which are the only data commonly available in Russia at this time.

[5] Cancer risks were also estimated as part of the overall risk assessment. See VHRAWG (1997) for detailed results for cancer risks.

PM_{10}. Instead, they report individually emissions of several pollutants that would typically be included in the definition of total particulates. For this study, various types of particulate matter emitted by facilities were aggregated into one measure of total particulates (TSP). Table 3 indicates the various materials that are aggregated into TSP for two facilities with high levels of total particulates.[6] [7]

Step 2. Exposure Assessment

For this analysis, the population is defined as the approximately one million individuals who live within Volgograd city boundaries and are at risk of exposure to industrial air emissions. Population densities vary throughout the city, and residential and industrial areas are located in close proximity to one another in several parts of the city.

Based on a city map of population densities, 20 receptor locations were chosen throughout the city. Each receptor location represents 5 percent of the city's population, or 50,000 individuals. Given varying population densities, the land area associated with the receptor points also varies.

Following the selection of 20 receptor points, annual average concentrations (AAC) at the selected receptor points were estimated using the Ecolog air dispersion model, which is the standard air contaminant transport model utilized in Russia for setting emission limits for enterprises' pollution permits.[8] [9] As is typical of Russian industrial sites, the 29 facilities analyzed in this study include several thousand individual stacks and fugitive sources. Given the rather small quantity of particulates emitted by many of the 29 facilities, and to save time and financial resources during the analysis, each facility was modeled as an aggregate stack. The characteristics of the aggregate stack (e.g. height and location within the facility) were defined as a rough average across all sources in the facility.[10] Using this "aggregate-stack" assumption, a base modeling run to estimate concentrations at the 20 receptor points was

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[6] Indirect particulate emissions are not included in this study. For example, SO2 is a gas that can be converted into particulate sulfate many kilometers downwind of the facility that emits it.

[7] Russian emissions regulations do contain something called "suspended matter," defined as dust and aerosols that are not differentiated by composition and for which an individual PDK does not exist. This definition of suspended matter is only a subset of what would be classified as total suspended particulates. In Russian emission inventories, known as 2TP-Air, the aggregate "solids" near the top of the reporting form (line 002) is much closer to the notion of total particulates used in other parts of the worlds.

[8] Additional details of the exposure assessment and air dispersion modeling are discussed in VHRAWG (1996).

[9] The basic reference in Russia for air dispersion modeling is OND 86 (State Hydro meteorological Committee, 1987).

[10] A sample modeling run was used to investigate how best to aggregate the various sources to produce minimum errors as compared to running the model using individual sources at the single facility. A single facility and five receptor points were chosen at random in Volgograd and the air dispersion model was run in four ways: (1) as individual sources utilizing the individual source characteristics for each stack and then summing the impacts across each receptor for all of the stacks; (2) as a single area source using ground level as the stack height; (3) as a single stack based on the mean of the individual stack heights; and (4) as a single stack using the top of the highest stack as the source height. The results from modeling runs #2, #3, & #4 were compared with the full run #1, and it was concluded that run #3 agreed best with the full model #1.

completed assuming an emission rate normalized to 1 g/sec.[11] [12] Some sample results of the normalized emission modeling runs are presented in Column 1 of Table 4 and Table 5 for two facilities.

Ecolog, the Russian air dispersion model used in this study, is designed to model 20-minute air concentrations assuming both "maximum emission rates" and weather conditions that lead to the highest possible concentration estimate.[13] Ecolog is *not* designed for estimating ambient air concentrations for long periods of time under general weather conditions.[14] The short-term maximum emission estimates generated by Ecolog are useful in the Russian context to compare modeled ambient air concentrations with short-term regulatory criteria (i.e. 20-minute ambient air concentration standards). Short-term maximum emission estimates are not useful for estimating chronic air pollution impacts that result from long-term exposures, however, and this presented a major obstacle to carrying out the risk assessment.

To solve this problem, a method was developed for converting Ecolog's 20-minute maximum emission rates into annual average emission rates based on two weighting factors. The first factor, WF1, is defined as the fraction of time the facility operates throughout an entire year, based on the number of shifts per day (1,2, or 3, each representing eight-hour work shifts), and the number of weeks in the year the facility operates (52 weeks maximum). The second factor, WF2, is defined as the ratio of annual average emissions of the facility (based on total tons/year for all pollutants converted to an average g/s) to the maximum 20-minute emission rates.[15] For the 29 facilities in Volgograd, WF1 ranges from 0.14 to 1, while WF 2 ranges from 0.09 to 1.

Annual average concentrations (AAC) from each facility are then given by:

$$AAC = a * p * WF1 * WF2 * d$$

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[11] This normalized emissions rate approach to air dispersion modeling has been used elsewhere for assessing cancer risks from multiple facilities in a region (see, e.g., Industrial Economics and Sullivan Environmental Consulting, 1994).

[12] Concentrations were not modeled for all receptor points for all facilities. Since Volgograd is a large and long city, air concentrations at receptors located great distances from specific emission sources are negligible. Thus, only receptors located mainly in the same part of Volgograd (northern, central, southern) as the source were included in the modeling run.

[13] In general, data on two types of emissions rates are available in Russia: (1) "maximum" emission rates (g/s) for each source, assumed to occur for a maximum 20-minute duration and used in setting annual emission limits (tons per year); and (2) facility-level annual average emission rates which can be calculated based on reported tons of emissions per year. Actual annual average emission rates by stack are not available. The Ecolog dispersion model is designed to use maximum emission rates in its calculations.

[14] There is clearly the need for annual average air quality modeling capabilities in Russia. However, implementing such models will require good access to existing meteorological data, which currently is not available. Until the necessary models and related weather data exist, and the Russian regulatory system evolves to include somewhat different emissions data, practical risk assessments conducted by city and regional officials will need to continue to rely on existing officially-approved air dispersion models, like Ecolog.

[15] Scott Wolff and John Evans developed this weighting factor approach. As an example, assume that facility X operates 2 shifts per day for 8 months of the year, has total emissions of 500 tons/year, and has a maximum 20-minute emission rate for all pollutants of 150 g/sec. The first weighting factor is (2/3)*(8/12) to account for 2 shifts out of 3 per day for 8 out of 12 months. To compute the second weighting factor, note that 500 tons/year equates to 14.4 g/sec assuming the facility operates throughout an entire year. Thus, the second weighting factor is 14.4/150. The product of these two weighting factors, 0.04, must be multiplied by the Ecolog results to derive an annual average emission estimate.

where

AAC	=	average annual concentration estimate for PM_{10} ($\mu g/m^3$)
a	=	normalized 20-minute exposure point concentration for TSP from Ecolog ($\mu g/m^3/g/s$)
p	=	0.6, which is the assumed proportion of PM_{10} in TSP
WF1	=	weighting factor representing fraction of facility operating time
WF2	=	weighting factor representing average/maximum emissions
d	=	maximum emission rate of TSP (g/s).[16]

For example, in Tables 4 and 5, column 2 provides the estimated annual concentration of PM_{10} generated by one facility at each modeled receptor point.

Step 3: Dose-Response Assessment for PM_{10}

The literature on the health effects of air pollution provides clear evidence of mortality from inhalation exposure to particulates, and this information is regularly used in health impact studies (see, for example NRDC, 1996 and Wilson and Spengler, 1996). For particulates, short-term or acute effects are summed over longer time periods to approximate mortality risks from long-term or chronic exposure to PM_{10}.

For the Volgograd study, a PM_{10} mortality coefficient linking ambient concentrations to health risks was developed based on a review of previous studies of acute exposure to ambient air particles. Acute-exposure (daily) epidemiology studies, conducted under wide ranging environmental, climatic, demographic, and geographic conditions, consistently indicate that an approximate 1 percent increase in daily mortality rates occurs for every 10 $\mu g/m^3$ of PM_{10} in ambient air (NRDC, 1996, p. 23). Using this relationship, an individual particulate coefficient describing the additional mortality risk per person per year per 1 $\mu g/m^3$ of PM_{10} (assuming linearity) can be developed as follows.

With about 2,100,000 total deaths in the U.S. annually, there are on average 5,753 deaths per day.[17] A one-percent increase in daily deaths equals 57.53 additional deaths per day. If the entire U.S. population of 250,000,000 individuals were exposed to 10 $\mu g/m^3$ of PM_{10} daily, then the additional deaths per day per person per 10 $\mu g/m^3$ exposure would be estimated by $57.53/250,000,000 = 2.3*10^{-7}$. Multiplying this daily effect by 365 to convert it to an annual effect, the resulting *individual annual particulate mortality coefficient* (PC) is $8.5*10^{-5}$ per 10 $\mu g/m^3$ of PM_{10}, or equivalently PC = $8.5*10^{-6}$ per 1$\mu g/m^3$ of PM_{10}. This is the particulate coefficient used for the base estimate in this study [18] [19]

[16] As will be discussed in the exposure assessment section, particulate matter is included in the risk assessment directly as PM_{10} and not as total suspended particulates (TSP), with the assumption that $PM_{10} = 0.6*TSP$ (Wilson and Spengler, 1996).

[17] The population figures and death rates used in this calculation are slightly lower than population figures and death rates reported for 1995 in the U.S. in World Resources Institute (1996, p. 191).

[18] This PC estimate is higher than the $6.1*10^{-6}$ reported in Wilson and Spengler (1996, p. 202). On the other hand, this PC is substantially lower than would be obtained using current Russian mortality figures. With a current Russian population reported at 148 million and an average death rate in Russia of 12.4 per 1000 per year in 1995 (World Resources Institute, 1996, p. 191, p. 195), while maintaining the assumption of 1% increase in daily mortality for a 10$\mu g/m^3$ increase in PM_{10}, the PC would be estimated at $10.9*10^{-6}$. In Volgograd, the 1995 mortality rate was somewhat higher than the national average at about 13.4 per 1000, which would imply an even higher particulate coefficient. Thus, a PC of $8.5*10^{-6}$ per 1$\mu g/m^3$ of PM_{10} is a reasonable (i.e. not too high) starting point for this study.

Since each receptor point in the Volgograd study represents a population of 50,000 people, a *Volgograd receptor point PM_{10} coefficient* (VRPC) can be estimated as $8.5*10^{-6}*50,000 = 0.42$ additional deaths per year per 1 $\mu g/m^3$ PM_{10}. For example, if an estimate of annual average PM_{10} concentrations in Volgograd were 50 $\mu g/m^3$ PM_{10} at each receptor point, (which is the U.S. annual average PM_{10} ambient air concentration standard), a VRPC of 0.42 would imply an average of 0.42*50 = 21 additional deaths per year at each receptor point, or a total of 21*20 = 420 additional deaths per year city-wide.

Step 4: Risk Characterization for the Base Case

To estimate the additional annual average mortality risks from PM_{10} faced by Volgograd's population, the health criterion (VRPC)must be multiplied by the annual average exposure point concentrations (AACs). These results are summarized in Table 6 for all 29 facilities for the 20 receptor points. As indicated in the lower right corner of Table 6, this study estimates approximately 2,700 additional deaths per year in Volgograd from particulate emissions from the 29 facilities. This overall annual mortality impact of PM_{10} on the Volgograd population is 200 times greater than the estimated impact of potential human carcinogens from stationary sources, even assuming that all cancers cause death. (VHRAWG,1997).

The estimate above reflects the total mortality risk generated by moving from 0 $\mu g/m^3$ (no PM_{10} in the air) to the estimated ACC at each receptor point. Another way to assess the mortality risk from particulates in Volgograd is to evaluate the level of "excess" risk over some specified ambient air concentration level. For example, if an annual average PM_{10} concentration of 50 $\mu g/m^3$ is considered acceptable (as it is in the U.S.), this analysis would estimate that the first 21 deaths per year at each receptor point would fall within the "acceptable" range. If an acceptable annual average PM_{10} concentration is 100 $\mu g/m^3$, this analysis would estimate that the first 42 deaths per year at each receptor point is acceptable.

As can be seen in the last row of Table 6, all estimated receptor point risks are greater than 21 except for receptor 6, which is 20.46, while estimated receptor point risks are greater than 42 at fourteen receptor points. Thus, if an annual concentration of 50 $\mu g/m^3$ PM_{10} were considered a reasonable policy goal in Volgograd, then this analysis would predict 2,247 "excess" deaths per year in Volgograd. Similarly, if an annual concentration of 100 $\mu g/m^3$ PM_{10} were taken as a reasonable policy goal in Volgograd, this analysis would predict 1,894 "excess" deaths per year from the concentrations estimated in this study. In a city of 1 million, these levels of "excess" risk are substantial.

Table 6 also reveals that PM_{10} mortality risks are highly concentrated, with receptor points 12-20 in the northern part of the city bearing about 80 percent of the city=s total risk. Similarly, a population of just 250,000 people at receptor points 16-20 bears about 58 percent of the city's total risk, 64 percent of the "excess" risk above 50 $\mu g/m^3$ PM_{10}, and 73 percent of the "excess" risk above 100 $\mu g/m^3$ PM_{10}.

Mortality risks are concentrated in the northern part of city because the two largest emitters of particulates, facilities 314 and 315, are located in the north. As Table 6 indicates, these two facilities (314 and 315) contribute 85 percent of the mortality impacts attributable to PM_{10} exposure. Tables 4 and 5 provide additional details on estimated concentrations for these two facilities.

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[19] Given the assumption in this study that $PM_{10} = 0.6*TSP$, a TSP annual mortality coefficient can be estimated directly as $0.6*8.5*10^{-6} = 5.1*10^{-6}$ per 1 $\mu g/m^3$ TSP, which is somewhat higher than the estimate of $3.4*10^{-6}$ per 1 $\mu g/m^3$ TSP discussed in Evans, Tosteson, and Kinney (1984). On the other hand, Industrial Economics and Sullivan Environmental Consulting (1994) use a mortality coefficient of $6.1*10^{-6}$ per 1 $\mu g/m^3$ TSP as an average coefficient.

The importance of evaluating risks, rather than just total emissions, can easily be seen by comparing the detailed results for Facility 214 with those for Facility 315. The emissions inventories in Table 2 show that Facility 214 emitted 2,480 tons of particulates in 1995, while Facility 315 emitted 2,169 tons, about 300 tons less than Facility 214. From a glance at total particulate emissions, it is tempting to conclude that these two facilities' emissions are of roughly equal importance for environmental policy attention. The risk assessment concludes, however, that mortality risks from particulates are 1,200 percent higher for Facility 315 (in the north) than for Facility 214 (in the south), mainly because of plant location in relation to residential centers.

2.1 Uncertainties in the Volgograd Risk Assessment

Health risk assessment is a multi-step, multidisciplinary process that estimates the potential human health risks that result from exposure to chemicals in the environment. This process requires both scientific analysis and professional judgment to capture the complexity of the variables and input parameters needed to quantify health risk. This complexity gives rise to considerable uncertainty, and the numerical risk estimates derived from any risk assessment should be viewed in light of this uncertainty.

There are several sources of uncertainty in the Volgograd study. Some of these sources are inherent in the risk assessment process, while others are specific to the data and methodology used in the Volgograd study. For example, it is not known whether the general composition of particulates and the population mix in Volgograd differ substantially from those found in areas where previous PM_{10} epidemiology studies have been conducted. Regarding exposure and demographic factors, only ambient exposures and the inhalation pathway have been considered. Human mobility patterns and indoor/outdoor differences in exposures may bias the results differentially depending on age, residence location, building characteristics, and occupational and personal characteristics.

Regarding basic data, emissions estimates used in the analysis are highly uncertain. Only maximum emissions based on engineering estimates are presented, and no distribution of emissions is available. There is no way to verify the accuracy or completeness of the 1995 emissions inventory. Since enterprises pay penalty charges for emitting more than their annual emission limits, there may be some incentive to under-report emission levels.[20] If Russian industrial output, and thus air emissions, begin to grow again in coming years, as is expected, risk estimates based on 1995 emissions may understate future risks if no additional measures are taken to abate pollution. It is important to bear in mind, however, that even though the data used in this study cannot be “verified,” they are the core emissions data that policy makers actually use for implementing and evaluating air pollution policy in Volgograd.

There is clearly uncertainty in the approach used to model ambient concentrations. Ecolog, the air dispersion model used in this study, was designed primarily to establish emission limits within the existing Russian regulatory framework. Information on the underlying accuracy and precision of this model, as well as the accuracy of the approach used to convert 20-minute Ecolog results into annual average concentrations, is not available. Even the best air quality dispersion models are only accurate within a factor of two or three (U.S. EPA 1992a; 1986), however, making the modeling of ambient concentrations an important source of uncertainty in most risk assessments.

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[20] Russian air pollution policy requires enterprises to pay a (low) base charge for emissions up to their annual limits and to pay a (higher) penalty charge for emissions exceeding their annual limits. See Bluffstone and Larson (1997) for more on the experience with pollution permits and charges in the transition economies of Europe and the former Soviet Union.

Some of the assumptions used to implement the Volgograd risk assessment are likely to overstate risks and some are likely to understate risks. Although the overall effect is unclear, it is possible to evaluate some of the uncertainties through a simple sensitivity analysis of the base-case risk estimates. Table 7 reports the results of such an analysis. Because even the base-case risk estimates for the three cutoff points (0, 50, and 100 $\mu g/m^3$ PM_{10}) are all serious enough to warrant policymakers' attention, the main concern in evaluating uncertainties is that these risks have been *overestimated*. The sensitivity analysis focuses on changes in two key parameters: the choice of the particulate coefficient (PC) and the percentage of PM_{10} in TSP.

Column 2 in Table 7 shows how estimated risks change if a lower PC--$6.1*10^{-6}$ per 1 $\mu g/m^3$, which is reported in Wilson and Spengler (1996)--is used in the analysis. On the other hand, if it is assumed that daily mortality increases by only 0.5 percent per 10 $\mu g/m^3$, instead of by 1 percent per 10 $\mu g/m^3$, but actual Volgograd mortality rates of 13.4 per 1000 in 1995 are used, then the PC would be estimated at about $6.4*10^{-6}$ per 1 $\mu g/m^3$. We thus conclude that $6.1*10^{-6}$ per 1 $\mu g/m^3$ provides a good lower-range estimate of a PM_{10} annual mortality coefficient for Volgograd. With this lower-range assumption, which is a 28 percent reduction in the PC from the baseline PC of $8.5*10^{B6}$, annual total mortality risks would fall to 1,920, while risks above 50 $\mu g/m^3$ would fall to 1,506 and risks above 100 $\mu g/m^3$ would fall to 1,224.

Column 3 in Table 7 shows how estimated risks change if it is assumed that PM_{10} constitutes 30 percent of TSP, instead of the 60 percent used in the base case, while maintaining the original PC. Industrial Economics and Sullivan Environmental Consulting (1994), based on U.S. EPA documents, reports the ratio of fine particles in TSP in the range of 28-50 percent in U.S. cities with industrial sources and uses 42 percent as an average. We thus conclude that 30 percent is a reasonable lower-range estimate of the proportion of PM_{10} in TSP. With this lower-range assumption, which is a 50 percent reduction from the percentage of PM_{10} used in the baseline case, annual total mortality risks in the city fall to 1,333, while risks above 50 $\mu g/m^3$ fall to 947 and risks above 100 $\mu g/m^3$ fall to 709.[21]

Finally, Column 4 in Table 7 assumes both a PC reduction of 28 percent (as in Column 2) *and* a reduction of the share of PM_{10} in TSP to 30 percent (as in Column 3). In this case, called the "low-end" case, total city risk is still 960 additional deaths per year, and "excess" risks above 50 $\mu g/m^3$ are still about 612 additional deaths per year. All of these excess risks are borne by a population of 500,000 people in the northern part of the city (receptor points 11-20). Thus, even with three new conservative assumptions (a 28 percent lower PC, 50 percent less PM_{10} in TSP, and a 50 $\mu g/m^3$ cutoff), people in the northern part of the city still face additional annual mortality risks of about 1,200 per million (1.2 per 1000).

2.2 Placing the Volgograd Results in Context

To our knowledge, no other quantitative health risk assessments of PM_{10} have yet been published for Russia. Studies from other countries may provide some context for interpreting the Volgograd results, however. For example, a recent study by the Natural Resource Defense Council (NRDC, 1996) estimated mortality health risks from PM_{10} for 48 locations in the U.S. Average annual mean concentrations of PM_{10} based on directly measured air quality ranged from about 30 to 60 $\mu g/m^3$ (at the time U.S. annual standard was 50 micrograms per cubic meter) (NRDC, 1996, p. 78), and total annual mortality risks were estimated in the range of 450 deaths per million to 1,230 deaths per million (NRDC, 1996, p. 79). At some local "hot spots," average annual concentrations were as high as 80-90 $\mu g/m^3$.

[21] Column 3 can also be interpreted as the case where PM_{10} = 0.6*TSP, but the AAC is reduced 50% to reflect uncertainty in the air dispersion modeling results.

In Volgograd, the baseline mortality estimate is 2,667 deaths per million. Average annual PM_{10} levels are estimated at greater than 100 μg/m^3 at ten receptor points for facility 314 and at three receptor points for facility 315. Based on emissions from these two facilities alone, average annual estimated concentrations in Volgograd are 100 percent higher than the U.S. PM_{10} annual ambient air quality standard, and at some locations are considerably higher than that (the concentration at receptor 19 for facility 314 is more than 1,000 μg/m^3). The concentration of risk in the northern part of the city is further cause for concern. In terms of public understanding and acceptance of risks, there is some indication from other parts of the world that geographically concentrated risks are perceived as “worse” by affected populations than are more widely dispersed risks (Percival, et al. 1992).

The Volgograd results can also be compared to actual mortality figures for Russia and for Volgograd. The Russian gross mortality rate for 1995 is reported to have been 12.4 per 1000 (World Resources Institute, 1996), and the Volgograd city mortality rate for 1995 is reported to have been 13.4 per 1000, up about 17 percent from 11.4 per 1000 in 1991 (Filatov, 1997). In Volgograd, about 60 percent of gross mortality (about 8 deaths per year per 1000 or 8,000 deaths per year total) is due to cardiovascular disease, one of the main causes of death associated with particulates. Table 7 contains risk estimates of 2,667 deaths per year in the city as a high and 960 deaths per year as a low estimate. The risk assessment thus suggests that 12-28 percent of cardiovascular deaths each year in Volgograd, and 7-20 percent of total Volgograd mortality, may be linked to particulate emissions.

In comparison, NRDC (1996, Table 10, p. 73) estimates that about 9 percent of adult cardiopulmonary mortality is attributable to particulates in over 200 metropolitan areas in the United States. Given the higher gross mortality rates in Russia as compared to the United States and the higher ambient concentrations of particulates in Volgograd (even for the low-end estimate from Table 7), the low-end risk estimate is solidly in the range of reasonability. The base-case estimate is consistent with a “health conservative” approach, in which it is unlikely that actual mortality risks are greater than estimated.

3. COST-EFFECTIVENESS AND BENEFIT-COST ANALYSES

The previous section concludes that PM_{10} mortality risks in Volgograd are substantial, with a reasonable estimate of total deaths due to PM_{10} in the range of 960-2,667 per year and deaths due to PM_{10} concentrations above 50 μg/m3 in the range of 612-2,247 per year. Because concentrations will never be reduced to zero, the excess deaths above 50 μg/m3 can be considered the policy-relevant mortality effect for pollution management purposes. The majority of this mortality effect occurs in the northern part of the city, and facilities 314 (an aluminum factory) and 315 (a steel products factory) are the major contributors to the effect. Thus, for practical management of particulate mortality risks in Volgograd, it makes sense for the city□s policymakers to focus on facilities 314 and 315 and on the northern part of the city.

This section presents the results of a simple cost-effectiveness analysis of particulate emission reduction options for facilities 314 and 315. A cost-effectiveness analysis has five steps: identification of baseline risk (completed in previous section); identification of various investment and policy options to reduce risks; determination of the cost of each option; determination of the change in risk from the baseline from each option; and prioritization of the options from lowest to highest cost per change in risk.
As part of the U.S. Government-funded Russian Air Management Project (RAMP), a set of particulate emission reduction options were recently developed for facilities 314 and 315 (SAIC, 1995). RAMP identified several low- and high-cost particulate control measures for the aluminum factory (facility 314) and the steel products factory (facility 315), calculated capital and annual operating costs for each measure, and determined the annual particulate emission reductions that would be achieved by implementing each measure. The detailed cost and emissions-reduction information generated by RAMP

(SAIC, 1995), combined with the risk assessment results described above, provide the basis for this cost-effectiveness analysis.

3.1. A Simple Cost-Effectiveness Criterion

The place to begin is a definition of the costs of risk reduction options, which for this study are all investment projects. As a result, the present discounted value of costs is used. For example, based on RAMP documents, it is assumed for this analysis that all capital costs Ki are paid at the beginning of periods i=1,..m, operating costs Ci are paid at the end of each period i=m,..n, and the interest rate is constant and equal to r in each time period. In this case, the present value of costs (PVC) of a project can be calculated as:

$$PVC = \sum_{i=1}^{m} K_i \left(\frac{1}{1+r} \right)^{i-1} + \sum_{i=m}^{n} C_i \left(\frac{1}{1+r} \right)^{i} \qquad (1)$$

In equation (1), time periods from 1 to m represent construction time and time periods from m to n represent the project operating life of the project. The annual interest rate r reflects the opportunity cost of allocating money to the project instead of to the next best alternative. If the decision maker is an enterprise, then the relevant interest rate is the rate the enterprise could earn by putting its money in the bank or in some other relatively riskless asset. If the decision maker is "society," then the social rate of return should be used. The choice of a private or social rate in equation (2) will depend on the reasons for doing the cost-effectiveness analysis.

Assuming each project yields a change in risk dRi during the operating period (time periods m to n), a general "cost-effectiveness" criterion P could be defined as:

$$P = \frac{PVC}{\sum_{i=m}^{n} dR_i \left(\frac{1}{1+d} \right)^{i}} \qquad (2)$$

where d is a "discount factor" used to translate changes in risk across different periods into a common denominator in period 1. Since dRi represents changes in mortality risks, the cost-effectiveness criterion P can be interpreted simply as the cost per life saved (or cost per death avoided).

While choosing an interest rate to compare costs across time periods is not too difficult, choosing a discount rate to compare risk changes across time periods is challenging. While it is not conceptually clear that risks themselves should be discounted, as opposed to discounting the monetary value of the risks as would be done in a cost-benefit analysis, empirical research from the U.S. suggests that people do make such discounting considerations in their opinions of the benefits of risk reduction (e.g., see Cropper, Aydede, and Portney, 1992). Moore and Viscusi (1990) find that estimated rates of discount for health risks are not substantially different from financial rates of time preference. In terms of the above equation (2), this would imply that d is roughly equal to r.[22]

[22] It is commonly the case that an enterprise will want or need to use credit markets to assist in financing a project (i.e. take out a loan). In this case, the company should include financing costs (as well as an evaluation of cash flow) in its evaluation of project costs.

3.2 Emission Reduction Options for the Volgograd Enterprises

The Volgograd aluminum plant (facility 314) emitted about 10,000 tons of particulates in 1995 (VHRAWG, 1997). Various measures to reduce particulates at the aluminum factory are presented in SAIC (1995, p. 49-50) and Volgograd Air Protection Program (1995). From these measures, the following three emission and risk reduction options were selected for the cost-effectiveness analysis for the aluminum factory:

Option A1. Reduction of emissions by reducing depressurization of pots (measure 1 in SAIC) through the material and technical provision of dust/gas control service, although with the addition of operators and workers for dust/gas control and their training (measure 2 in SAIC).

Option A2. Replacement of scrubbers by more effective ones (measure 3 in SAIC).

Option A3. Unspecified technical changes in electrolytic cells (from Volgograd Air Protection Program, 1995).

The "Red October" steel products factory (315) emitted about 2,200 tons of particulates in 1995. Low-cost measures to reduce particulate emissions at this facility are also identified in SAIC (1995, p. 74-76). Based on these measures, the following three emission and risk reduction options are included in the cost-effectiveness analysis for the steel factory:

Option R1: Maintenance of furnaces in a staggered production cycle and the regulation of primary and secondary hood suction from furnaces.

Option R2: Arrangement of wet dust/gas control equipment on exhaust pipes of cupolas.

Option R3: Reduction of dust on roads within the facility's territory.[23]

3.3 Cost-Effectiveness Results

Tables 8 and 9, line items 1-4, provide a summary of all cost and emission reduction information needed for the cost-effectiveness analysis. As is commonly the case in pollution control project documentation in Russia, the emission reduction associated with each project is assumed to be constant throughout the project life. In Tables 8 and 9, line item 3 reports the estimated annual change in risk associated with the annual emission reductions, with Table 8 results based on the base case risk assumptions in Table 7 (column 1) and Table 9 results based on the low-end risk assumptions in Table 7 (column 4).[24] [25] In Tables 8 and 9, line item 4 is the present discounted value of project costs, PVC, calculated using

[23] An estimate of particulate emission reductions from this option is not reported directly in SAIC (1995) but is indicated only as substantial. It also cannot be determined from the facility's emissions inventory if such area source emissions are included in the emissions inventory reported to the local environmental committee. For now, it is assumed in Table 7 that option R3 reduces particulate emissions by 1,000 tons per year. The sensitivity of the cost-effectiveness results to this assumption will be discussed after the presentation of the basic analysis.

[24] In this example, each project is independent from the other projects, and they could be implemented in any sequence. In some situations, projects are likely to be linked technically, so that one project must be implemented first before another project could be implemented. The implications of such linked projects would have to be included in the cost-effectiveness analysis.

[25] Due to linearity in the air dispersion model and risk calculations, a 1 percent reduction in annual emissions is equivalent to a 1 percent annual reduction in estimated risk.

equation (1). Project life is not included in the project documentation, and a 10-year project working life is assumed for the analysis here. The annual real interest rate is assumed to be 10 percent, which is probably a realistic estimate of the risk-free opportunity cost of money for enterprises in Russia.
For this analysis, since annual change in risk dRi in equation (2) is constant and equal to dR in Tables 8 and 9 for all periods, equation (2) simplifies to:

$$P = \frac{PVC}{dR \;\; * D} \quad where \;\; D = \sum_{i=m}^{n} \left(\frac{1}{1+d} \right)^{i} \tag{3}$$

With dR constant across periods for each project, the choice of a discount factor d for risks will influence the overall level of P as defined in (3). For example, if d = 0 so that risks are not discounted, then D = 10. If on the other hand d = 0.10, then D = 6.14.[26]

The cost-effectiveness criterion P from equation (3) is provided for the six projects in Tables 8 and 9, line items 5 and 6. Line item 5 assumes no discounting of future risk reduction, while line item 6 uses a 10 percent discount rate. These varying assumptions affect the magnitude of the cost calculations (i.e. cost per life saved), but the ranking of the various projects from least to highest cost is not affected.[27] As reported in Tables 8 and 9, the projects are ranked as R2, R3, A1, R1, A3 from lowest to highest cost. For any combination of assumptions on risk (i.e. base case or low-end case) and discounting of lives (i.e. 0 percent or 10 percent), options R2, R3, A1, and R1 are clearly low-cost options for reducing mortality risks, while options A3 and A2 begin to look substantially more expensive under some combinations of assumptions. For example, project costs in terms of P are estimated in the range of 0.430-1.95 million rubles (about $90-$400) for R2 and 1.16-5.22 million rubles for A1 ($200 - $1,000) depending on whether the base case level of risk with no risk discounting or the low-end level of risk with 10 percent discounting is used for defining the cost per life saved.[28]

3.4 Benefit-Cost Analysis

Finally, while the previous section focused on project costs defined as cost per life saved P, the concept of the "Value of a Statistical Life" (VOSL) can be used to value the monetary benefits of reducing mortality risks—the "benefit-per-life saved" (see, e.g., Viscusi, 1993; Freeman, 1979). While empirical estimates from Russia are not yet available, a simple benefits transfer approach using U.S. estimates can be applied as a first approximation. For example, Viscusi (1993) reports VOSL estimates from twenty-four empirical studies from the United States. The *lowest* estimate was $600,000. To transfer this U.S. figure to the Russian context, consider that Russian GNP per capita is about 18 percent, or just under one fifth, of U.S. GNP per capita on a purchasing power parity basis (World Development Report, 1996). If this one-fifth fraction is used to weight the lower bound U.S. VOSL, the result would be a Russian VOSL

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[26] Notice that for six projects from these two enterprises, a cost-effectiveness criterion defined in terms of emissions reductions rather than risks, P* = PVC/(10*dR) would yield the same rankings as above. This result was essentially predetermined because facilities 314 and 315 are relatively close to each other. As a result, risk impact of one ton of $PM_{10}$from facility 314 is close to the risk impact of one ton of PM_{10} from facility 315. This is an important point. Since most Russian cities cannot yet afford to carry out risk assessments, a cost-effectiveness analysis based on emissions reductions (rather than risk reductions) is still useful (and much better than doing nothing).

[27] The cost-effectiveness criterion P increases by about 62 percent from the base case using the 10 percent rate to discount future risk reductions.

[28] Since the emission reduction estimate for R3 (reduction of road dust) is highly uncertain, and reduction of road dust may imply fairly large particles, it is likely that the emission reduction estimate of 1,000 tons per year for R3 is an overestimate, perhaps moving project R1 up one place in the ranking.

estimate of about $120,000.[29] To ensure that benefits of mortality risk reduction in Russia are not overstated, and acknowledging the basic uncertainty in the precision of the risk estimates, a weight of 1/20 is used to convert between U.S. and Russian contexts, producing a Russian VOSL estimate of about $30,000 (about 171 million rubles). Using these assumptions, it is unlikely that a "true" VOSL in Russia at this time is less than 171 million rubles. Even with these conservative estimates on the VOSL, using the low-end risk case in Table 9, and assuming a 10 percent rate for discounting mortality risks, the cost per life saved criterion P is substantially lower than the VOSL for all projects except A2.

The conservative VOSL estimate and the project information in Table 8 and 9 can be used to conduct a standard cost-benefit calculation for all six projects (present discounted value of project benefits minus project costs assuming a 10 percent discount rate). For all assumptions, all projects except A2 pass a cost-benefit test, with discounted benefits being substantially larger than discounted costs. Project A2 passes a cost-benefit test using the base-case risk assumptions from Table 8, but it does not pass the test using the low-end risk assumptions. If the five projects (A1, A3, R1, R2, R3) were implemented simultaneously in Volgograd, and assuming the low-end risk case, about 227 deaths per year would be avoided in the city for a total net benefit of about $39.5 million (i.e. discounted net present value of all five projects). This reduction represents about 25 percent of total risks in the low-risk case. The net benefits would be substantially higher if the base-case risk assessment results were used.

4. SUMMARY AND FINAL COMMENTS

This study provides a practical example for environmental and health officials in Russia of how the logic of quantitative health risk assessment, cost-effectiveness analysis, and benefit-cost analysis can be used in their cities for managing health risks. Since this analysis uses data and air dispersion modeling capabilities that are commonly available throughout Russia, the approach taken here can be widely and relatively easily applied in many other cities in Russia and the former Soviet Union. It is likely, of course, that the methods used in this study will be refined over time as the quality of data and access to data improve in Russia and elsewhere (especially emissions data).

The main results of this analysis can be summarized as follows. First, it is clearly possible to combine existing data and air dispersion modeling capabilities in Russia with modern methods of quantitative health risk assessment to evaluate health risks from particulates. The results of risk assessments can be used immediately by city and regional environmental officials, within the existing Russian regulatory framework for setting pollution limits for enterprises, granting pollution charge credits, and allocating resources from environmental investment funds. There remain substantial uncertainties about the resulting risk estimates, however.

Second, annual particulate related mortality risks in the city of Volgograd are estimated in the range of 960 to 2,667 additional deaths per year in the city of one million, with the great majority of risks generated by two major polluters in the northern part of the city. "Policy-relevant risks," defined as risks created above 50 micrograms annual average air concentrations of particulate matter of less than 10 microns diameter size (PM_{10}), are in the range of 612 to 2,247 additional deaths per year in the city.

Third, the marginal cost of mortality risk reduction for six projects is in the range of 0.43 to 1.95 million rubles (about $74 to $336 per death avoided at the current exchange rate of 5,800 rubles per U.S. dollar) for the lowest cost option to a range of 39 to 178 million rubles (about $6,724 to $30,689) for the highest

[29] Using essentially the same logic to determine a reasonable Russian VOSL, except starting with a range of accepted average values, Markandya (1997) suggests a range of $500,000 - $1.7 million in 1996 prices.

cost option. These ranges depend on various assumptions in the risk estimate and the rate used to discount future risk reductions.

Fourth, with a lower-bound estimate of the value of a statistical life in Russia of 171 million rubles (about $30,000), all projects except one pass a cost-benefit test using the low-end risk estimates. Morbidity effects of particulates are not included in the analysis. If included, they would increase the benefits of particulate reductions.

While particulate-related mortality risks are estimated to be very serious in Volgograd, and are likely to be important in many Russian industrial cities, the Russian regulatory framework is not yet well designed for managing particulate health risks. Perhaps most important, Russia does not regulate “particulates” as a single substance and does not have an annual ambient standard for TSP or PM_{10}. As indicated in Table 3, the category “particulates” includes many different substances. Russian practice has been (and remains) to set emission limits for each individual substance based on the air dispersion modeling process outlined in Section 2 for maximum emissions and worst weather conditions. Short time-period concentrations (20 minutes) are then compared to 20-minute ambient concentration standards (called *PDK* in Russian) for each individual substance. There is no PDK (ambient concentration standard) for particulates as a group, however.

To address the serious health risks its population faces from particulates, Russia should (a) develop a regulatory framework for managing particulates as an aggregate category, such as TSP and/or PM_{10}; and (b) include considerations of chronic health risks in its regulatory framework through, for example, longer time period PDKs (such as 24-hour and annual average PDKs). It is not evident why there has been so little emphasis in recent years on developing a better regulatory framework for managing particulates in Russia. In part, there remains substantial confusion about the overall issue of particulate pollution in Russia, and it will be necessary to provide additional technical training to city, regional, and federal environmental and health policy officials to allow them to implement and enforce a new approach to particulates.

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Table 1. List of Volgograd Facilities Included in the Study

Facility Code	Industrial Sector
CENTRAL ZONE	
105	Construction materials
117	Construction materials
122	Machine building
136	Construction materials
137	Construction materials
144	Agricultural machinery
168	Furniture, armoured kiosks
169	Machine building
174	Construction materials
180	Food industry
SOUTHERN ZONE	
211	Energy sector
213	Wood processing
214	Chemical industry
218	Locomotive shop
220	Oil-organic synthesis
237	Ferrous metallurgy
239	District heating
241	District heating
249	Shipbuilding Plant
250	Ferrous metallurgy
252	Oil-organic synthesis
253	Chemical industry
254	Ceramics
NORTHERN ZONE	
303	Agricultural machinery
310	Agricultural machinery
312	Construction materials
314	Non-ferrous metallurgy
315	Ferrous metallurgy
341	Military

Table 2. Summary Emission Inventory for Selected Facilities, 1995

Facility Code	Total Particulates (TSP), t/yr	Facility Code	Total Particulates (TSP), t/yr
105	1523	237	6
117	10	239	42
122	20	241	22
136	280	249	43
137	272	250	199
144	8	252	45
168	4	253	264
169	150	254	334
174	13	303	6
180	less than 1 ton	310	228
211	809	312	less than 1 ton
213	100	314	10003
214	2480	315	2169
218	12	341	102
220	22		

Table 3. Definition of Total Suspended Particles (TSP)

TSP Composition for Facility 314

Aluminium Oxide
Barium Chloride
Vannadium Pentoxide
Iron Oxide
Manganese and its compounds
Sodium hydroxide
Soot
Fluoride non-organic compounds
Naphthalene
Antracene
Phenantrene
Pyrene
Stearine
Suspended solids
Boiler oil soot in terms of Vanadium
Non-organic dust with Silicon oxide content 20-70%
Non-organic dust with Silicon oxide content more than 70%
Asbestos dust
Wood dust

TSP Composition for Facility 315

Iron Oxide
Potassium Oxide
Manganese and its compounds
Sodium hydroxide
Chromium Hexavalent
Colophony
Dust non-differentiated
Non-organic dust with Silicon oxide content 20-70%
Non-organic dust with Silicon oxide content more than 70%
Wood dust

Table 4. Annual PM_{10} Concentrations for Facility 315

Receptor Points	Normalized Modeling Results (µg/ m^3)/(g/s)	Annual PM_{10} Concentration (µg/ m^3)
1		
2		
3		
4		
5		
6		
7	0.2	12.60
8	0.3	18.89
9	0.5	31.49
10	0.7	44.09
11	0.9	56.68
12	1.7	107.06
13	6.7	421.96
14	0.8	50.38
15	1.4	88.17
16	1.7	107.06
17	1.4	88.17
18	0.7	44.09
19	0.8	50.38
20	0.2	12.60
Nearest point for maximum impact	4.6	289.70

For this facility, WF1 = 1, WF2 = 0.71, and the maximum emission rate is 147.88 g/s.

Table 5. Annual PM_{10} Concentrations for Facility 314

Receptor Points	Normalized Modeling Results (µg/ m^3)/(g/s)	Annual PM_{10} Concentration (µg/ m^3)
1		
2		
3		
4		
5		
6		
7	0.2	58.9880
8	0.2	58.99
9	0.2	58.99
10	0.2	58.99
11	0.4	117.98
12	0.4	117.98
13	0.4	117.98
14	0.5	147.47
15	0.6	176.96
16	1.4	412.92
17	1.3	383.42
18	3.1	914.32
19	3.5	1032.29
20	2.0	589.88
Nearest point for maximum impact	8.8	2595.48

For this facility, WF1 = 1, WF2 = 0.76, and the maximum emission rate is 646.79 g/s.

Table 6. Estimated Annual Additional Mortality due to PM_{10} Emissions in Volgograd, by Facility and by Receptor Points

	Receptors 1-20. Each receptor represents 50,000 individuals.																				
Facility #	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20	Annual Mortality
105							0.93	1.70	1.55	1.08	12.07	3.72	1.55	18.33	4.80	1.39	0.77	0.16			48.04
117							0.04	0.05	0.05	0.05	0.24	0.19	0.10	1.34	0.48	0.14	0.06	0.06	0.09		2.90
122							0.40	0.72	0.56	0.40	1.71	0.72	0.36	1.59	0.68	0.32	0.24	0.04	0.04		7.76
136							0.71	0.95	1.07	1.07	4.39	3.56	2.14	14.96	8.79	3.80	1.66	1.42	2.61	0.59	47.73
137						0.28	0.51	0.68	0.68	0.57	2.72	1.30	0.74	9.13	1.93	0.79	0.57	0.51	0.68		21.08
144							0.09	0.11	0.19	0.31	0.35	1.28	1.41	0.21	0.38	0.15	0.12	0.09	0.10		4.79
168				0.04	0.08	0.22	0.36	0.11	0.12	0.08	0.04	0.04	0.03								1.12
169			0.24	0.48	1.68	4.86	3.36	1.32	1.26	0.90	0.42	0.36	0.30	0.30	0.24						15.73
174			0.18	0.27	0.94	2.54	2.41	0.89	0.89	0.63	0.27	0.27	0.22								9.51
180			0.01	0.28	0.07	0.16	0.35	0.13	0.13	0.09	0.04	0.04	0.03	0.03	0.02						1.35
211	1.61	1.61	4.04	8.07	4.04	2.42	0.81	0.81	0.81												24.22
213	0.32	0.40	0.64	2.95	12.60	3.91	0.80														21.61
214	2.15	3.58	7.87	12.17	5.01	2.86	1.43	1.43	1.43	1.43											39.37
218	0.50	0.85	5.26	0.46	0.24	0.18	0.12														7.62
220																					0.00
237	3.58	1.85	0.30	0.12	0.09	0.09	0.06														6.08
239	0.15	0.15																			0.31
241	0.14	0.14	0.05	0.05	0.05																0.42
249	0.71	2.79	1.27	0.21	0.13	0.09															5.21
250	0.29	0.48	1.87	0.80	0.32	0.21															3.97
252	25.21	8.62	4.64	1.99	1.66	1.33															43.45
253	4.78	2.94	1.47	0.74	0.37	0.37															10.66
254	52.21	10.67	2.29	1.14	1.14	0.76															68.21
303											0.00	0.00	0.00	0.00	0.01	0.01	0.01	0.03	0.02	0.10	0.19
310						0.16	0.24	0.24	0.32	0.40	0.48	0.64	0.97	0.56	0.89	1.77	11.27	9.66	3.78	1.13	32.53
312								0.00	0.00	0.00	0.01	0.01	0.01	0.01	0.01	0.02	0.02	0.04	0.06	0.03	0.22
314							24.58	24.58	24.58	24.58	49.16	49.16	49.16	61.45	73.74	172.05	159.76	380.97	430.12	245.78	1769.64
315							5.25	7.87	13.12	18.37	23.62	44.61	175.82	20.99	36.74	44.61	36.74	18.37	20.99	5.25	472.34
341							0.04	0.04	0.04	0.04	0.04	0.06	0.06	0.04	0.06	0.08	0.10	0.10	0.06	0.08	0.81
Annual mort.	91.66	34.09	30.12	29.76	28.42	20.46	42.48	41.64	46.79	50.00	95.55	105.95	232.88	128.94	128.74	225.14	211.32	411.44	458.55	252.96	2666.88

Table 7. Sensitivity of Results to Changes in Basic Assumptions

	AThe Base Case≅ (PM_{10} = 0.6TSP) (PC = $8.5*10^{-6}$)	28% Lower PC (PM_{10} = 0.6TSP) (PC = $6.1*10^{-6}$)	50% Lower PM_{10} (PM_{10} = 0.3TSP) (PC = $8.5*10^{-6}$)	The ALow-End Case≅ (PM_{10} = 0.3TSP) (PC = $6.1*10^{-6}$)
Total Risk from 0 $\mu g/m^3$ annual average concentration	2667	1920	1333	960
Total Risk above 50 $\mu g/m^3$ annual average concentration	2247	1506	947	612
Total Risk above 100 $\mu g/m^3$ annual average concentration	1894	1224	709	401

Table 8. Evaluation of Emission Reduction Options for Base Case Risk Assessment Assumptions (Column 1 of Table 7)

	a1	a2	a3 **	r1	r2	r3
1. Capital Expenditures (K)	0.00	9,314.00	3,000.00	95.00	56.10	1,605.80
2. Annual O&M Costs (C)	647.00	414.00	0.00	10.10	22.50	89.00
3. Annual Risk Reduction (dR)	344.00	30.00	18.00	5.00	45.00	220.00
4. Present Value of Costs (pvc)	3,975.53	11,857.85	3,000.00	157.06	194.35	2,152.67
5. P= Cost Per Unit Risk Reduction (d = 0)	1.16	39.53	16.67	3.14	0.43	0.98
6. P= Cost Per Unit Risk Reduction (d = 10%)	1.88	64.33	29.84	5.11	0.70	1.59
7. Value of a Statistical Life (VOSL)	171.00	171.00	171.00	171.00	171.00	171.00
8. Net Present Value of Project (NPV)	357,472.48	19,663.78	14,193.62	5,096.54	47,088.09	229,005.95

** Special notes for project a3: Capital costs are 1000 in period 1 and 2200 in period 2, for a present value of 3000 with a 10% interest rate. Since risks do not begin to change until the beginning of period 3 (end of period 2), it is necessary to discount any risk changes back to the beginning of period 1

Table 9. Evaluation of Emission Reduction Options for Lower Range Risk Assessment Assumptions (Column 4 of Table 7)

	a1	a2	a3 **	r1	r2	r3
1. Capital Expenditures (K)	0.00	9,314.00	3,000.00	95.00	56.10	1,605.80
2. Annual O&M Costs (C)	647.00	414.00	0.00	10.10	22.50	89.00
3. Annual Risk Reduction (dR)	123.84	10.80	6.48	1.80	16.20	79.20
4. Present Value of Costs (pvc)	3,975.53	11,857.85	3,000.00	157.06	194.35	2,152.67
5. P= Cost Per Unit Risk Reduction (d = 0)	3.21	109.79	46.30	8.73	1.20	2.72
6. P= Cost Per Unit Risk Reduction (d = 10%)	5.22	178.69	82.88	14.20	1.95	4.42
7. Value of a Statistical Life (VOSL)	171.00	171.00	171.00	171.00	171.00	171.00
8. Net Present Value of Project (NPV)	126,145.75	-510.06	3,189.70	1,734.24	16,827.33	81,064.43

** Special notes for project a3: Capital costs are 1000 in period 1 and 2200 in period 2, for a present value of 3000 with a 10% interest rate. Since risks do not begin to change until the beginning of period 3 (end of period 2), it is necessary to discount any risk changes back to the beginning of period 1.