10 Intake Fraction

Julian D. Marshall
University of British Columbia

William W. Nazaroff
University of California

CONTENTS

10.1 SYNOPSIS

Intake fraction, a metric that summarizes the emission-to-inhalation relationship, facilitates comparisons among sources in terms of their exposure potential. For a given emission source and pollutant, intake fraction is the cumulative mass inhaled by the exposed population divided by the cumulative emissions. One way to estimate the environmental health impact of a pollution source or source class is as the product of three terms: emission rate (mass per time), intake fraction (mass inhaled per mass emitted), and toxicity (health impact per mass inhaled). In the ideal situation, one would know all three terms for all major emission sources. However, important insight can be gained even without complete information. For example, if two sources are identical except that the intake fraction is twice as high for source A as for source B, then the health benefit per mass emission reduction is expected to be twice as large for A as for B.

Intake fraction is a metric not a method. Values of the intake fraction can be determined from models or from measurements. Typical values for the intake fraction are as low as 0.1 per million for releases to outdoor air in remote rural areas, roughly 10 per million for releases near ground level in urban areas, and roughly 5,000 per million for indoor releases in occupied buildings. Thus, releases to indoor air have roughly 500 times as great an intake fraction as for outdoor releases to urban air. In other words, a gram released indoors while painting your living room is, from the standpoint of population exposure, roughly equivalent to half a kilogram released into the urban atmosphere from the paint plant.

This chapter illustrates the use of a simple model, the one-compartment box model, to estimate intake fraction values and compare values among types of sources. Examples are included of how one might compare intake fraction values for two sources and then use this information to prioritize emission reductions. For example, since intake fraction values are expected to be higher in urban areas than in rural areas, all else being equal, the health benefits attributable to an emission reduction are expected to be greater if that emission reduction occurs in an urban area than if it occurs in a rural area. As another example, because people are, on average, closer to on-road emissions than to other ambient sources, emission reductions targeted at on-road sources will have a greater health impact per mass emission reduction than reductions targeted at other ambient sources. As a third example, "self-pollution" of school buses, whereby a small fraction of emissions migrate inside the same vehicle that generated the pollution, has the potential to greatly increase intake fractions. Mitigating self-pollution represents "low-hanging fruit" in terms of exposure control: the mass of pollution inhaled could be significantly reduced even without reducing emissions.

Intake fraction can be useful in a variety of situations where a summary of the emission-to-inhalation pathway is needed. It can be used in cost-effectiveness analyses to compare emission reduction options in terms of the cost per gram inhaled rather than the cost per gram emitted. Because it can be disaggregated according to who inhales the pollution, intake fraction may also be useful when considering environmental justice concerns related to the distribution of exposure concentrations among the population.

10.2 INTRODUCTION

The effectiveness of air pollution control measures may be evaluated in terms of changes in emissions rates, using measures such as tons per year. Indirectly, the effects of such reductions may be observed through changes in ambient air concentrations as measured at ambient monitoring stations. It might be assumed that decreases in ambient air concentrations cause commensurate decreases in human exposure. However, this is not necessarily the case, because personal exposures can vary substantially from what ambient air monitors indicate. For example, measured ambient benzene concentrations decreased in the area of Los Angeles, California, from 1989–1997 by a factor of four, from 4 to 1 ppb. However, exposure concentrations were calculated to have decreased by only a factor of three, from 6 to 2 ppb (Fruin et al. 2001). In this case, only about half of the exposure reductions occurred because of reductions in ambient air concentrations. Other contributions came from reduced exposure to environmental tobacco smoke and from decreased benzene concentrations in cars and garages, improvements that would not be detected at ambient monitoring stations.

This chapter presents ideas about how to prioritize emission reductions based on their effectiveness in reducing exposures. These include considering the location of the emissions source, the surrounding population densities, and the factors affecting dilution of the emissions.

10.3 BACKGROUND

There are many sources of environmental pollution. Important environmental health goals include identification of sources, estimation of personal and population exposures, and effective prioritization of emission reductions. Scientific and engineering analyses are needed to determine which sources to mitigate and by how much. Because the most important reason for regulating air pollution is to reduce its adverse effects on public health, environmental health impact is a logical basis for prioritizing emission reductions. While this chapter focuses on pollutant emissions to air, similar approaches are applicable to groundwater, surface water, and soil pollution.

One way to estimate the environmental health impact of an air pollution source or source class is as the product of three terms: emission rate (mass per time), intake fraction (mass inhaled per

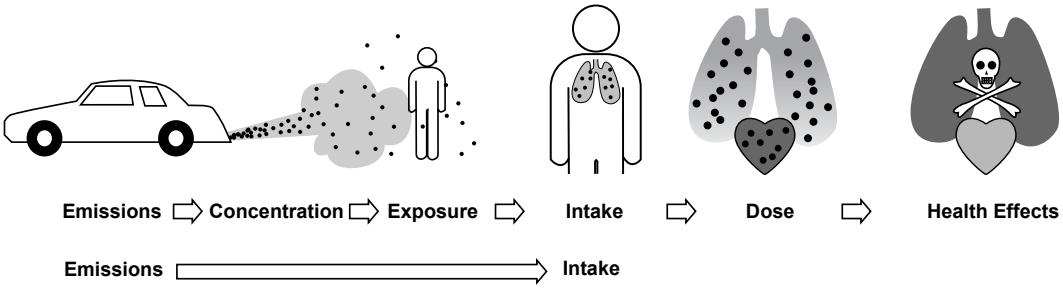

FIGURE 10.1 The air-pollution emission-to-effects paradigm. (After Smith 1993.)

mass emitted), and toxicity (health impact per mass inhaled). In the ideal situation, one would know all three terms for all major emission sources. However, as we describe below, one can gain important insight even without complete information. This chapter focuses on the second term in this relationship (intake fraction).

10.4 WHAT IS INTAKE FRACTION?

Intake fraction summarizes in a compact and transparent form the relationship between emissions and inhalation of these emissions. Intake fraction is useful in connecting emissions to effects because mass inhaled is a much better indicator of potential adverse health impacts than either mass emitted or airborne concentration.

The emission-to-effects relationship involves a series of causally related steps. As illustrated in Figure 10.1, emissions are transported and transformed to generate pollutant concentrations that generally vary in space and time. Human encounters with concentrations constitute exposures, and inhalation of pollutants results in intake. Pollutant transfer into the body of an exposed individual leads to doses to physiological targets, such as organs, which in turn can elevate the risk of adverse health effects. Intake fraction quantitatively summarizes an important portion of this chain of events by describing the emission-to-intake relationship as a single number.

Intake fraction should be understood to be a metric not a method. Like emissions and concentrations, intake fraction can be determined through several different methods. Investigations that generate intake fraction results can range from simple to complex and can rely on modeling or on experimental measurement.

Intake fraction for a primary pollutant is the total mass inhaled from an emission source divided by the total mass emitted from that source. The emission source evaluated in the denominator can be a single emitter, such as an industrial stack or a cigarette, or a broad source class, such as motor vehicles or household cleaning products. When considering an entire population, the value of the numerator would be the cumulative mass inhaled by all exposed individuals. When considering a subpopulation or an individual, the value in the numerator would be the mass inhaled by that subpopulation or individual. Mass inhaled can be determined as the average intake rate multiplied by exposure duration.

$$\text{intake fraction} = \frac{\text{mass inhaled}}{\text{mass emitted}}$$

The expression can be evaluated in terms of cumulative intake per unit emissions for a release episode. In this case, both numerator and denominator would have units of mass. Or, for processes that continuously emit pollutants, the intake fraction can be evaluated as the ratio of the time-averaged inhalation rate to the time-averaged emission rate. In this case, both numerator and

denominator would have units of mass per time. Intake rate (mass per time) can be evaluated as exposure concentration (mass per volume) times breathing rate (volume per time). In any event, the intake fraction is a dimensionless ratio that reflects the fraction of pollution released to the environment that is taken in by an exposed population.

Intake fraction depends on many parameters that influence the emission-to-intake relationship, such as whether the emission occurs indoors or outdoors. Therefore, intake fraction can vary with location and over time. For example, if two outdoor emission sources emit the same mass of pollution, but one source is in a densely populated urban area while the other is in a rural area, the first source will have a higher associated intake fraction because there are more people in the vicinity of the emissions. For a given indoor or outdoor emission source, the intake fraction varies with the pollutant's removal rate from the environment of concern. For ground-level outdoor emissions, intake fraction is smaller during periods of rapid mixing and dispersion than during stagnant air conditions. For indoor emissions into a building of a given size, intake fraction is smaller for a high air-exchange rate (a "leaky" building) than for a low air-exchange rate (a "tight" building).

One important attribute of intake fraction is that it can be applied to groups of pollutants, rather than only to specific species. For example, if two pollutants are emitted from the same source and have the same fate and transport characteristics, then their intake fraction values will be the same, even if their chemical composition and mass emission rates differ. Consider emissions from passenger vehicles in a specific urban environment. To the extent that $PM_{2.5}$ from gasoline-powered motor vehicles behaves like a conserved (nonreacting) pollutant, then its intake fraction would be similar to the intake fraction of other conserved pollutants from motor vehicles, such as carbon monoxide. Similarly, the indoor fate and transport of acrylonitrile and 1,3-butadiene, two toxic chemicals found in environmental tobacco smoke (ETS), are similar, indicating that their intake fractions associated with ETS would be similar. This characteristic offers the promise of efficiency in the use of intake fraction. One can envision that as more studies of intake fraction are completed, a compendium of intake fraction results could be compiled that would provide useful guidance about expected values for sources not yet assessed.

10.5 TYPICAL INTAKE FRACTION VALUES

Intake fraction values vary over several orders of magnitude. Three important factors affecting intake fraction are the size of the exposed **population**, the **proximity** between the emission source and the exposed population, and the **persistence** of the pollutant in the air parcel. These three factors are informally known as "the three P's." Typical intake fraction values for some release categories are presented in Figure 10.2.

For outdoor releases in rural or urban areas, intake fraction values are typically in the range 0.1–100 per million. An intake fraction of 1 per million means that for every kilogram emitted, 1 mg is collectively inhaled. This intake fraction value also means that to reduce inhalation intake by 1 mg would require reducing emissions by 1 kg. Intake fraction values are much higher for indoor releases than for outdoor releases because dilution and dispersion rates are lower indoors than outdoors. A typical intake fraction difference between indoor and outdoor releases can be as large as three orders of magnitude. This result leads to the "rule of 1,000," which states that from an inhalation intake standpoint, 1 gram of emissions to indoor environments is roughly equivalent to 1000 grams of emissions to outdoor environments.

The term "intake fraction" was first introduced in the literature in 2002 (Bennett et al. 2002a). However, there is a much longer history of the idea of quantitatively relating pollutant emissions to inhalation intake, as reviewed by Bennett et al. (2002a) and by Evans et al. (2002). Smith and colleagues have explored the policy implications of intake fraction and related concepts (e.g., Smith and Edgerton 1989; Smith 1995, 2002). Although not yet large, the literature on intake fraction is diverse, addressing primary and secondary pollutants, inhalation and other intake pathways, and varied sources such as motor vehicles, power plants, and dry cleaners (see Table 10.1).

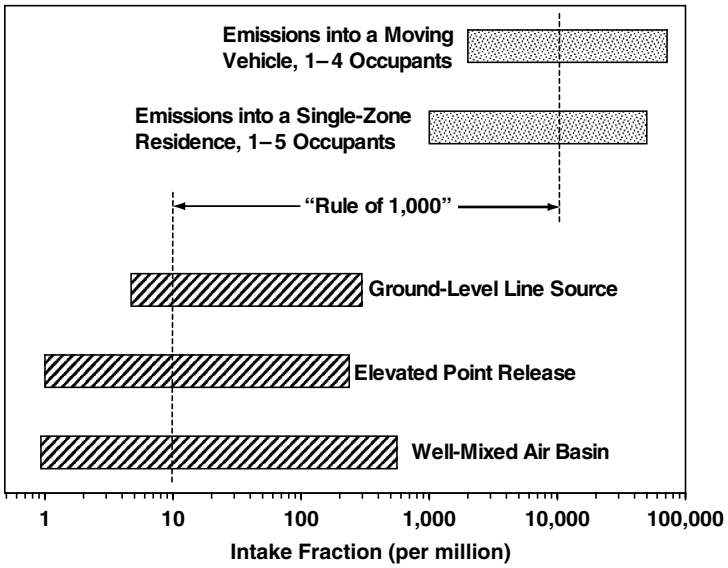

FIGURE 10.2 Typical intake-fraction values for nonreactive air pollutants emitted from different source classes. (Data from Lai, Thatcher, and Nazaroff 2000.) The upper two bars represent indoor emissions and the lower three bars correspond to outdoor emissions. Within each category, considerable variability is possible, depending largely on population exposed, their proximity, and the persistence of the pollutants. The difference in central tendency between the bars is ~ 3 orders of magnitude, as illustrated by the arrow labeled "rule of 1,000."

Broadly, there are two approaches for quantifying the emission-to-intake relationship: models and measurements. During the past several decades, much work in air-quality engineering has developed and used these approaches to understand emission-to-airborne concentration relationships. The methods developed and the results obtained can also be used to inform the emission-to-intake relationship. Models range from simple, one-compartment representations of a household or an urban area, to complex, three-dimensional urban airshed models. Measurement methods include experiments involving the deliberate release of a tracer gas as well as utilization of "tracers-of-opportunity" (i.e., chemical compounds that act as a "fingerprint" for an emission source).

The following factors have been found to have important influence on intake fractions:

- Whether a release occurs within a confined space (indoors) or into open air
- Population density and size of the exposed population in the vicinity of the release
- Meteorological conditions controlling air dispersion, such as wind speed and mixing height, or analogously for indoor releases, the ventilation rate of a building
- Pollutant persistence, depending on the rate of mechanisms such as deposition
- Dominant exposure pathway (inhalation, dermal absorption, ingestion) and emission media (air, soil, surface water, groundwater)
- Transformations such as bioaccumulation (for ingestion) and secondary formation (for air pollutants)

10.6 ESTIMATING INTAKE FRACTION VALUES USING A ONE-COMPARTMENT MODEL

In this section we demonstrate how a one-compartment box model can be used to estimate inhalation intake fraction. The one-compartment model is straightforward, produces reasonable quantitative intake fraction estimates, and provides insight about the dependence of intake fraction on key

TABLE 10.1
Recently Published Studies Reporting Evaluations of Intake Fractions

Sources	Pollutants	Media	Pathways	Methods	References
Dry cleaners	Perchloroethylene	Air	Inhalation	Dispersion model, box model	Evans, Thompson, and Hattis (2000)
General	Organic pollutants	Air, water, soil, food	Inhalation and ingestion	Multimedia model	Bennett et al. (2002b)
General	Benzene, carbon tetrachloride, benzo[a]pyrene, and dioxin	Air, water, soil, food	Inhalation and ingestion	Multimedia model	MacLeod et al. (2004)
General	PCDDs/DFs and PCBs	Air, water, soil, food	Inhalation and ingestion	Multimedia model, data analysis	Hirai et al. (2004)
General	Semivolatile organics	Air, water, soil, food	Ingestion	Multimedia model	Lobscheid, Maddalena, and McKone (2004)
General air emissions	Primary air pollutants	Air	Inhalation	Box models, dispersion models	Lai, Thatcher, and Nazaroff (2000)
General air emissions	PCDDs/DFs	Air, water, soil, food	Ingestion	Multimedia model, data analysis	Margni et al. (2004)
Motor vehicles	Primary pollutants (CO, benzene)	Air	Inhalation	Data analysis of tracers of opportunity	Marshall et al. (2003)
Motor vehicles	Primary pollutants	Air	Inhalation	Models	Marshall, Teoh, and Nazaroff (2005)
Motor vehicles: school buses	Primary pollutants	Air	Inhalation	Tracer gas	Marshall and Behrentz (2005)
Power plants, motor vehicles	Primary and secondary PM	Air	Inhalation	Dispersion model	Levy, Wolff, and Evans (2002)
Power plants motor vehicles	Primary pollutants	Air	Inhalation	Dispersion model	Nigge (2001)
Power plants	SO_2, sulfate	Air	Inhalation	Dispersion model	Hao et al. (2003)
Power plants	Primary and secondary PM	Air	Inhalation	Dispersion model	Levy et al. (2003)
Power plants	Primary and secondary PM	Air	Inhalation	Dispersion model	Li and Hao (2003)
Power plants	SO_2, primary and secondary PM	Air	Inhalation	Dispersion model	Zhou et al. (2003)
Power plants, distributed electricity generation	Primary pollutants ($PM_{2.5}$, formaldehyde)	Air	Inhalation	Dispersion model	Heath, Hoats, and Nazaroff (2003)

Note: PCDD = polychlorinated dibenzo-dioxins; DF = dibenzo-furans; PCB = polychlorinated biphenyls.

parameters. We begin by applying this model to indoor release of a nonreactive pollutant in a household, and then apply it to similar conditions in an urban area.

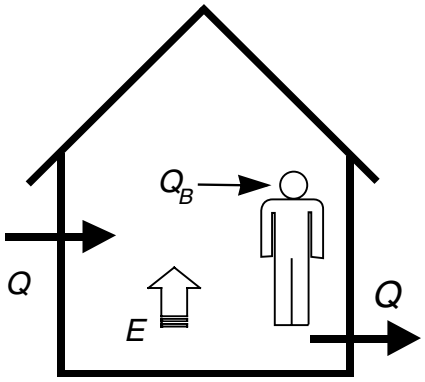

FIGURE 10.3 Schematic for estimating intake fraction for indoor release of a nonreactive pollutant. The indoor environment is treated as well mixed, meaning that the pollutant concentration is assumed to be uniform throughout the indoor air.

For the case of a residence, as illustrated in Figure 10.3, the parameters needed to determine intake fraction are the ventilation rate of the building (Q, units: $m^3\ h^{-1}$), the number of individuals exposed (P, units: person), and the average volumetric breathing rate (Q_B, units: $m^3\ h^{-1}\ person^{-1}$). The ventilation rate, Q, can be represented as the product of the house volume (V, units: m^3) and the air exchange rate (X, units: h^{-1}). If the pollutant release rate (E, units: $\mu g\ h^{-1}$) and the ventilation rate are constant, and the pollutant is nonreactive and well mixed throughout the indoor air, then the steady-state concentration of the contaminant attributable to the indoor release is simply

$$C = \frac{E}{Q} . \tag{10.1}$$

With P individuals in the space, each breathing at an average volumetric rate Q_B, the cumulative mass inhalation rate of pollutants owing to the indoor source, I, is

$$I = CPQ_B = \frac{EPQ_B}{Q} . \tag{10.2}$$

The intake fraction is the intake rate, I, normalized by the emission rate, E:

$$\text{iF} = \frac{PQ_B}{Q} = \frac{PQ_B}{VX} . \tag{10.3}$$

In this model, a pollutant emitted inside can have one of two fates: either it is inhaled, or it exits via air exchange. Intake fraction is the probability that an emitted pollutant is inhaled rather than removed by ventilation. Although Equation 10.3 was derived for steady-state conditions with constant emissions and ventilation, the same relationship holds for the intake fraction of episodic releases (Nazaroff 2006).

To generate quantitative estimates of intake fraction, we next consider typical values for each of the terms in Equation 10.3. Estimates in the literature of population-average breathing rate, Q_B, vary. Commonly used values for daily-average conditions (units: $m^3\ person^{-1}\ d^{-1}$) are 12 (Layton 1993; USEPA 1997), 15 (Marty et al. 2002), and 17 (OEHHA 1996). Air-exchange rates vary among houses and over time. In tropical climates, homes can be constructed with large designed openings for natural ventilation. On a windy day, the air exchange rate in such a building may be

of the order of 10 h^{-1}. In contrast, a modern building in a cold climate may be well sealed, having an air exchange rate as low as of the order of 0.3 h^{-1}. The median value for single family homes in the United States is 0.5 h^{-1} (Murray and Burmaster 1995). Of course, the number of occupants in a residence can vary widely. In the United States, there are an average of three persons living in each occupied household (U.S. Census 2004). The median household size is 160 m^2 (U.S. Census 2004); assuming a ceiling height of 2.4 m, this corresponds to a volume of approximately 400 m^3. Building volume and indoor population can be significantly larger for nonresidential buildings, such as shopping centers, restaurants, and offices, than for residences. However, in temperate climates the ventilation rate per occupant (XV/P) is of consistent magnitude across major building classes. Consequently, intake fraction values are expected to be consistent in magnitude across different building types.

Using the values Q_B = 12 m^3 $person^{-1}$ d^{-1}, P = 3, X = 12 d^{-1} (= 0.5 h^{-1}), and V = 400 m^3, the intake fraction for releases into a residence is estimated to be 0.75%, or 7,500 per million. That is, in this case people would inhale 7.5 mg per gram of pollutant emitted into an indoor environment. This estimate assumed constant occupancy of the indoor space. Taking into account that people only spend about two thirds of their time indoors in their own residences (Klepeis et al. 2001), a better estimate for the central tendency of intake fraction in U.S. residences might be 0.5% or 5,000 per million. This correction would only be expected to apply for certain emission sources that emit continuously, such as furnishings that emit volatile organic compounds. Many important sources, such as cooking, cleaning, and cigarette smoke, only occur when people are present. For these sources, the time-activity correction of two thirds may not apply.

As a comparison, the box model can also be applied to an urban area. The air inflow rate, Q, can be estimated as the product of the wind speed (u, units: m s^{-1}), the height of the atmospheric mixing layer (H, units: m), and the width of the urban area (W, units: m). For the United States, a typical value for the term uH is 42 million m^2 d^{-1}, and the population-weighted median value for the term P/W in U.S. metropolitan areas is 43 people m^{-1} (Marshall, Teoh, and Nazaroff 2005). The ratio of these two groups of terms indicates that a typical value of outdoor "ventilation" of urban areas per person in the United States is $Q/P \sim 10^6$ m^3 $person^{-1}$ d^{-1}. Combining this result in Equation (10.3) with Q_B = 12 m^3 $person^{-1}$ d^{-1} yields typical intake fraction for outdoor urban releases of 12 per million, which is roughly 400 times smaller than the value of 5,000 per million estimated for indoor releases. We can see from this comparison that the "rule of 1,000" is only an approximation, providing a magnitude estimate of the difference in intake fractions between indoor and outdoor releases. In large urban areas, the difference between indoor and outdoor releases can be less than 1,000. In small urban areas or in rural areas, the difference can be more than 1,000. In this particular comparison, the factor of 400 reflects the difference in per-capita ventilation rates that is available to cause dilution of pollutant emissions, comparing the amount of ventilation that is provided to buildings with the amount that nature provides through winds that ventilate urban air basins.

10.7 THE USE OF INTAKE FRACTION IN PRIORITIZING EMISSION REDUCTION EFFORTS

There are many situations in which one can use intake fraction to help in prioritizing air pollution control policies without full information about emissions and toxicity. Each of the next four paragraphs presents a situation wherein different pieces of information are available. In each case, we assume that there are two emission sources, and that the question at hand is how to prioritize between these two sources as the target of emissions reduction policies. We further assume that the pollutant of concern exhibits a linear, no-threshold dose-response relationship. This assumption, which is commonly applied in public health protection for carcinogens and certain other pollutants, means that the adverse health impact associated with a source scales in direct proportion to the intake, regardless of how that intake is distributed within the population. Not all important air

pollution problems can be treated in this way. The intake fraction can be useful even when this treatment is not possible, but a more sophisticated assessment is required.

1. When all three terms — emissions, intake fraction, and toxicity — are known, one can estimate the overall health impact from the two sources. The source with the higher health impact would be identified as a higher priority for control. If information about the costs of control technologies is also known, then one could prioritize emission reductions based on a cost-effectiveness analysis. In this case, one would seek to maximize the reduction in adverse health effects per unit cost (Smith and Edgerton 1989, Smith 1995).
2. When only emissions and intake fractions are known, one could prioritize emission source reductions based on total emissions, but using the intake fraction values as multipliers. For example, considering a specific pollutant, if the intake fraction is two times greater for emission source *A* than for emission source *B*, an emission reduction of 1 kg from *A* could be given the same policy "priority" as an emission reduction of 2 kg from *B*. Assuming pollutants have the same toxicity regardless of the emission source (which is usually the case), then comparisons between sources can be made assuming that inhalation intake is a suitable proxy for adverse effects.
3. When only intake fractions and control costs are known, one can carry out certain cost-effectiveness analyses. For example, if control costs per kg *emitted* are the same for emission sources *A* and *B*, but the intake fraction is larger for *A*, then the control cost per kg *inhaled* is less for *A* than for *B*.
4. Finally, when only intake fractions are known, one can compare sources that are similar. For example, comparing natural-gas power plants in different locations, one could prioritize for control the emissions location with the higher intake fraction. This intake fraction difference could be attributable to a variety of factors, including the size of the exposed population, proximity between the population and emissions, and meteorology.

In prioritizing among the thousands of emission sources, it may be useful to group sources into broad classes, such as indoor vs. outdoor, stationary vs. mobile, urban vs. rural, or — for motor vehicles — diesel-powered vs. gasoline-powered. The adverse impact for each source class could be estimated as proportional to the product of the total mass emissions times an emission-weighted estimate of intake fraction. Information on toxicity could be incorporated into this approach independently, to the extent that it is available.

Below are two examples of how intake fraction might be used to prioritize emission source reductions: on-road vs. off-road sources and emissions sources with self-pollution.

10.7.1 Particulate Matter from On-Road Sources

People in urban areas typically spend some time in or near vehicles each day. Particulate matter (PM) concentrations are several times higher, on average, in vehicles than in buildings for three reasons: (1) vehicles are an important source of PM emissions, (2) the in-vehicle environment is closer than the indoor environment to vehicle emissions, and (3) buildings offer more protection than vehicles against outdoor PM. These near-source (in-vehicle) exposures increase the intake fraction associated with on-road emissions as compared with off-road emissions.

For example, focusing on urban diesel PM emissions, one can use published data to estimate the typical intake fraction differences between on-road sources and other sources. Recent measurements for diesel vehicles in California suggest a factor of 4–14 difference between in-vehicle and nearby ambient concentrations (Fruin, Winer, and Rodes 2004). The result is that the ~6% of time (80 minutes per day) spent in vehicles (Klepeis et al. 2001) contributes ~25–54% of total exposure to diesel PM, rather than 6%. If it is assumed that 25% of California's diesel PM emissions are

from on-road sources (CARB 2000), then on-road sources contribute ~39–63% (rather than 25%) to total diesel PM exposure. On average, on-road sources are estimated to contribute between 1.9 and 5.1 times more diesel PM inhalation per unit emissions than other sources. Thus, from an exposure standpoint, on-road diesel particle emissions should be given a "weighting" of ~2 to 5 relative to other diesel sources. The width of this range reflects uncertainty in diesel PM concentrations in vehicles and in the ambient environment.

10.7.2 Self-Pollution

Combustion sources often possess an exhaust system to deliver emissions to ambient air. The exhaust manifold of a car conducts effluents from the engine to the tailpipe; wood stoves emit their effluents through a chimney that runs from the fireplace to the rooftop. Generally, exhaust systems work well but not perfectly, and a small fraction of emissions can enter the indoor or in-vehicle environment. Such leaks lead to a condition known as "self-pollution."

Intake fractions for pollutant releases into moving vehicles are comparable to those for residences (Lai, Thatcher, and Nazaroff 2000). Because of the "rule of 1,000," even a small amount of self-pollution — on the order of 0.1% of emissions or higher — can significantly increase the intake fraction associated with sources like motor vehicle exhaust and residential wood combustion.

School buses are a good example of the high impacts possible from self-pollution. Tracer-gas experiments indicate that the self-pollution intake fraction is significant for school buses in California, to a degree that depends on the age of the bus. The self-pollution intake fraction reported by Marshall and Behrentz (2005) was 70 per million for the oldest bus investigated (model year: 1975), and averaged 20 per million for the remaining five buses (model years between 1985 and 2002). These values are larger than typical outdoor intake fractions (e.g., a typical value of 12 per million was derived above). On the basis of this work it is estimated that in a typical U.S. urban area, the cumulative mass of a school bus's emissions inhaled by the roughly 40 students on that bus is larger than the cumulative mass of pollution from that school bus inhaled by all of the other exposed individuals. Addressing bus self-pollution could markedly reduce the inhalation intake of diesel PM, even before bus emission reductions are achieved. More broadly, controlling self-pollution wherever it occurs offers the potential to be "low-hanging fruit" for effectively achieving exposure reduction.

10.8 USING INTAKE FRACTION WHEN CONSIDERING ENVIRONMENTAL JUSTICE CONCERNS

Understanding and addressing distributional issues related to air pollution exposure is an important aspect of air quality management. Air pollution control policies need not only reduce the total health impact of emissions, but also ensure that the distribution of burden among the population is not unfair or unjust. Throughout most of this chapter, intake fraction has been based on the total population intake. However, intake fraction can be estimated for population subgroups, even down to the level of individuals. So, for example, if the population is divided into a set of groups, the total population intake fraction can be considered as the sum of the partial intake fractions associated with each group. One indicator of environmental justice would be the degree to which partial intake fractions per capita are consistent among different demographic indicators. Heath, Hoats, and Nazaroff (2003) considered how intake fraction for an electricity-generation station depends on the station's location. They presented, for specific locations, the percentage of the downwind population that is white vs. non-white, and the percentage of total intake that occurs in the white and non-white populations. In three of the five case studies, non-white populations were significantly more exposed than white populations. For example, for a hypothetical small-scale electricity generator

located in downtown Los Angeles, 32% of the exposed (i.e., downwind) population is non-white, but this population would receive 69% of the total intake (Heath, Hoats, and Nazaroff 2003).

There are several attributes that can be used to divide the population into groups that could be relevant in considering air pollution exposure aspects of environmental justice, including ethnicity, gender, neighborhood, income, age, and health status. As information emerges about different degrees of susceptibility of demographic subgroups to air pollution exposure, intake fraction analyses could also be conducted to highlight the levels of exposure that these subgroups encounter in relation to the other parts of the population. By doing so, additional control efforts could be targeted at protecting those who are most vulnerable to air pollution.

10.9 ACKNOWLEDGMENTS

The Research Division of the California Air Resources Board provided partial funding for the work described in this chapter. The contents are the opinions of the authors and do not represent the views of the Air Resources Board. Dr. Scott Fruin provided helpful review comments and suggestions on a draft of this work. Figure 10.1 was generated by The Linus Group.

10.10 QUESTIONS FOR REVIEW

1. What is meant by the "three P's" and the "rule of 1,000"? [Answer: The three P's are *population*, *proximity*, and *persistence*. These are three variables that have a large influence on intake fraction. The *rule of 1,000* says that typical intake fraction values are roughly 1,000 times larger for indoor releases than for outdoor releases. This means that indoor releases are roughly 1,000 times more efficient at delivering their dose to people than are outdoor releases.]
2. The health risk attributable to an emission source can be estimated as the product of three parameters. One of these three parameters is intake fraction. What are the other two parameters, and what are the units on each of the three terms? [Answer*:* The three terms are emissions (units are mass, or mass per time); intake fraction (mass inhaled per mass emitted, effectively dimensionless); and toxicity (health impact per mass inhaled).]
3. The outdoor intake fraction value in the text (12 per million) is based on the value P/W = 43 people m^{-1}, where P is the population and W is the width of the metropolitan area. Using equations given in this chapter, perform a similar calculation: estimate intake fraction values in the following table using the given values for P and W. We have used the square root of the land area as an estimate of W.

Area	Population (thousands)	Land Area (km^2)	P/W (people m^{-1})	Intake Fraction (mg inhaled per kg emitted)
Los Angeles, CA	12,400	5,800	163	
Sacramento, CA	1,510	990	48	
Rural NV	170	280,000	0.3	
Area where you live				

How well does the one-compartment intake fraction estimate for Los Angeles compare with the published estimate of 48 per million (Marshall et al. 2003)? [Answer: The results are presented in the table below. Note the wide variation between sparsely populated

rural Nevada and the populous Los Angeles area. The estimate of 58 per million for Los Angeles is ~20% larger than the published value of 48 per million. Given the considerable simplification inherent in the one-compartment model, this level of agreement is considered good.]

Area	Population (thousands)	Land Area (km^2)	P/W (people m^{-1})	Intake Fraction (mg inhaled per kg emitted)
Los Angeles, CA	12,400	5,800	163	58
Sacramento, CA	1,510	990	48	17
Rural NV	170	280,000	0.3	0.1
Area where you live				

4. The one-compartment model presented in this chapter can be modified to account for pollutant removal mechanisms other than advection (or ventilation). For example, pollutant removal via deposition can be characterized in terms of a deposition velocity (v_d, units: m s^{-1}) onto a surface of area A (units: m^2). The one-compartment intake fraction equation, accounting for deposition, is

$$\text{iF} = \frac{PQ_B}{Q + Av_d}$$

Consider an urban area for which the following values apply: P = 1 million people; Q_B = 15 m^3 $person^{-1}$ d^{-1}; $Q = 10^{12}$ m^3 d^{-1} (i.e., 1 trillion m^3 d^{-1}); A = 700 million m^2. Assume deposition velocities (units: cm s^{-1}) for fine particulate matter (PM) and for coarse PM are 0.03 and 3, respectively. What are the intake fractions for fine PM and coarse PM, and how do these values compare to the intake fraction for a non-depositing conserved pollutant? (*Hint:* A non-depositing pollutant has a deposition velocity of zero.) [Answer: Intake fractions values (units: per million) for fine PM and coarse PM are 14.7 and 5.3, respectively. These values are 2% and 64% less, respectively, than the intake fraction for a non-depositing pollutant (15.0 per million).]

5. A specific household contains two combustion sources: an incense stick and a fireplace. Assume that in 1 hour, 3 grams of incense (about 2 incense sticks) are burned and 3 kg of wood are burned. Assume that 0.3% of the incense mass is emitted as fine particles and that 0.03% of the wood mass is emitted as fine particles. Finally, assume that all incense emissions are to the indoor space, while only 0.5% of the wood emissions enter the indoor space as self-pollution. (The remaining 99.5% of the wood emissions exit via the chimney.) What are the total mass emission rates of fine particles *to the indoor space* for the two sources, in units of mg per hour of use? Using an intake fraction of 1%, what is the total mass of incense particle emissions and of wood particle emissions inhaled, in units of mg per hour of use? What steps could be taken to reduce mass inhalation values? [Answer: The indoor mass emission rates (units: mg per hour of use) are 9 and 4.5, respectively, for the incense emissions and for the wood emissions. The mass inhalation rates (units: mg per hour of use) are 0.09 and 0.045, respectively, for the incense emissions and for the wood emissions. To reduce the intake rate, one can reduce the mass emission rate or the intake fraction. Steps that would reduce the mass emission rate include using low-smoke incense; altering the wood combustion conditions to reduce particle formation rates; and, reducing the mass of fuel combusted. Steps to reduce the intake fraction include separating the emissions and the people (e.g., go in the other room and shut the door); increasing the ventilation rate (e.g., crack open a

window); and, reducing the self-pollution rate (e.g., fully enclose the wood fire to ensure that all of the smoke exits via the chimney rather than into the room).]

6. The "rule of 1,000" indicates that indoor sources are roughly 1,000 times more potent than outdoor sources at delivering inhalation intake to people. Similarly, calculations in the section "Particulate Matter from On-Road Sources" indicate that on-road PM emissions are approximately two to five times more effective than off-road PM emissions at delivering inhalation intake to people. In other words, the intake fraction for PM is two to five times larger for on-road emissions than for off-road emissions. A similar calculation indicates that the intake fraction for nonreactive gas molecules, such as benzene and carbon monoxide, is ~25% higher for on-road emissions than for off-road emissions. Derive this ~25% difference, using the following assumptions:

   - 66% of emissions are on-road, and the remainder are off-road
   - People spend 80 minutes per day in a vehicle
   - Concentrations are 4 times higher in-vehicle than not-in-vehicle
   - Buildings do not offer protection against outdoor concentrations (i.e., the indoor concentration attributable outdoor pollution is equal to the outdoor concentration)
   - Note that because we are taking the ratio of two intake fractions, the ~25% value does not depend on the ambient concentration, the volume of air breathed per day per person, or the total emission rate.

   [Answer: Let C = the ambient concentration, Q_B = the volume of air breathed per day, and E = the total emission rate. Note that 80 minutes per day is (80/1440) = 5.6% of the day.
   Mass inhaled per day = Mass inhaled during driving + Mass inhaled not during driving
   = $Q_B(5.6\%)(4C) + Q_B(94.4\%)(C) = (1.17)Q_BC$
   Mass inhaled per day attributable to off-road sources = $0.34Q_BC$
   Mass inhaled per day attributable to on-road sources = $1.17Q_BC - 0.34Q_BC = 0.83\ Q_BC$
   Intake fraction for on-road sources = intake/emissions = $0.83Q_BC/0.66E$
   Intake fraction for off-road sources = intake/emissions = $0.34\ Q_BC/0.34E = Q_BC/E$
   Ratio of these intake fraction values = $(0.83Q_BC/0.66E)/(Q_BC/E) = 0.83/0.66 = 1.26$

   Thus, on-road emissions are ~25% more effective than off-road emissions at delivering inhalation intake.]

REFERENCES

Bennett, D.H., McKone, T.E., Evans, J.S., Nazaroff, W.W., Margni, M.D., Jolliet, O., and Smith, K.R. (2002a) Defining Intake Fraction, *Environmental Science & Technology,* **36**: 206A–211A.

Bennett, D.H., Margni, M.D., McKone, T.E., and Jolliet, O. (2002b) Intake Fraction for Multimedia Pollutants: A Tool for Life Cycle Analysis and Comparative Risk Assessment, *Risk Analysis,* **22**: 905–918.

CARB (2000) *Risk Reduction Plan to Reduce Particulate Matter Emissions from Diesel-Fueled Engines and Vehicles,* Planning and Technical Support Division, Sacramento, California Air Resources Board, Sacramento, CA, http://www.arb.ca.gov/diesel/documents/rrpapp.htm. Accessed April 15, 2006

Evans, J.S., Thompson, K.M., and Hattis, D. (2000) Exposure Efficiency: Concept and Application to Perchloroethylene Exposure from Dry Cleaners, *Journal of the Air & Waste Management Association,* **50**: 1700–1703.

Evans, J.S., Wolff, S.K., Phonboon, K., Levy, J.I., and Smith, K.R. (2002) Exposure Efficiency: An Idea Whose Time Has Come? *Chemosphere,* **49**: 1075–1091.

Fruin, S.A., St. Denis, M.J., Winer, A.M., Colome, S.D., and Lurmann, F.W. (2001) Reductions in Human Benzene Exposure in the California South Coast Air Basin, *Atmospheric Environment,* **35**: 1069–1077.

Fruin, S.A., Winer, A.M., and Rodes, C.E. (2004) Black Carbon Concentrations in California Vehicles and Estimation of In-Vehicle Diesel Exhaust Particulate Matter Exposures, *Atmospheric Environment,* **38**: 4123–4133.

Hao, J.M., Li, J., Ye, X.M., and Zhu, T.L. (2003) Estimating Health Damage Cost from Secondary Sulfate Particles — A Case Study of Hunan Province, China, *Journal of Environmental Sciences*, **15**: 611–617.

Heath, G.A., Hoats, A.S., and Nazaroff, W.W. (2003) *Air Pollution Exposure Associated with Distributed Electricity Generation*, CARB Contract No. 01-341, Department of Civil and Environmental Engineering, University of California, Berkeley, CA, http://ftp.arb.ca.gov/research/apr/abstracts/01–341.htm. Accessed April 15, 2006.

Hirai, Y., Sakai, S., Watanabe, N., and Takatsuki, H. (2004) Congener-Specific Intake Fractions for PCDDs/DFs and Co-PCBs: Modeling and Validation, *Chemosphere,* **54**: 1383–1400.

Klepeis, N., Nelson, W., Ott, W., Robinson, J., Tsang, A., Switzer, P., Behar, J., Hern, S., and Engelmann, W. (2001) The National Human Activity Pattern Survey (NHAPS): A Resource for Assessing Exposure to Environmental Pollutants, *Journal of Exposure Analysis and Environmental Epidemiology,* **11**: 231–252.

Lai, A.C.K., Thatcher, T.L., and Nazaroff, W.W. (2000) Inhalation Transfer Factors for Air Pollution Health Risk Assessment, *Journal of the Air & Waste Management Association,* **50**: 1688–1699.

Layton, D.W. (1993) Metabolically Consistent Breathing Rates for Use in Dose Assessments, *Health Physics,* **64**: 23–36.

Levy, J.I., Wolff, S.K., and Evans, J.S. (2002) A Regression-Based Approach for Estimating Primary and Secondary Particulate Matter Intake Fractions, *Risk Analysis,* **22**: 895–904.

Levy, J.I., Wilson, A.M., Evans, J.S., and Spengler, J.D. (2003) Estimation of Primary and Secondary Particulate Matter Intake Fractions for Power Plants in Georgia, *Environmental Science & Technology,* **37**: 5528–5536.

Li, J. and Hao, J.M. (2003) Application of Intake Fraction to Population Exposure Estimates in Hunan Province of China, *Journal of Environmental Science and Health Part A — Toxic/Hazardous Substances & Environmental Engineering,* **38**: 1041–1054.

Lobscheid, A.B., Maddalena, R.L., and McKone, T.E. (2004) Contribution of Locally Grown Foods in Cumulative Exposure Assessments, *Journal of Exposure Analysis and Environmental Epidemiology,* **14**: 60–73.

MacLeod, M., Bennett, D.H., Perem, M., Maddalena, R.L., McKone, T.E., and Mackay, D. (2004) Dependence of Intake Fraction on Release Location in a Multimedia Framework: A Case Study of Four Contaminants in North America, *Journal of Industrial Ecology,* **8**: 89–102.

Margni, M., Pennington, D.W., Amman, C., and Jolliet, O. (2004) Evaluating Multimedia/Multipathway Model Intake Fraction Estimates Using POP Emission and Monitoring Data, *Environmental Pollution,* **128**: 263–277.

Marshall, J.D. and Behrentz, E. (2005) Vehicle Self-Pollution Intake Fraction: Children's Exposure to School Bus Emissions, *Environmental Science & Technology*, **39**: 2559–2563.

Marshall, J.D., Riley, W.J., McKone, T.E., and Nazaroff, W.W. (2003) Intake Fraction of Primary Pollutants: Motor Vehicle Emissions in the South Coast Air Basin, *Atmospheric Environment,* **37**: 3455–3468.

Marshall, J.D., Teoh, S.K., and Nazaroff, W.W. (2005) Intake Fraction of Nonreactive Vehicle Emissions in US Urban Areas, *Atmospheric Environment,* **39**: 1363–1371.

Marty, M.A., Blaisdell, R.J., Broadwin, R., Hill, M., Shimer, D., and Jenkins, M. (2002) Distribution of Daily Breathing Rates for Use in California's Air Toxics Hot Spots Program Risk Assessments, *Human and Ecological Risk Assessment,* **8**: 1723–1737.

Murray, D.M. and Burmaster, D.E. (1995) Residential Air Exchange Rates in the United States: Empirical and Estimated Parametric Distributions by Season and Climatic Region, *Risk Analysis,* **15**: 459–465.

Nazaroff, W.W. Inhalation Intake Fraction of Pollutants from Episodic Indoor Emissions, in *Building and Environment* (accepted).

Nigge, K.M. (2001) Generic Spatial Classes for Human Health Impacts, Part I: Methodology. *International Journal of Life Cycle Assessment,* **6**: 257–264.

OEHHA (1996) Technical Support Document for Exposure Assessment and Stochastic Analysis, California Office of Environmental Health Hazards Assessment, http://www.oehha.org/airhot_spots/final-Stoc.html. Accessed April 15, 2006.

Smith, K.R. (1993) Fuel Combustion, Air Pollution Exposure, and Health: The Situation in Developing Countries, *Annual Review of Energy and the Environment,* **18**: 529–566.

Smith, K.R. (1995) *The Potential of Human Exposure Assessment for Air Pollution Regulation*, Report No. WHO/EHG/95.9, Collaborating Centre for Studies of Environmental Risk and Development, World Health Organization, Geneva.

Smith, K.R. (2002) Place Makes the Poison: Wesolowski Award Lecture (1999) *Journal of Exposure Analysis and Environmental Epidemiology,* **12**: 167–171.

Smith, K.R. and Edgerton, S.A. (1989) Exposure-Based Air Pollution Control Strategies: Proposed Three-Way Integration, in *Proceedings of the EPA/A&WMA Specialty Conference on Total Exposure Assessment Methodology,* Las Vegas, NV, 630–641.

U.S. Census (2004) American Housing Survey for the United States: 2003, Report No. H150/03, Washington DC, http://www.census.gov/prod/2004pubs/H150-03.pdf. Accessed April 15, 2006.

USEPA (1997) *Exposure Factors Handbook*, Report No. EPA-600-P-95-002Fa, Office of Research and Development, National Center for Environmental Assessment, U.S. Environmental Protection Agency, Washington, DC, http://www.epa.gov/ncea/pdfs/efh/front2.pdf. Accessed April 15, 2006.

Zhou, Y., Levy, J.I., Hammitt, J.K., and Evans, J.S. (2003) Estimating Population Exposure to Power Plant Emissions Using CALPUFF: A Case Study in Beijing, China, *Atmospheric Environment,* **37**: 815–826.